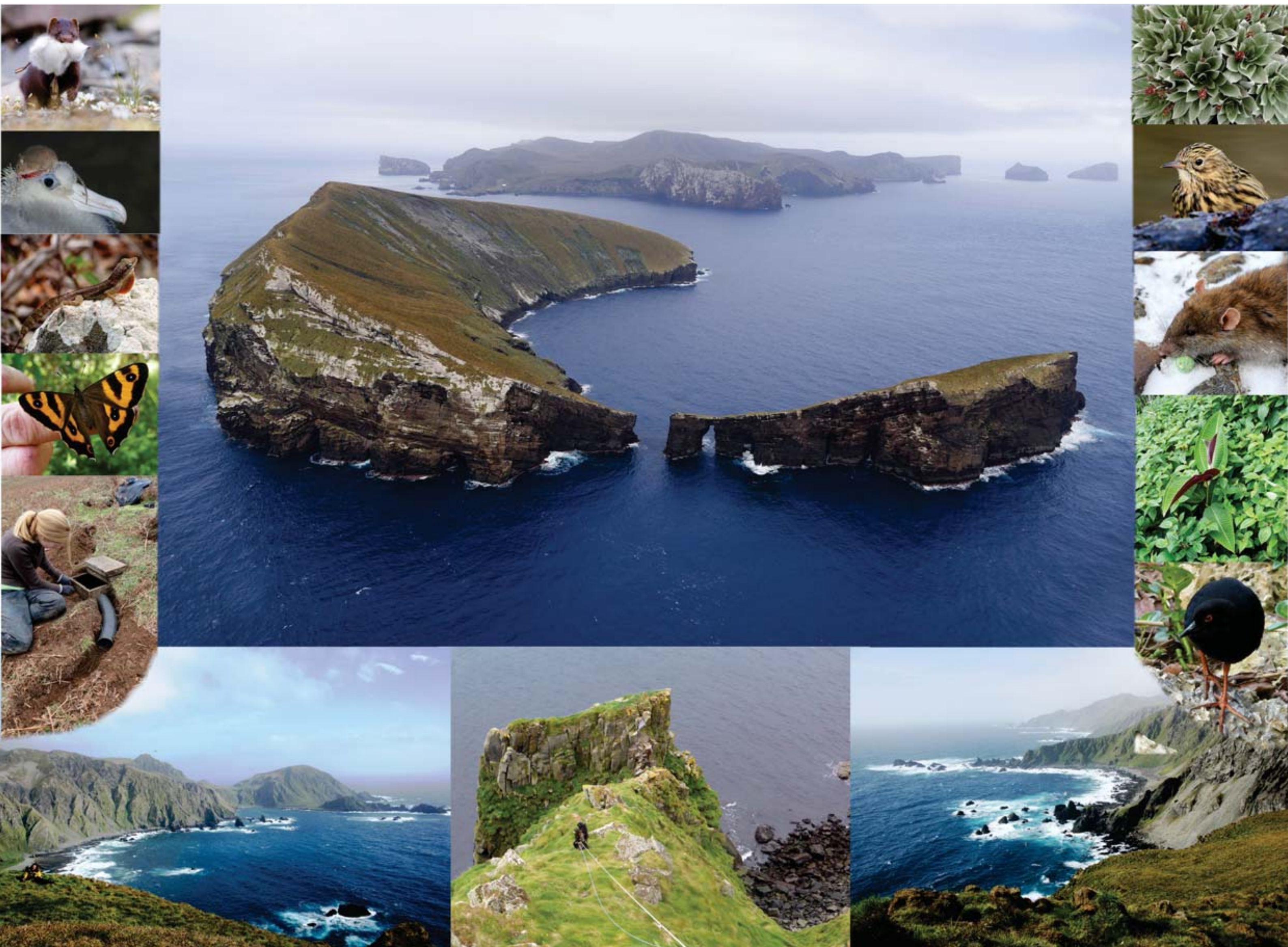
Island invasives: scaling up to meet the challenge

Proceedings of the international conference on island invasives 2017

Edited by C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West



Occasional Paper of the IUCN Species Survival Commission Nº 62

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The papers and abstracts published in this book are the outcome of the Island Invasives 2017 Conference co-hosted by the University of Dundee and the South Georgia Heritage Trust, held at the University of Dundee, Scotland, from 10 to 14 July 2017.

The guidelines for this conference were: "any topic relating to invasive alien species on islands, where the term 'island' is broadly interpreted and (rather ironically from a classical perspective) may include a submarine island – e.g. a coral reef. The invasive species involved may be flora or fauna. Particularly encouraged were papers that relate to the theme of the conference – scaling up to meet the challenge – or to either biosecurity/quarantine or post eradication impacts on native biota."

The editors thank Carola Warner for her assistance with editing of many papers.

All papers have been peer reviewed and we thank all reviewers. The content of the papers is the choice of the authors. The style of presentation has been modified in consultation with the editors. Nomenclature follows international published standards.

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PREFACE

Addressing the challenge

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The papers in this volume were, with a few exceptions, presented at the third Island Invasives conference, held in Dundee, Scotland in July 2017. The conference was attended by 254 people from 41 countries or territories, reflecting growing global recognition of the problems caused by invasive alien species (IAS) on islands and recent progress in solving those problems.

The prefaces of the Proceedings of the two earlier conferences in this series (Veitch & Clout, 2002, Veitch, et al., 2011) discussed many of the threats posed by IAS on islands, conditions for eradication feasibility, possible complications and successes to date. They remain as relevant today as when they were first published. So, what has changed in this field of conservation in the seven years since the last conference proceedings? In a nutshell, scale, diversity and experience.

The first two conferences in the series were influential in bringing together people from many parts of the globe to discuss and exchange ideas, to learn from the experience of others, to inform and inspire. The principal motive for the South Georgia Heritage Trust and University of Dundee to host the third conference in Dundee was a desire to give something back to the island invasives community in recognition of the enormous support and assistance provided by so many people to the South Georgia Habitat Restoration Project. The 2010 Auckland conference, and the contacts made there, were undoubtedly pivotal in guiding the South Georgia operation to success (Veitch, et al., 2011).

The sub-title of the Dundee conference was 'Scaling up to meet the challenge'. The papers in this volume and in the recent literature demonstrate up-scaling in several aspects of eradication operations – not least in ambition, land area, operational size, global reach and of course financial cost. In the space of a few decades, the size of islands treated for invasive species has increased by five orders of magnitude – from a few hectares to over 100,000 ha or 1,000 km^2. Meanwhile, the diversity of species being tackled has increased, as has the range of countries now actively carrying out island restoration work. Inspired by pioneers from New Zealand and Australia, principally, today the movement has spread to islands in all oceans and off all continents. This expansion has been informed by, and has in turn produced, growing experience in all aspects of this field, from non-target impacts to ecological responses to factors affecting eradication success. We now know much more about why some eradication attempts fail, and consequently how to prevent subsequent failures. We know how much operations will cost, and what level of budget contingency to allow – hugely important considerations for potential organisers, fund-raisers and sponsors. Crucially, and due in large measure to the internationally recognised work of the Island Eradication Advisory Group (IEAG – staff of the New Zealand Government's Department of Conservation), operation planners now have access to Best Practice guidelines, and these have underpinned much of the work reported upon in this volume. This field of conservation is remarkable in the degree of mutual support and encouragement between individuals, organisations, countries, and between Government and non-Government institutions.

The Dundee conference was opened by Her Royal Highness The Princess Royal, Patron of the South Georgia Heritage Trust. Lord Gardiner, Parliamentary Under Secretary of State for Rural Affairs and Biosecurity, spoke to the conference about the British Government's support of the South Georgia Habitat Restoration Project and its commitment to confront problems caused by Invasive Alien Species in the United Kingdom and its overseas territories more widely. Both addresses are published in these Proceedings, with kind consent.

Indicative of the level of ambition now influencing the field, several papers in this volume address topics related to the unveiling of *Predator Free 2050*, a campaign to rid New Zealand of its most damaging invasive mammalian predators by the year 2050. If this bold objective is to be achieved, novel tools will be needed to complement the existing arsenal of traps, bait, shooting etc. Among the concepts being considered is that of gene drives – a means of reducing an invasive population to zero by genetic engineering. The potential power of such developments generates both excitement and concern, was the subject of much discussion at the conference and illustrates how the field is adapting to the ever-increasing challenges posed by invasive species worldwide.

Learning from mistakes is a vital driver of progress, yet authors and journal editors alike are often reluctant to publish papers that discuss failure. The editors of this volume have consequently encouraged practitioners to write about what went wrong or could have been improved in their own projects. Such openness is a sign of confidence and of a desire to advance the field, with each operation being informed by the experiences of those that have gone before.

The editors of this volume have summarised what they consider to be key conclusions and lessons to be drawn from the many, diverse papers published within it. They are:

- The size of islands successfully cleared of invasive alien species that have been the target of eradications has increased by an order of magnitude since the previous Island Invasives conference.
- Successful and large-scale eradications of invasive mammals other than rodents from islands continue to occur, although some (e.g. mustelids) present significant challenges.
- There are still relatively few examples of successful eradications of invasive birds, but some have been achieved, despite management challenges and the threat of new incursions.
- The herpetofauna papers really highlight the need for effective border biosecurity to exclude pests as well as information to guide the importation of exotic organisms.
- For invertebrate eradications, principles are the same as for mammal eradications however revisions to criteria to guide terrestrial arthropod eradications are proposed. Adaptive management during eradication attempts is a consistent theme and methodologies to evaluate the response of invertebrate communities to mammal eradications and non-target impacts of vertebrate toxins on endemic molluscs are proposed.
- Plant eradications require persistence over the long term because many species have a seed bank (or similar cryptic life-stage) of high and often unknown longevity: regular surveillance is essential to detect and remove plants as they germinate from the seed bank and before they reproduce. In many situations, eradication is the optimal solution rather than ongoing control.
- Successful eradications of invasive aquatic species continue to be reported and can be achieved using tools and knowledge currently available.
- The presence of human populations can raise the cost and complexity of invasive eradication operations, but investment in community engagement and participation may remove barriers and should be factored in to all future operations on inhabited islands.
- Reviews of single or across multiple operations show the breadth of scope of invasive rat eradications and are important for knowledge sharing and understanding failure.
- Lessons from invasive rat eradications, particularly from those facing complex or novel challenges, are important to inform attempts on other islands.
- Effective biosecurity is essential to prevent new invasions and re-invasions and requires community involvement, proactive planning, monitoring and rapid response.
- Cooperation between indigenous (local) and national governments may allow projects to expand beyond biodiversity conservation to become culturally significant as well, i.e. restoring or aligning with existing traditional knowledge or resource use practices or refining and improving these and bringing them into the realm of the total, diverse human population locally, regionally or nationally.
- Successful eradications of invasive species often yield significant benefits to native species, natural ecosystems and local communities.
- The effect of climate change on invasive species impacts is poorly known, so further research is needed as well as application of the precautionary principle.
- Genetic techniques such as gene drives offer the potential to facilitate eradications on very large scales, but must be treated with caution until more research is conducted on impacts and feasibility.
- As ambition grows, so does the need for new techniques to facilitate eradications on geographical scales never previously considered.

Our field is as much practical as it is academic, and a major aim of publishing these Proceedings is to inform people who are, or will in the future be, planning new projects to free islands of invasive species. Regardless of its location or the target species involved, each successive operation builds on the experience of those who have gone before, and the papers in this volume represent an invaluable wealth of such experience.

References

C.R. Veitch and M.N. Clout (eds.) (2002). *Turning the tide: the eradication of invasive species.* Occasional Paper SSC no. 28 IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

C.R. Veitch, M.N. Clout and D.R. Towns (eds.) (2011). *Island invasives: eradication and management*. Occasional Paper SSC no. 42 Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

OPENING ADDRESS

Her Royal Highness, the Princess Royal

Princess Anne

VICE-CHANCELLOR, LADIES AND GENTLEMEN,

it is a real pleasure to be able to join you for this, the third of the Island Invasive conferences. I can work out by the very fact of their subject matter being islands that these conferences are few and far between, but judging by the programme I have seen for the next five days you are really making up for the gap in between. There are a lot of good things to report and to talk about and to share.

I am delighted that one of those discussions will be about the South Georgia Heritage Trust and the rat eradication scheme that has been underway. It has been a real privilege to visit South Georgia, not once but twice, in order to see some of that work and understand the environment in which the Trust is functioning. I am hugely impressed by just how successful that work appears to have been, though we are not counting our chickens, or indeed any of the other birds that appear to be doing better as a result of the rat eradication so far, and we look forward to being able to prove the project's success in the future.

The progress that has been made in many of the eradication schemes must give enormous encouragement to others, to know now what is really possible and to believe that they can do the same. It perhaps underlines just how important it is to have your methodology, logistics and various other aspects in line before you can begin to have a successful project.

You have here an astonishing gathering of those who have been most involved at both the research and the practical delivery end of these eradication schemes, which covers a pretty wide range of talents. Looking at your conference book there is an enormous diversity of success stories to tell. You have seven of the world's foremost specialists in their respective fields who have agreed to give keynote presentations at this conference. They will talk on a wide range of topics, from island biosecurity to invasive plants and international policy.

Contemplating invasive plants, I have to say, makes eradicating rats look positively straightforward, but I'm sure we will come up with a process for tackling those too. Scotland of course has its own conservation challenges, as we will hear in the keynote speech, which will talk about the Shiant Islands, somewhere that I have also had the pleasure of visiting, not officially but as part of a sailing trip. If you time it right and the puffins are still there it is an extraordinary place, well worth a visit, but it faces similar challenges as South Georgia.

In other islands around Scotland there are other issues to face. An invasive species I can think of in Coll came all the way from New Zealand and is composed of some large and rather successful worms, which everybody hoped would eat themselves out of house and home but have failed to do so yet!

The challenges facing the organisations attempting to remove invasive species from islands are diverse. Progress can be made from the experience of earlier operations and oddly the operations that failed can teach us even more. The South Georgia Heritage Trust has hugely benefitted from the knowledge of the Island Invasives community in tackling South Georgia's eradication work. In deciding to help host this next Island Invasives conference in partnership with the University of Dundee, we really do believe that that will inspire the next generation of island conservationists. Dundee is a very good place to be able to do that, and to the Vice-Chancellor and everybody here from the University of Dundee, thank you for your hospitality.

On our last trip to the island we were in Possession Bay to celebrate Captain Cook's first arrival at South Georgia two hundred years before. When we looked at Possession Bay at that particular moment one could sympathise with Cook as he wondered what on earth he was doing there, but it would be very nice to be able to say that South Georgia had returned to the condition in which Captain Cook found it and claimed possession. We still believe that might be possible, so we look forward to the next couple of years and being able to revisit South Georgia to really prove that the eradication has been a success before we declare the island rodent free. We all understand that 99% success is not quite enough when it comes to removing rats and mice.

I am sure that this conference is something that you have all been looking forward to, but I hope that you enjoy it on a number of different levels, not least for the chance to make friends because however successful you are at communicating online, it is really nice and possibly more encouraging to meet the people who have been involved and can give you that very personal information about what really works, what didn't work and the little things that caused the big problems. And those valuable lessons from each other's experiences will be something that you can all take away, as well as happy memories of your time in Dundee.

KEYNOTE ADDRESS

Protecting the biodiversity of the UK Overseas Territories

Lord Gardiner of Kimble

LADIES AND GENTLEMEN,

It is a great privilege to be here at this Conference and I am delighted to learn more about how we can better protect the biodiversity of our island ecosystems from the threat of invasive alien species.

As you will hear, this week, much has been achieved by passionate and committed conservationists around the world since you last met in Auckland seven years ago. We have a great opportunity this week to celebrate successes and learn from these experiences.

The eradication work completed by the South Georgia Heritage Trust, which you will hear about later this evening from Professor Tony Martin is undoubtedly among the most remarkable of recent island conservation efforts.

As we approach 2020, it is also a good time to reflect on our progress towards the ambitious targets adopted by the global community on invasive alien species as part of the Aichi Targets and the Sustainable Development Goals.

UK successes

The past decade has seen a step-change in how the UK responds to invasive non-native species. We now have a co-ordinating secretariat, a risk analysis mechanism, a GB Strategy and are prioritising species and pathways for action.

My ministerial colleagues and I meet each month to consider emerging threats across the biosecurity spectrum – including animal diseases, plant pests and invasive non-native species. The UK Department for Environment, Food and Rural Affairs and the Scottish and Welsh Governments will soon be putting in place contingency plans to stop over 30 high-risk invasive species getting a foothold in the UK.

This approach was tested last autumn when one of our top threats – Asian hornet – was spotted in the south west of England. My Department had a team on the ground within 48 hours and had successfully eliminated this specific threat within 10 days.

The UK has also completed three further national rapid response eradications, targeting two fish and one amphibian species. Another six eradication campaigns are underway. The biggest of these, the eradication of the Ruddy duck, a world class effort covering the whole of the UK, is now almost complete after more than a decade of concerted effort.

Sadly, some non-native species are here to stay, yet we seek to mitigate their impact. To this end, we have invested over £1m in research on biocontrol agents for several invasive plants.

The Department has also invested in a public-private partnership to research novel methods of grey squirrel control. Scotland is fortunate to have the Red Squirrel still relatively widespread here.

We also recognise that effective awareness-raising is key – for example, the GB Non-Native Species Secretariat is leading an awareness-raising campaign called *Check Clean Dry*, aimed at encouraging anglers and boaters to reduce the risk of moving invasive species between waterways. We adapted this from an excellent New Zealand campaign of the same name.

I believe that it is vital to learn from the experiences and good practice of others. We within the UK are keen to share the lessons that we have learnt and the expertise that we have developed – and to put these at the disposal of the Overseas Territories and in collaboration more widely.

Focus on Overseas Territories

It is clear that much remains to be done to tackle the issue of invasive non-native species. This is why the International Union for Conservation of Nature launched the global Honolulu Challenge last year. In December, the UK Government pledged £2.75 million for priority activities to tackle invasive species in the UK Overseas Territories.

The UK is proud to be custodian of the precious and unique biodiversity of 14 Overseas Territories, which account for over 90% of UK biodiversity and contain two World Heritage sites.

Our pledge shows our commitment to working in partnership with the Territories to address what is probably the single greatest threat to these unique places.

Many of the Overseas Territories are small island environments that are highly vulnerable to environmental change. They contain rare species found nowhere else on the planet; species that have often evolved over thousands of years in isolation from predators, competitors and diseases, and are therefore highly susceptible to invasive threats.

Sadly, we have already seen the loss of some unique species, like the giant earwig of St Helena. Others, like the endangered Henderson petrel, and the Cayman blue iguana, remain under pressure from invasive species. Crucial work is underway to save the Monserrat mountain chicken, a unique frog which nearly disappeared from the island following the incursion of an aggressive fungal disease.

Territory Governments are increasingly alive to these issues and addressing them head on, putting in place biosecurity regimes to prevent new introductions and manage existing threats.

On the ground, the National Trusts of Monserrat and Saint Helena, for instance, have worked in partnership with the RSPB to protect critical habitats and manage the impact of invasives on two unique bird species: the Monserrat Oriole, the national bird of Monserrat, and the Saint Helena Plover, the island's last remaining unique bird. In no small part thanks to these efforts, as of last December, these unique birds are no longer listed as critically endangered.

I am delighted that the Governments of Ascension, the British Virgin Islands, the Falkland Islands, and Tristan da Cunha are here in Dundee this week to share their experiences so we can learn from them.

Honolulu Challenge projects

As part of the UK's contribution to the Honolulu Challenge, the UK Government has committed £1m to support Territories in improving their biosecurity. The GB Non-Native Species Secretariat has already identified the key gaps in practices and capacity. They will now take targeted action to address them, sharing UK expertise on pathway management, horizon scanning, pest identification, and the development of effective legislation.

The UK Government is also contributing £1.75m to support the work led by the Royal Society for the Protection of Birds to restore Gough Island. Seabird populations on the island – including the critically endangered Tristan albatross and Gough bunting – are threatened by invasive house mice that have evolved to become the largest in the world. Every year, an estimated 900,000 seabird chicks are killed. The aerial eradication operation planned for 2019 can turn things around for this precious World Heritage Site. The RSPB, working in partnership with the Tristan da Cunha Government, is making excellent progress in preparing for the operation and is working hard to attract further support for this vital project.

The teams delivering both of these projects are here in Dundee this week. I know that they have planned useful discussions and are eager to draw on your expertise to advance their work. I wish you all possible success.

Learning from experience: South Georgia

Over the past 20 to 30 years, the pace of island eradications has quickened and projects have become increasingly ambitious. But each project tends to build on what has gone before.

We in the UK have learnt a lot from the ground-breaking work that New Zealand, Australia and South Africa have carried out in this field. It is their systems that have been adapted for recent work in South Georgia and that will be applied on Gough Island in 2019.

We have also learnt from the failure of the eradication project on Henderson Island. I know that the island restoration community remains committed to solving such difficult issues.

Tonight, you will hear about what has undoubtedly been one of the most ambitious island eradications carried out to date.

Less than a decade ago, seabirds in South Georgia, including the unique South Georgia pipit, were in decline and increasingly confined to a small number of rodent-free areas.

Remarkably, by 2015, the South Georgia Heritage Trust had completed the final steps of what has been the largest island rodent baiting operation ever attempted.

Great credit is due to the Trust for completing this ambitious operation – and for raising the majority of the funds needed to support it.

When you see Tony's pictures, you will appreciate the harsh terrain and weather conditions faced by the Trust in delivering this project.

Initial reports suggest that the endangered South Georgia pipits are already returning to areas where populations had previously been decimated by rats. This is a great success for the Trust, South Georgia, and the protection of UK biodiversity.

The UK Department for Environment, Food and Rural Affairs is proud to have supported the Trust's work with £885k, including through our dedicated Overseas Territories Environment and Climate Change grant scheme, Darwin Plus.

The South Georgia Habitat Restoration Project sets an outstanding example for island eradications in the UK Overseas Territories and beyond. I hope future island restorations will be able to emulate the project's success.

The Trust's work also well and truly establishes the place of conservation charities – working in partnership with local governments – in the field of island eradications. I would like to pay tribute to the Trust and our other partners in the Overseas Territories, both Governments and charitable organisations, for their vital work.

I am delighted to now be able to hand over to Professor Tony Martin for a full account of the Trust's exceptional work in South Georgia.

I regret that I will not be able to stay for Tony's lecture and the fascinating discussions planned for the rest of this week, as I have to return to London to answer questions in Parliament tomorrow.

It has been a privilege to join your discussions today. I am sure that this will be an inspirational week, which will be the basis for vital progress in the years to come: 44 countries collaborating together is an inspirational force.

Invasive alien species management is an area where we can have a real and immediate positive impact and in many cases reverse the errors of our ancestors and leave the environment in a better state than we found it. It is our generation's responsibility to rise to this challenge and the expertise of all of you at this Conference give confidence and, importantly, hope.

PRESENTED PAPERS

Chapter 1: Rodents

With Sections: A. Planning
B. Review
C. Lessons

A potential new tool for the toolbox: assessing gene drives for eradicating invasive rodent populations

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Abstract Invasive rodents have significant negative impacts on island biodiversity. All but the smallest of rodent eradications currently rely on island-wide rodenticide applications. Although significant advances have been made in mitigating unintended impacts, rodent eradication on inhabited islands remains extremely challenging. Current tools restrict eradication efforts to fewer than 15% of islands with critically endangered or endangered species threatened by invasive rodents. The Genetic Biocontrol of Invasive Rodents partnership is an interdisciplinary collaboration to develop and evaluate gene drive technology for eradicating invasive rodent populations on islands. Technological approaches currently being investigated include the production of multiple strains of *Mus musculus* with a modified form of the native t-complex, or a CRISPR gene drive, carrying genes or mechanisms that determine sex. These systems have the potential to skew the sex ratio of offspring to approach 100% single-sex, which could result in population collapse. One goal proposed is to test the ability of constructs to spread and increase in frequency in *M. musculus* populations in biosecure, captive settings and undertake modelling to inform development and potential deployment of these systems. Structured ecologically-based risk assessments are proposed, along with social and cultural engagement to assess the acceptability of releasing a gene drive system. Work will be guided by an external ethics advisory board. Partners are from three countries with significant regulatory capacity (USA, Australia, New Zealand). Thus, we will seek data sharing agreements so that results from experiments may be used within all three countries and treat regulatory requirements as a minimum. Species-specific, scalable, and socially acceptable new eradication tools could produce substantial biodiversity benefits not possible with current technologies. Gene drive innovation may provide such a tool for invasive species management and be potentially transformative and worthy of exploring in an inclusive, responsible, and ethical manner.

Keywords: conservation, CRISPR, genetic biocontrol, invasive species, mice, *Mus musculus*, pest management, public engagement, risk assessment, transgenic

INTRODUCTION

Three *Rattus* species (*R. rattus, R. norvegicus, R. exulans*) and house mice (*Mus musculus*) are, outside of their native ranges, globally widespread invasive species (Capizzi, et al., 2014). These invasive rodents negatively impact stored foods, crops, and infrastructure and can carry pathogens that impact the health of people and their livestock (Stenseth, et al., 2003; Meerburg, et al., 2009; Banks & Hughes, 2012). Invasive rodents cause population declines and extinctions of island floras and faunas and interrupt ecosystem processes with negative cascading effects (Towns, et al., 2006; Jones, et al., 2008; Kurle, et al., 2008; Doherty, et al., 2016). To recover endangered populations and restore ecosystem processes, invasive rodents on islands are increasingly targeted for eradication, with at least 650 eradication attempts of introduced *Rattus* spp. populations to-date (Russell & Holmes, 2015). These and other island-based invasive mammal eradications have resulted in positive responses by native species with few exceptions (Jones, et al., 2016).

Anticoagulants are the most common control method for invasive rodents (Capizzi, et al., 2014). Rodent eradication on any island typically >5 ha has relied exclusively on the use of anticoagulant toxicants incorporated into cereal or wax baits (DIISE, 2016). Second generation anticoagulants are most commonly used and have had the highest success rate (Howald, et al., 2007; Parkes, et al., 2011). However, their broad-spectrum toxicity to vertebrates, duration of persistence, ability to biomagnify, mode of death and negative public perception limit their responsible use (Eason, et al., 2002; Fitzgerald, 2009; Broome, et al., 2015). These features can lead to negative impacts, including for conservation targets (e.g. Rueda, et al., 2016), although significant advances in strategies to mitigate these impacts have been made (e.g. Rueda, et al., 2019). Inhabited islands with children, livestock and pets present significant challenges because eradication is currently limited by a lack of species-specific methods, animal welfare issues, high fixed costs, and socio-political opposition (Campbell, et al., 2015). Hence, even with optimistic assessments for current methods (islands up to 30,000 ha and/or 1,000 people), eradications are possible on fewer than 15% of islands with critically endangered or endangered species threatened by invasive rodents (Campbell, et al., 2015). New species-specific, scalable tools are needed if we are to prevent extinctions.

Genetic biocontrol in the form of gene drives coupled with sex-determining genes to produce single-sex offspring, offers a potentially transformative new tool to add to the rodent eradication toolbox, by offering species-specificity not readily achievable in existing technology (Campbell, et

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al., 2015). Gene drives cause a gene to spread throughout a population at a rate higher than would normally occur (Champer, et al., 2016). Gene drives occur naturally and are not recent phenomena (Lindholm, et al., 2016); for example, mice with the native *t*-complex gene drive were first described in 1927 (Schimenti, 2014). Attempts to harness naturally-occurring gene drive systems, primarily for invertebrate pests and disease vectors have had mixed results (Sinkins & Gould, 2006; Champer, et al., 2016). In 2012, the Genetic Biocontrol of Invasive Rodents (GBIRd) partnership was formed between North Carolina State University (NCSU), Island Conservation (IC) and later Texas A&M University (TAMU). GBIRd started exploring opportunities for harnessing the native *t*-complex gene drive in mice to eradicate invasive mouse populations on islands (Kanavy & Serr, 2017; Piaggio, et al., 2017). Other partners were identified through professional networks and during searches for specific skillsets. GBIRd currently includes seven partners in three countries: TAMU, NCSU, University of Adelaide (UA), USA Department of Agriculture's National Wildlife Research Center (NWRC), the Agriculture and Food Business Unit of the Commonwealth Scientific and Industrial Research Organisation (CSIRO), Landcare Research (LR), and IC.

Beginning in 2013, a harnessed bacterial immune response system called CRISPR/Cas9 revolutionised the field of genetic engineering. CRISPR/Cas9 can be used to delete, modify or insert new genes more precisely, effectively, time- and cost-efficiently than previous gene editing tools (NASEM, 2016). Multiple genes can also now be edited simultaneously. In 2014, a landmark paper (building upon earlier concepts of Burt, 2003), described how a cassette encoding the CRISPR/Cas9 machinery could be precisely inserted into an organism's DNA, creating a self-replicating gene drive with potential to modify wild populations by design (Esvelt, et al., 2014). Since then, CRISPR/Cas9 gene drives have been developed in yeast *Saccharomyces cerevisiae* (DiCarlo, et al., 2015), fruit fly *Drosophila melanogaster* (Gantz & Bier, 2015) and both *Anopheles stephensi* (Gantz, et al., 2015) and *A. gambiae* (Hammond, et al., 2016) mosquitoes as proof-of-concept demonstrations in biosecure laboratories. This field has become a significant focus of research, and USA and Australian Academies of Science have provided recommendations aimed at guiding its development (NASEM, 2016; AAS, 2017). GBIRd, with its partnership already established, adopted CRISPR as a gene editing and potential gene drive tool.

Gene drives are a technology platform. GBIRd partnership considers *Mus musculus* the logical starting point for developing, exploring, and providing proof-of-concept for a genetics-based invasive vertebrate eradication tool. They are the model vertebrate species for genetics, possess a short generation-time, are small, husbandry is straight-forward, and they are invasive around the world including on many islands (Guénet & Bonhomme, 2003; Phifer-Rixey & Nachman, 2015). Mice are also among the best studied species in terms of mammalian sex determination, reproductive biology, behaviour, genetic manipulation and genetic control of phenotypic traits (Guénet & Bonhomme, 2003; Eggers, et al., 2014; Phifer-Rixey & Nachman, 2015; Singh, et al., 2015). If proof-of-concept, safety, and efficacy are demonstrated in *Mus musculus*, it should be possible to apply this approach to *Rattus* species.

The GBIRd programme (<http://www.geneticbiocontrol.org/>) aims to develop multiple gene drive systems in mice for simultaneous evaluation of safety and efficacy, while carefully assessing the social, cultural and policy acceptability of such an approach. Our staged inclusive approach reflects USA and Australian Academies of Sciences' recommendations (NASEM, 2016; AAS, 2017) that we treat as our minimum standards. The GBIRd partnership aims to provide vital data for conducting risk assessments, determining efficacy, and engaging stakeholders and communities in order to inform and enhance progress, or identify limitations, of future research. A potential longer-term goal is submission of an application to a regulatory agency for release of gene drive constructed mice on a small, biosecure island to test eradication of the wild, invasive mouse population.

This paper provides an overview of the GBIRd programme as it has developed to-date, including the risks and opportunities as they are currently envisioned and understood. These will certainly evolve, and the programme must strategically evolve with them.

Genetic Biocontrol of Invasive Rodents programme

The programme's guiding principles provide context for decision making:

- Proceed cautiously, with deliberate step-wise methods and measurable outcomes;
- Engage early and often with the research community, regulators, communities and other stakeholders;
- Maintain an uncompromising commitment to biosafety, existing regulations, and protocols as minimum standards (e.g. NASEM, 2016; AAS, 2017);
- Use, and participate in developing best practices;
- Only operate in countries with appropriate regulatory capacity; and
- Be transparent with research, assessments, findings, and conclusions.

1. Governance and Coordination

GBIRd involves seven organisations from Australia, New Zealand and the USA; three universities (NCSU, TAMU, UA), three governmental research (CSIRO, LR, NWRC) and one non-governmental non-profit (IC). Each has specific roles and responsibilities (Fig. 1) as detailed in the memorandum of understanding that formalises the partnership. A steering committee comprised of one or two representatives from each organisation provides direction and decision making, and a programme coordinator facilitates activity. The consortium is inclusive and, indeed, strengthened by a transparent internal dialogue in both the scientific positioning (e.g. Gemmell & Tompkins, 2017) and societal/values realm (e.g. Webber, et al., 2015). GBIRd has 14 component areas and three cross-cutting themes (Fig. 1) being investigated, as follows.

2. Gene drives

Three gene drives are currently being investigated; a modified *t*-complex, a CRISPR/Cas9 and a CRISPR/Cpf1 gene drive. The *t*-complex on chromosome 17 in mice is a natural male-transmitted meiotic drive (Lyon, 2003; Schimenti, 2014). The *t*-complex impairs sperm not carrying the *t*-complex, leading to an increased frequency of *t*-complex carrying sperm fertilising ova. The frequency of the *t*-complex in natural populations of house mice is typically lower than predicted given the often very strong transmission ratio distortion displayed. This phenomenon is not completely understood (see Lindholm, et al., 2016), but may imply that a sex-biasing system based on the *t*-complex would require ongoing releases to be effective (Backus & Gross, 2016). The *t*-complex haplotype we are using is free of recessive lethals and has a high rate (>95%) of inheritance, also called transmission distortion (Kanavy

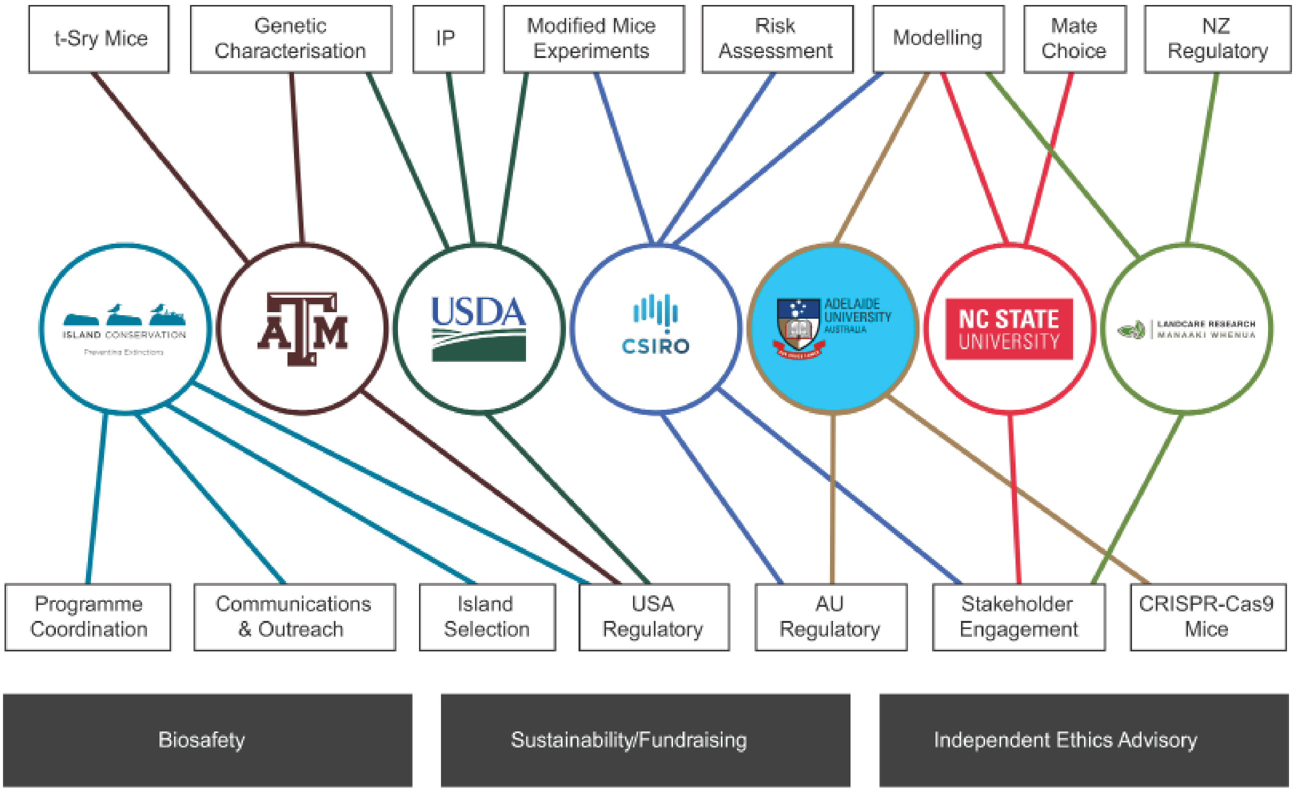

Fig. 1 Programme map, showing 14 component areas being investigated by partners of the Genetic Biocontrol of Invasive Rodents programme. The three components not linked to any organisation are cross-cutting themes.

& Serr, 2017; Piaggio, et al., 2017). The remaining offspring (<5%) would not carry the gene drive or exhibit the phenotypic traits of the genes being driven (Piaggio, et al., 2017).

CRISPR/Cas9 gene drives are capable of >94% inheritance (Gantz, et al., 2015; Hammond, et al., 2016). Once inserted within one individual's genome, a gene drive can work in one of two ways. A zygotic gene drive works when that individual's ova or sperm are fertilised. If the gene drive cassette is activated in the fertilised egg (zygote), the guide RNA (gRNA) directs Cas9 to produce a double-stranded break in the DNA at the target site in the chromosome lacking the gene drive. This triggers the cell's repair mechanism to repair the break using the gene drive-containing chromosome as a template resulting in self-replication of the gene drive. Alternatively, in a germline gene drive, germ cells can be targeted as the stage for self-replication of the gene drive.

3. Targeted genes

Genes can be targeted for deletion, modification or insertion of new genes in conjunction with a gene drive to increase inheritance of specific traits. Investigations currently focus on the appropriateness of two target genes (*Sry*, *Sox9*) to be inserted and one chromosome to be deleted (Y-'shredder'), each in coordination with a gene drive. The *Sry* gene is found on the Y chromosome and is considered the master sex-determining gene in most mammals (Kashimada & Koopman, 2010; Eggers, et al., 2014). Another key component of the testis pathway is the autosomal gene *Sox9*, which acts immediately downstream of *Sry* (Eggers, et al., 2014). Both genes drive the development of male testes in mammals and sex reversal has been demonstrated in transgenic female (XX) mice (Koopman, et al., 1991; Vidal, et al., 2001; Eggers, et al., 2014). A Y-shredder (Adikusuma, et al., 2017) promotes solely offspring with one (XO) or two X (XX) chromosomes, i.e. females. Initial developments focus on *t*-complex with *Sry* inserted (*t-Sry*), and CRISPR/Cas9 and CRISPR/Cpf1 gene drives with *Sox9* and Y-shredder.

As of June 2018, partners attempting to incorporate *Sry* into a *t*-complex drive have been challenged by the large construct size of *Sry*. If that technological hurdle can be overcome, these mice are expected to produce >95% phenotypically male offspring (Kanavy & Serr, 2017; Piaggio, et al., 2017). The mice currently under development in Australia are expected to test the functionality of a split CRISPR/Cas9 gene drive that uses phenotypic coat markers as genetic 'cargo'. A 'split gene drive system' has the gene drive in two separate 'cassettes' (DiCarlo, et al., 2015). This design is a safety feature for laboratory testing where the separation of the cassettes results in drive components being inherited separately even if a drive carrier were to escape, thus preventing drive function (since both are necessary for function). Development of CRISPR/Cpf1 gene drives and incorporating *Sox9* and the Y-shredder are underway.

4. Spatial control of gene drive

Spatially or temporally limiting drive function is one of the major research challenges for CRISPR gene drives, e.g. restricting a gene drive to affect only a single island's rodent population. Our programme is investigating genome-level targeting of population-specific locally-fixed alleles as a potential spatial control mechanism. It is likely that through the process of invasion, founder effects and population bottlenecks, certain alleles across the genome have become fixed in any island population (Britton-Davidian, et al., 2000; Hartl & Clark, 2006). This pattern of fixation is likely a unique genomic signature in every genetically isolated island population. Similar to the molecular confinement strategy being implemented in the laboratory (see *Biosafety*), population-specific locally-fixed alleles (and their sequence) could act as unique gRNA targets for a CRISPR gene drive that will not function outside the island population. Others are investigating alternative approaches to temporally and/or spatially contain gene drives and their relative effectiveness (e.g. Dhole, et al., 2018).

5. Biosafety

Multiple biocontainment strategies accompany all laboratory work and are part of our staged testing pathway (following the recommended approach by NASEM, 2016). Recommended containment standards for gene drives include at least two stringent confinement strategies wherever possible, in addition to containment (Akbari, et al., 2015; NASEM, 2016), and our programme exceeds these standards. For example, the CRISPR gene drive studies are using physical containment at the currently required level (PC2) (AAS, 2017) and three containment/confinement methods; a 'split gene drive system' as explained above (DiCarlo, et al., 2015); coat colour (white or black) to identify the zygotic homing in offspring – white mice (Cas9-positive) are less likely to survive in the wild (Vignieri, et al., 2010); and gRNA exclusively targeting a synthetic sequence not present in wild mice, providing molecular confinement to transgenic laboratory mouse populations. For scaled laboratory trials, CSIRO and NWRC state-of-the-art facilities provide the opportunity to safely conduct trials with colonies of mice that could originate from islands.

6. Safety and efficacy experiments

Experiments demonstrating that constructs work effectively and efficiently, are species-specific and safe to the environment are needed. Data needs for risk assessments and field trial applications have yet to be determined in conjunction with regulatory agencies, and this will dictate minimum requirements for experiments. Experiments will inform risk assessments to reduce uncertainty surrounding outcomes and probabilities. Phased testing and experiments are viewed as part of the development process, and occur at each tier (i.e. molecular level, individual mice, mouse population, ecological community). This phased development process incorporates feedback loops to developers, and evaluates efficiency, stability, specificity and safety to determine whether a specific construct proceeds to the next stage (e.g. molecular to insertion in a mouse or going from individual mice to a colony). Constructs that pass will go on to more rigorous testing, and those that don't will either be dropped or modified and then re-evaluated. No functional CRISPR drives have yet been reported for vertebrates. Attempting development of multiple combinations of gene drives and gene targets within our programme increases the likelihood of success, and, if successful, would provide opportunities for comparative analyses and risk assessments. High-quality data for modelling and risk analyses will be necessary.

7. Mate choice

Behavioural barriers to mating success and resulting gene flow must be considered, as to how (or if) a gene drive will successfully spread through a population, and if understood and used correctly may provide significant advantage. Key characteristics influencing male reproductive success in mice include aggressive dominance for securing territories, and a preference among females for unfamiliar males (Gray & Hurst, 1998; Cunningham, et al., 2013). Promiscuity of male mice and their ability to inseminate many females provides males the potential to disproportionately influence the genetic makeup of future generations. Experiments in the 1980s introducing Isle of Eday mice to the Isle of May (57 ha) demonstrate the power of selecting appropriate stock for facilitating introduced individuals 'invading' another population (Berry, et al., 1991; Jones, et al., 1995). A Y-chromosome (i.e. male) linked marker spread across the Isle of May site within six months and in 18 months only hybrids could be detected (Berry, et al., 1991; Jones, et al., 1995). The 42 Isle of Eday males introduced were estimated at <5% of May's resident mouse population, demonstrating differential success of introduced versus resident males (Berry, et al., 1991; Jones, et al., 1995). We aim to rank the 'invasability' of males from laboratory strains, selected islands and mainlands so that appropriate stock may be selected for backcrossing in gene drives and their cargo. Initial trials involve *t*-complex carrying laboratory mice (C57BL/6/129 strain), Southeast Farallon Island, and F1 hybrid Farallon-laboratory mice in small cages with single males and females, to determine if mating would occur (Serr & Godwin, 2019). (Note: Southeast Farallon Island is not considered a potential site for field trials at this time). Larger arenas were used to determine mate choice and male competition where males from different populations would have to compete for females and resources (Serr & Godwin, 2019).

Behavioural experiments to-date indicate that *t*-complex carrying lab mice can successfully mate with island mice in captivity (Serr & Godwin, 2019). Other mate competition results indicate that male F1 hybrid Farallon-laboratory mice may be able to outcompete male Farallon island mice.

8. Island selection

As part of our staged, stepwise approach, if biosecure laboratory studies support safety and efficacy in biasing sex ratios and supressing test populations, the next stage will involve studies in natural settings under conditions where dispersal or persistence of the organisms outside the evaluation area is restricted (NASEM, 2016). We have identified a suite of ecological criteria for initial selection of potentially appropriate islands for trials, including 1. the island is biosecure (i.e. closed to public or infrequent/controlled visitation; and remote enough (>1 km from other land masses) to avoid unassisted immigration or emigration), 2. no significant challenges exist to treatment using traditional toxicant-based methods to eradicate mice (e.g. no major non-target species, regulatory environment allows the use of brodifacoum bait products, single land manager), 3. *M. musculus* are the only rodent present *or* could be introduced, and 4. the island is reasonably economical and feasible to visit year-round (see Harvey-Samuel et al., 2019 for a more detailed account and rationale). By selecting islands where the use of traditional eradication methods could readily be used to eradicate all rodents (Howald, et al., 2007) a contingency (i.e. exit strategy) explicitly exists. However, these ecological criteria are just a first filter and additional steps would be required prior to any field trial, including engagement with stakeholders (e.g. land managers, local communities) and regulators to determine final approval (Harvey-Samuel et al., 2019).

9. Population genetic characterisation

Genetic characterisation of mouse populations from islands selected for potential trials will occur using next-generation sequencing technologies (e.g. Illumina Mi-Seq). Analyses of these data will inform the feasibility of using population-specific fixed allele sequences as gRNA targets to provide spatial control of any gene drive trialled. They will also provide baseline assessments of genetic characteristics of target island populations, and potentially inform future strategies.

10. Modelling

Modelling can be used to inform broad strategies, such as male or female biasing gene drives and, within those strategies, to identify heritable traits or environmental conditions that provide disproportionate advantages (Bax & Thresher, 2009; Backus & Gross, 2016). Modelling is contemplated at each development stage (i.e. molecular, individual mouse, mouse population, ecological community), incorporating data from experiments and trials, and providing feedback to developers and trial designs. It aims to predict outcomes, reduce the number of animals required in experiments and trials and provide insight on strategies. At the molecular level, for example, the efficiency and stability of homing and non-homologous end joining for Cas9 and Cpf1 zygotic and germline homing approaches can be modelled based on data from experiments informing on likelihood of failure (Prowse, et al., 2017). Models also consider individual mouse characteristics and the effects these may have at the population level. A population model would estimate the number of gene drive mice with certain characteristics required for release to a specific island, the optimal frequency, timing and location of releases, and time until eradication. The impacts of changes to specific mouse characteristics (or other variables) can then be estimated. As data sets accumulate, the accuracy and sophistication of models will increase. The opportunity exists to leverage a 30+ year dataset and existing mouse population models, which will facilitate sophisticated analyses and allow the development of advanced deployment strategies that optimise seasonal and climatic variation (Singleton, et al., 2005; CSIRO, unpub. data). The use of these and other models will be critical in the development of robust ecologically-based risk assessments.

11. Risk assessment

There is the possibility that releases of gene drive-modified organisms will lead to unpredicted and undesirable side effects. Ecologically-based risk assessments (EBRA) aim to reduce some types of uncertainty surrounding outcomes and probabilities (NASEM, 2016; AAS, 2017). They are used to estimate the probability of immediate and long-term environmental and public health harms. EBRAs allow alternative strategies to be compared (e.g. traditional use of toxicants), incorporate the concerns of relevant publics, and can be used to identify sources of uncertainty, making them well-suited to inform research directions and support public policy decisions about emerging gene drive technologies. EBRAs provide the ability to trace cause-and-effect pathways and the ability to quantify the probability of specific outcomes. We regularly consult with risk assessment experts leading other gene drive EBRAs and plan to apply specific tools to identify where, within our development process, additional studies are required to reduce uncertainties, complementing regulatory requirements. The large existing body of work on rodent eradications, including the potential ecological impacts from toxicant use (Broome, et al., 2015) and probability of success of traditional methods (DIISE, 2016), along with meta-data analyses on the ecological impacts of removing invasive rodents (Jones, et al., 2016) will facilitate rigorous EBRAs. Our staged experimental approach prior to any potential release would culminate in trials within biosecure simulated natural environments with colonies of mice imported from the target island(s) with the most efficacious gene drive mice. This allows simulations of various ecological scenarios and increases the power of predictive analyses, resulting in increased levels of certainty around potential outcomes and ecological impacts.

12. Social engagement

The emergence of gene drives and other genetic technologies will force not only technologists, but conservationists, other environmentalists and the public to "negotiate with unfamiliar interest groups and perhaps compromise on deeply held positions if they are going to succeed in a complex world of contradictory perspectives" (McShane, et al., 2011, p. 969). We hope to develop guiding principles to establish dialogue between these disparate groups to identify and eventually negotiate trade-offs, things that should not be traded off, and also to "render explicit the relevant justice dimensions and principles at play in particular contexts" (Martin, et al., 2015, p. 176). The programme aims to establish a transparent process that both encourages public participation and offers a trustworthy and responsible decision pathway for making decisions about releases of gene drive organisms.

Specifically, members of our team have developed a three-part plan for social engagement. First, we will conduct a stakeholder landscape analysis to understand the mix of interests, priorities, concerns, and hopes of diverse stakeholders that surround the programme. Second, we will convene a stakeholder workshop to create a forum for discussion, provide feedback to the technical project team, and strategise the design of community engagements. Third, we propose to organise community focus groups near potential island release sites to engage relevant publics sufficiently early to influence technological innovation and field trial research (see Chapter 7, NASEM, 2016). Importantly, the international nature of our partnership will foster the sharing of best practices – and challenges – of social engagement across different cultural contexts.

To-date, engagements have occurred with publics, scientists, conservationists, indigenous groups and other stakeholders (including those opposing gene drive research, Borel, 2017; Reese, 2017), but more work is required.

13. Communications and outreach

The investigation requires clear, concise, and transparent communications to ensure public perceptions by target audiences are based on facts, and not unduly influenced by scientifically-unsubstantiated fears and hyperbole. Communicating to stakeholders, researchers, communities, and decision-makers interested in this evaluation is the foundation of the programmatic principle of transparency. Coordinated external communications by the partnership's representatives through media, in peer-reviewed publications, presentations, and one-on-one outreach have and will continue to be core to our mission. Informing stakeholders and decision-makers in fora such as the IUCN's World Conservation Congress and the United Nations' Convention on Biological Diversity encourages public discourse about this innovation, engages thought leaders in making our investigations more robust, ensures that fact-based concerns can be addressed while unsubstantiated fears can be allayed, and helps guide decision-makers in developing policies and guidelines complementary to the precautionary, stepwise research guiding principle, even as the technology is being developed.

14. Ethics

There are considerable potential benefits of this technology and we are committed to exploring it in a responsible and inclusive manner. But the question remains, if the technology works, should it be used? This key ethical question is best answered once robust EBRAs have been completed and in the context of rigorous social and regulatory engagement. The USA and Australian

Academies of Science recommend that research continue and decisions to release gene drives continue to be made on a case-by-case basis following a comprehensive environmental risk assessment that includes ecological and evolutionary modelling (NASEM, 2016; AAS, 2017). We have volunteered our programme as a case study for discussion at various fora, including ethical deliberations amongst ethicists and peers (e.g. NCSU Genetic Engineering and Society Center, 2016; Leitschuh, et al., 2018), on national radio (Barclay, 2017) and for the USA National Academies of Sciences Engineering, and Medicine's report on gene drives (case study 4, NASEM, 2016). Emulating the Target Malaria partnership (<http://targetmalaria.org/>), an independent ethics advisory board has been established to provide advice on ethical matters and identify issues for the partnership's consideration.

15. Regulatory

Our regulatory engagement strategy is to ensure transparent and early engagement with the regulatory agencies responsible for the oversight and review of the program. Varying regulatory maturity exists around the world, with Australia and New Zealand having possibly the most developed and mature biotechnology regulatory review processes. The USA is revising regulatory guidelines through the Coordinated Framework for the Regulation of Biotechnology (Barbero, et al., 2017). Currently, in the USA it is likely the Food and Drug Administration will lead regulatory review of GBIRd.

Regulatory data-sharing agreements for registration of pesticides exist between Australia, New Zealand, and USA, and we anticipate that this will carry over to review of biotechnology. The design, execution, and data collection will be compliant with all three countries' regulatory agency requirements or under data sharing agreements.

The regulatory oversight and testing is intended to demonstrate efficacy and safety of the construct, i.e. does it work and what are the ecological consequences. Managing risks associated with its potential release, including capacity to "shut off" in vivo in case of unanticipated consequences is one hallmark of our programme. Testing will take place in a step-wise manner, laboratory development and characterisation, laboratory testing, pen trials and field trials. With the lack of clarity of regulatory pathways at this time, we are engaging regulators early, and have done so in Australia, New Zealand and USA to inform and ideally strengthen regulatory standards, while ensuring open dialogue and regulatory awareness of GBIRd exists.

16. Intellectual property

A patent for RNA-guided gene drives was filed in 2014 and two competing patents exist over CRISPR gene editing technology (Egelie, et al., 2016; AAS, 2017). However, there may be little scope for commercialisation for CRISPR/Cas9 gene drives for conservation and public health purposes (AAS, 2017). The intent of our partnership is to safely and effectively develop and assess this technology in a socially responsible manner that democratises the science involved with the innovation. Our partnership is composed of organisations that are dedicated to the public good potential of this technology. We intend for intellectual property to be secured in a manner that prevents unintended use but allows maximum benefit for communities and environments in need. The mechanisms with which to do this have not yet been identified.

17. Financial

Budget estimates until completion of experimental biocontained trials are uncertain until refinement of constructs to ensure appropriate characteristics is clear. Technical issues may arise, and data needs for risk assessments and field trial applications have yet to be determined in conjunction with regulatory agencies. The timeline for completion of experimental biocontained trials is also uncertain as not all funding has been secured, processes are of uncertain duration in some cases and requirements for experiments have not yet been determined in conjunction with regulators. Considering these caveats, we estimate US$16–22M will be needed over the next 4–5 years to complete experimental biocontained trials.

All programme areas are unfunded or partially funded at this time. We are actively pursuing opportunities for complementary funding.

DISCUSSION

Unlike incremental advancements in current technology or tools, the development of transformative applications cannot be undertaken within existing rodent eradication projects on islands or as part of rodent control on mainlands. Transformative innovations require deliberate intent and focussed programmes. GBIRd includes interdisciplinary scientists, varied experience, backgrounds and viewpoints. An analysis of the hazards associated with a hypothetical split gene drive is underway. If proof of concept of the gene drive can be established in laboratory populations, and suitable target populations can be identified, funding will be sought to perform a risk assessment building on the results of the hazard analysis. GBIRd is also engaging with independent external ethicists to develop best practice ethical conduct for gene drives. Indeed, as a programme we have attempted to maintain a balanced approach and wish to inform future decisions with the best science at that time. This does not preclude pursuing a pathway to broader deployment of this type of technology if, indeed, it proves to be safe, efficacious, and socially accepted.

In addition to impacting biodiversity on islands, invasive rodents also negatively impact the health of people and their livestock, and greatly reduce agricultural productivity, stored food stocks and damage infrastructure. In the future, these problems may also benefit from the application of gene drive systems in invasive rodents. However, the GBIRd programme is currently focussed on the development and evaluation of gene drives in invasive rodents on islands to prevent biodiversity loss. We are committed to a deliberate and step-wise approach following National Academies' recommendations (NASEM, 2016; AAS, 2017).

Eradication is a biological extreme involving all individuals in a population (Parkes & Panetta, 2009). Populations hold a diversity of genes that provide plasticity in behaviours and susceptibilities (e.g. Buckle & Prescott, 2012; Cunningham, et al., 2013). Eradication of a population requires that eradication method(s) overcome this variability (Parkes & Panetta, 2009). That we are looking to develop an eradication (i.e. complete and permanent removal of a population), and not a control (i.e. frequent removal of a portion of a population for perpetuity) tool, is intentional and strategic. Eradication provides permanent solutions and for invasive species is nearly always desirable when it can be achieved (Parkes & Panetta, 2009). Eradication methods may be used for control, but not necessarily vice-versa. Our methods must be robust enough to eradicate populations independent of their variability but specific enough, or controlled in some way, that the global population (especially native populations) are not at risk. The concept of eradication units is a useful way to think of this (Robertson & Gemmell, 2004). Are there alleles shared by all individuals (i.e. fixed) within invasive populations that are not found in the native population, or only a subset of individuals have? Gene drive could be contained under either of these scenarios. GBIRd is attempting to identify island-specific locally-fixed alleles that would provide molecular confinement of the gene drive to the target island population. If this is possible, potential exists for the approach to be scaled (e.g. where locally-fixed alleles can be identified for archipelagos, or for invasive but not native populations). Further, our programme is also researching differential mating success of males between populations to be able to select the most effective stock for transmitting a gene drive and associated genes to a target population.

CRISPR has transformed gene editing and CRISPR gene drives are providing similar transformational opportunities for genetic pest management (Webber, et al., 2015; Harvey-Samuel, et al., 2017). Our partnership was formed prior to these revolutionary tools, providing a ready foundation upon which we expanded our partnership and incorporated these tools, increasing the number of technical approaches and likelihood of success. CRISPR, as an editing tool, has also increased the efficacy of inserting large genetic sequences (e.g. 10kb *Sry*) and due to its precision, efficacy and high success rate has often reduced the number of animals required compared to previous approaches. We anticipate there will be other opportunities, technological or otherwise, that emerge throughout the life of our programme.

CRISPR has been shown to be able to edit DNA in a range of taxa (NASEM, 2016; AAS, 2017) and a CRISPR gene drive has advantages when developing a technology platform, when compared to the *t*-complex drive which may not be effective in species other than mice. However, the *t*-complex provides options and, being naturally occurring in mice, may increase social acceptability, or be technically more appropriate for certain situations. Having multiple gene drives and target genes or mechanisms allows for many potential combinations and simultaneous comparisons in efficacy, safety and acceptability. We are currently investigating various combinations of gene drive mechanisms (i.e. *t*-complex, CRISPR/Cas9, CRISPR/Cpf1) and target genes or deletion mechanisms (i.e. *Sry*, *Sox9*, Y-shredder), providing multiple potential combinations.

Spatial control and remediation of CRISPR/Cas9 gene editing and gene drives has been a major concern and is the focus of significant research. We are keeping abreast of advances in this field and will look to incorporate mechanisms developed where appropriate. Recent research identified CRISPR/Cas9 inhibitors that can block genome editing, providing a means to spatially, temporally, and conditionally control Cas9 activity (Pawluk, et al., 2016; Rauch, et al., 2017). As a nascent field, it is understandable that not all technological concerns have yet been addressed (NASEM, 2016; AAS, 2017), but a significant amount of research is underway to do so.

Few, if any, people are opposed to preventing extinctions but there is mixed opinion about the methods by which this is done. Rodent eradication on islands of any significant size can currently only be implemented with toxicants, the least publicly accepted of all control methods (Fitzgerald, 2009). Gene drives hold promise as putting an additional tool in the practitioner's toolbox that could increase the feasibility and scale of conservation efforts. In contrast to toxicant-based invasive rodent eradication campaigns characterised by a short duration of implementation and high fixed costs (Howald, et al., 2007; Holmes, et al., 2015), gene drive approaches could provide an alternative and flexible financial model. Alternative financial mechanisms such as endowments covering annual costs instead of single campaigns costing tens of millions of dollars may be feasible. If the anticipated species specificity holds true, risks from methods to non-target species (e.g. raptors, Rueda, et al., 2016) would be eliminated and the ability for non-specialists to implement projects would increase. Animal welfare concerns over the mode of death of rodents and non-target species from toxicants could be alleviated by gene drives that bias the sex of invasive populations as no animals would be killed (Dubois, et al., 2017). This approach could also facilitate potential future developments with other invasive mammals beyond rodents, including foxes (*Vulpes vulpes*) and rabbits (*Oryctolagus cuniculus*) in Australia (Kinnear, et al., 2016; AAS, 2017), brushtail possums (*Trichosurus vulpecula*), and stoats (*Mustela erminea*; Owens, 2017) in New Zealand. New Zealand has set a goal of eradicating invasive mammal predators from their country ('Predator Free New Zealand 2050' – New Zealand, 2016). One interim 2025 goal in this strategy is to develop a scientific breakthrough capable of removing at least one small mammalian predator from New Zealand entirely (New Zealand, 2016), and gene drive is one of a suite of potential innovations currently being considered. Globally, invasive rodents are linked to 30% of all extinctions (Doherty, et al., 2016), and currently threaten 88% of all insular critically endangered or endangered terrestrial vertebrates (TIB Partners, 2014). New, scalable, species-specific tools are needed to prevent further extinctions. The opportunity that gene drives as a transformative technology may bring to invasive species management is significant and worthy of exploring in a responsible and inclusive manner.

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REFERENCES

AAS. (2017). *Synthetic Gene Drives in Australia: Implications of Emerging Technologies*. Australian Academy of Science. <www.science.org.au/gene-drives>. Accessed: 17 November 2017.

Adikusuma, F., Williams, N., Grutzner, F., Hughes, J. and Thomas, P. (2017). 'Targeted deletion of an entire chromosome using CRISPR/Cas9'. *Molecular Therapy* 25: 1736–1738.

Akbari, O.S., Bellen, H.J., Bier, E., Bullock, S.L., Burt, A., Church, G.M., Cook, K.R., Duchek, P., Edwards, O.R., Esvelt, K.M., Gantz, V.M., Golic, K.G., Gratz, S.J., Harrison, M.M., Hayes, K.R., James, A.A., Kaufman, T.C., Knoblich, J., Malik, H.S., Matthews, K.A., O'Connor-Giles, K.M., Parks, A.L., Perrimon, N., Port, F., Russell, S., Ueda, R. and Wildonger, J. (2015). 'Safeguarding gene drive experiments in the laboratory'. *Science* 349: 927–929.

Backus, G.A. and Gross, K. (2016). 'Genetic engineering to eradicate invasive mice on islands: Modeling the efficiency and ecological impacts'. *Ecosphere* 7: e01589.

Banks, P.B. and Hughes, N.K. (2012). 'A review of the evidence for potential impacts of black rats (*Rattus rattus*) on wildlife and humans in Australia'. *Wildlife Research* 39: 78–88.

Barbero, R., Kim, J., Boling, T. and Doherty, J. (2017). *Increasing the Transparency, Coordination, and Predictability of the Biotechnology Regulatory System*. The White House. < https://obamawhitehouse.archives.gov/blog/2017/01/04/increasing-transparency-coordination-and-predictability-biotechnology-regulatory>. Accessed 9 July 2017.

Barclay, P. (2017). 'New Weapons in the Battle against Invasive Pests'. In: P. Barclay (ed.) *Big Ideas*. Australian Broadcasting Corporation. <http://www.abc.net.au/radionational/programs/bigideas/new-weapons-in-the-battle-against-invasive-pests/8580602>.

Bax, N.J. and Thresher, R.E. (2009). 'Ecological, behavioral, and genetic factors influencing the recombinant control of invasive pests'. *Ecological Applications* 19: 873–888.

Berry, R., Triggs, G., King, P., Nash, H. and Noble, L. (1991). 'Hybridization and gene flow in house mice introduced into an existing population on an island'. *Journal of Zoology* 225: 615–632.

Borel, B. (2017). 'How Genetically Modified Mice Could One Day Save Island Birds'. *Audubon magazine* Summer 2017. <http://www.audubon.org/magazine/summer-2017/how-genetically-modified-mice-could-one-day-save>.

Britton-Davidian, J., Catalan, J., da Graça Ramalhinho, M., Ganem, G., Auffray, J.C., Capela, R., Biscoito, M., Searle, J.B. and da Luz Mathias, M. (2000). 'Environmental genetics: Rapid chromosomal evolution in island mice'. *Nature* 403: 158.

Broome, K.G., Fairweather, A.A.C. and Fisher, P. (2015). *Brodifacoum Pesticide Information Review. Version 2015/1*. Hamilton, NZ: Department of Conservation internal document DOCDM-25436.

Buckle, A. and Prescott, C. (2012). *The Current Status of Anticoagulant Resistance in Rats and Mice in the UK*. Unpublished report from the Rodenticide Resistance Action Group of the United Kingdom to the Health and Safety Executive. Reading, UK: The University of Reading. <http://www.pestmagazine.co.uk/_attachments/Resources/624_S4.pdf>.

Burt, A. (2003). 'Site-specific selfish genes as tools for the control and genetic engineering of natural populations'. *Proceedings of the Royal Society of London B: Biological Sciences* 270: 921–928.

Campbell, K.J., Beek, J., Eason, C.T., Glen, A.S., Godwin, J., Gould, F., Holmes, N.D., Howald, G.R., Madden, F.M., Ponder, J.B., Threadgill, D.W., Wegmann, A. and Baxter, G.S. (2015). 'The next generation of rodent eradications: Innovative technologies and tools to improve species specificity and increase their feasibility on islands'. *Biological Conservation* 185: 47–58.

Capizzi, D., Bertolino, S. and Mortelliti, A. (2014). 'Rating the rat: Global patterns and research priorities in impacts and management of rodent pests'. *Mammal Review* 44: 148–162.

Champer, J., Buchman, A. and Akbari, O.S. (2016). 'Cheating evolution: Engineering gene drives to manipulate the fate of wild populations'. *Nature Reviews Genetics* 17: 146–159.

Cunningham, C., Ruff, J., Chase, K., Potts, W. and Carrier, D. (2013). 'Competitive ability in male house mice (*Mus musculus*): Genetic influences'. *Behavior Genetics* 43: 151–160.

Dhole, S., Vella, M.R., Lloyd, A.L. and Gould, F. (2018). 'Invasion and migration of spatially self-limiting gene drives: A comparative analysis'. *Evolutionary Applications* 11: 794–808.

DiCarlo, J.E., Chavez, A., Dietz, S.L., Esvelt, K.M. and Church, G.M. (2015). 'Safeguarding CRISPR-Cas9 gene drives in yeast'. *Nature Biotechnology* 33: 1250–1255.

DIISE. (2016). 'The Database of Island Invasive Species Eradications, developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand'. <http://diise.islandconservation.org>. Accessed 16 February 2016.

Doherty, T.S., Glen, A.S., Nimmo, D.G., Ritchie, E.G. and Dickman, C.R. (2016). 'Invasive predators and global biodiversity loss'. *Proceedings of the National Academy of Sciences*113: 11261–11265.

Dubois, S., Fenwick, N., Ryan, E.A., Baker, L., Baker, S.E., Beausoleil, N.J., Carter, S., Cartwright, B., Costa, F., Draper, C., Griffin, J., Grogan, A., Howald, G., Jones, B., Littin, K.E., Lombard, A.T., Mellor, D.J., Ramp, D., Schuppli, C.A. and Fraser, D. (2017). 'International consensus principles for ethical wildlife control'. *Conservation Biology* 31: 753–760.

Eason, C.T., Murphy, E.C., Wright, G.R.G. and Spurr, E.B. (2002). 'Assessment of risks of brodifacoum to non-target birds and mammals in New Zealand'. *Ecotoxicology* 11: 35–48.

Egelie, K.J., Graff, G.D., Strand, S.P. and Johansen, B. (2016). 'The emerging patent landscape of CRISPR-Cas gene editing technology'. *Nature Biotechnology* 34: 1025–1031.

Eggers, S., Ohnesorg, T. and Sinclair, A. (2014). 'Genetic regulation of mammalian gonad development'. *Nature Reviews Endocrinology* 10: 673–683.

Esvelt, K.M., Smidler, A.L., Catteruccia, F. and Church, G.M. (2014). 'Concerning RNA-guided gene drives for the alteration of wild populations'. *eLife*: 10.7554/eLife.03401.

Fitzgerald, G. (2009). *Public Attitudes to Current and Proposed Forms of Pest Animal Control. A summary and review of the Australasian and selected international research.* Unpublished report. Canberra, Australia: Invasive Animals Cooperative Research Centre. <https://www.pestsmart.org.au/public-attitudes-to-current-and-proposed-forms-of-pest-animal-control/>.

Gantz, V.M. and Bier, E. (2015). 'The mutagenic chain reaction: A method for converting heterozygous to homozygous mutations'. *Science* 348: 442–444.

Gantz, V.M., Jasinskiene, N., Tatarenkova, O., Fazekas, A., Macias, V.M., Bier, E. and James, A.A. (2015). 'Highly efficient Cas9-Mediated gene drive for population modification of the malaria vector mosquito *Anopheles stephensi*'. *Proceedings of the National Academy of Sciences* 112: E6736–E6743.

Gemmell, N.J. and Tompkins, D.M. (2017). 'Gene drives and rodent control: Response to Piaggio et al.'. *Trends in Ecology and Evolution* 32: 314–315.

Gray, S.J. and Hurst, J.L. (1998). 'Competitive behaviour in an island population of house mice, *Mus domesticus*'. *Animal Behaviour* 56: 1291–1299.

Guénet, J.-L. and Bonhomme, F. (2003). 'Wild mice: An ever-increasing contribution to a popular mammalian model'. *Trends in Genetics* 19: 24–31.

Hammond, A., Galizi, R., Kyrou, K., Simoni, A., Siniscalchi, C., Katsanos, D., Gribble, M., Baker, D., Marois, E., Russell, S., Burt, A., Windbichler, N., Crisanti, A. and Nolan, T. (2016). 'A CRISPR-Cas9 gene drive system targeting female reproduction in the malaria mosquito vector *Anopheles gambiae*'. *Nature Biotechnology* 34: 78–83.

Hartl, D.L. and Clark, A.G. (2006). *Principles of Population Genetics*. Oxford: Sinauer Associates.

Harvey-Samuel, T., Ant, T. and Alphey, L. (2017). 'Towards the genetic control of invasive species'. *Biological Invasions* 19: 1683–1703.

Harvey-Samuel, T.O., Campbell, K.J., Edgington, M., and Alphey, L. (2019). 'Trialling gene drives to control invasive species: what, where and how?' In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 618–627. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Holmes, N.D., Campbell, K.J., Keitt, B.S., Griffiths, R., Beek, J., Donlan, C.J. and Broome, K.G. (2015). 'Reporting costs for invasive vertebrate eradications'. *Biological Invasions* 17: 2913–2925.

Howald, G., Donlan, C.J., Galván, J.P., Russell, J., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Jones, C.S., Noble, L.R., Jones, J., Tegelstrom, H., Triggs, G.S. and Berry, R. (1995). 'Differential male genetic success determines gene flow in an experimentally manipulated mouse population'. *Proceedings of the Royal Society of London. Series B: Biological Sciences* 260: 251–256.

Jones, H.P., Tershy, B.R., Zavaleta, E.S., Croll, D.A., Keitt, B.S., Finklestein, M.E. and Howald, G.R. (2008). 'Severity of the effects of invasive rats on seabirds: A global review'. *Conservation Biology* 22: 16–26.

Jones, H.P., Holmes, N.D., Butchart, S.H.M., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P., Cuthbert, R.J., Ebbert, S., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R. and Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences* 113: 4033–4038.

Kanavy, D. and Serr, M. (2017). 'Sry gene drive for rodent control: Reply to Gemmell and Tompkins'. *Trends in Ecology and Evolution* 32: 315–316.

Kashimada, K. and Koopman, P. (2010). 'Sry: The master switch in mammalian sex determination'. *Development* 137: 3921–3930.

Kinnear, J., Pentland, C., Moore, N. and Krebs, C. (2016). 'Fox control and 1080 baiting conundrums: Time to prepare for a CRISPR solution'. *Australian Mammalogy* 39: 127–136.

Koopman, P., Gubbay, J., Vivian, N. and Goodfellow, P. (1991). 'Male development of chromosomally female mice transgenic for Sry'. *Nature* 351: 117–121.

Kurle, C.M., Croll, D.A. and Tershy, B.R. (2008). 'Introduced rats indirectly change marine rocky intertidal communities from algae- to invertebrate-dominated'. *Proceedings of the National Academy of Sciences* 105: 3800–3804.

Leitschuh, C.M., Kanavy, D., Backus, G.A., Valdez, R.X., Serr, M., Pitts, E.A., Threadgill, D. and Godwin, J. (2018). 'Developing gene drive technologies to eradicate invasive rodents from islands'. *Journal of Responsible Innovation*. 5(supl): S121–S138.

Lindholm, A.K., Dyer, K.A., Firman, R.C., Fishman, L., Forstmeier, W., Holman, L., Johannesson, H., Knief, U., Kokko, H. and Larracuente, A.M. (2016). 'The ecology and evolutionary dynamics of meiotic drive'. *Trends in Ecology and Evolution* 31: 315–326.

Lyon, M.F. (2003). 'Transmission ratio distortion in mice'. *Annual Review of Genetics* 37: 393–408.

Martin, A., Akol, A. and Gross-Camp, N. (2015). 'Towards an explicit justice framing of the social impacts of conservation'. *Conservation and Society* 13: 166–178.

McShane, T.O., Hirsch, P.D., Trung, T.C., Songorwa, A.N., Kinzig, A., Monteferri, B., Mutekanga, D., Van Thang, H., Dammert, J.L. and Pulgar-Vidal, M. (2011). 'Hard choices: Making trade-offs between biodiversity conservation and human well-being'. *Biological Conservation* 144: 966–972.

Meerburg, B.G., Singleton, G.R. and Kijlstra, A. (2009). 'Rodent-borne diseases and their risks for public health'. *Critical Reviews in Microbiology* 35: 221–270.

NASEM. (2016). *Gene Drives on the Horizon: Advancing Science, Navigating Uncertainty, and Aligning Research with Public Values*. Washington DC: Committee on Gene Drive Research in Non-Human Organisms: Recommendations for Responsible Conduct; Board on Life Sciences; Division on Earth and Life Studies; National Academies of Sciences, Engineering, and Medicine. <https://www.nap.edu/catalog/23405/gene-drives-on-the-horizon-advancing-science-navigating-uncertainty-and>.

NCSU Genetic Engineering and Society Center. (2016). *A Roadmap to Gene Drives: A Deliberative Workshop to Develop Frameworks for Research and Governance*. North Carolina State University Genetic Engineering and Society Center. <https://research.ncsu.edu/ges/research/projects/gene-drives-grant/>. Accessed 30 June 2017.

New Zealand. (2016). *Predator Free NZ 2050 to be a Massive Team Effort*. Wellington, NZ: New Zealand Government, 25 July 2016. <https://www.beehive.govt.nz/release/predator-free-nz-2050-be-massive-team-effort>. Accessed 15 July 2017.

Owens, B. (2017). 'The big cull'. *Nature* 541: 148–150.

Parkes, J.P. and Panetta, F.D. (2009). 'Eradication of Invasive Species: Progress and Emerging Issues in the 21st Century'. In*:* M.N. Clout and P.A. Williams (eds.) *Invasive Species Management. A Handbook of Principles and Techniques,* pp. 47–60. New York, USA: Oxford University Press.

Parkes, J., Fisher, P. and Forrester, G. (2011). 'Diagnosing the cause of failure to eradicate introduced rodents on islands: Brodifacoum versus diphacinone and method of bait delivery'. *Conservation Evidence* 8: 100–106.

Pawluk, A., Amrani, N., Zhang, Y., Garcia, B., Hidalgo-Reyes, Y., Lee, J., Edraki, A., Shah, M., Sontheimer, E.J. and Maxwell, K.L. (2016). 'Naturally occurring off-switches for CRISPR-Cas9'. *Cell* 167: 1829–1838.

Phifer-Rixey, M. and Nachman, M.W. (2015). 'The natural history of model organisms: Insights into mammalian biology from the wild house mouse *Mus musculus*'. *eLife* 4: e05959.

Piaggio, A.J., Segelbacher, G., Seddon, P.J., Alphey, L., Bennett, E.L., Carlson, R.H., Friedman, R.M., Kanavy, D., Phelan, R., Redford, K.H., Rosales, M., Slobodian, L. and Wheeler, K. (2017). 'Is it time for synthetic biodiversity conservation?'. *Trends in Ecology and Evolution* 32: 97–107.

Prowse, T.A.A., Cassey, P., Ross, J.V., Pfitzner, C., Wittmann, T. and Thomas, P. (2017). 'Dodging silver bullets: Good CRISPR gene-drive design is critical for eradicating exotic vertebrates'. *Proceedings of the Royal Society B: Biological Sciences* 284(1860): 20170799.

Rauch, B.J., Silvis, M.R., Hultquist, J.F., Waters, C.S., McGregor, M.J., Krogan, N.J. and Bondy-Denomy, J. (2017). 'Inhibition of CRISPR-Cas9 with bacteriophage proteins'. *Cell* 168: 150–158.

Reese, A. (2017). 'New Zealand aims to eradicate invasive predators, but winning public support may be big challenge'. *Science* 10 July 2017. <http://www.sciencemag.org/news/2017/07/new-zealand-aims-eradicate-invasive-predators-winning-public-support-may-be-big>.

Robertson, B.C. and Gemmell, N.J. (2004). 'Defining eradication units to control invasive pests'. *Ecology* 41: 1042–1048.

Rueda, D., Campbell, K.J., Fisher, P., Cunninghame, F. and Ponder, J.B. (2016). 'Biologically significant residual persistence of brodifacoum in reptiles following invasive rodent eradication, Galapagos Islands, Ecuador'. *Conservation Evidence* 13: 38.

Rueda, D., Carrion, V., Castaño, P.A., Cunninghame, F., Fisher, P., Hagen, E., Ponder, J.B., Riekena, C.A., Sevilla, C., Shield, H., Will, D. and Campbell, K.J. (2019). 'Preventing extinctions: planning and undertaking invasive rodent eradication from Pinzon Island, Galapagos' In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 51–56. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Schimenti, J. (2014). 'The Mouse T Complex'. In*:* E.C.R. Reeve and I. Black (eds.) *Encyclopedia of Genetics,* pp. 287–292. Chicago, USA: Fitzroy Dearborn.

Serr, M. and Godwin, J. (2019). 'Towards a genetic approach to invasive rodent eradications: assessing reproductive competitiveness between wild and laboratory mice'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 64–70. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Singh, P., Schimenti, J.C. and Bolcun-Filas, E. (2015). 'A mouse geneticist's practical guide to CRISPR applications'. *Genetics* 199: 1–15.

Singleton, G.R., Brown, P.R., Pech, R.P., Jacob, J., Mutze, G.J. and Krebs, C.J. (2005). 'One hundred years of eruptions of house mice in Australia–a natural biological curio'. *Biological Journal of the Linnean Society* 84: 617–627.

Sinkins, S.P. and Gould, F. (2006). 'Gene drive systems for insect disease vectors'. *Nature Reviews Genetics* 7: 427–435.

Stenseth, N.C., Leirs, H., Skonhoft, A., Davis, S.A., Pech, R.P., Andreassen, H.P., Singleton, G.R., Lima, M., Machang'u, R.S. and Makundi, R.H. (2003). 'Mice, rats, and people: The bio-economics of agricultural rodent pests'. *Frontiers in Ecology and the Environment* 1: 367–375.

TIB Partners. (2014). *The Threatened Island Biodiversity Database (TIB): Developed by Island Conservation, University of California Santa Cruz Coastal Conservation Action Lab, Birdlife International and IUCN Invasive Species Specialist Group. Version 2014.1*. <www.tib.islandconservation.org>. Accessed 23 December 2016.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?'. *Biological Invasions* 8: 863–891.

Vidal, V.P., Chaboissier, M.-C., de Rooij, D.G. and Schedl, A. (2001). 'Sox9 induces testis development in XX transgenic mice'. *Nature Genetics* 28: 216–217.

Vignieri, S.N., Larson, J.G. and Hoekstra, H.E. (2010). 'The selective advantage of crypsis in mice'. *Evolution* 64: 2153–2158.

Webber, B.L., Raghu, S. and Edwards, O.R. (2015). 'Opinion: Is CRISPR-based gene drive a biocontrol silver bullet or global conservation threat?'. *Proceedings of the National Academy of Sciences* 112: 10565–10567.

Black rat eradication on Italian islands: planning forward by looking backward

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Abstract Since 1999, the black rat (*Rattus rattus*) has been eradicated from 14 Italian islands, and eradication is ongoing on a further five islands. Most projects were funded by the European Union (EU) Life Programme. Over the years, eradication techniques have been improved and adapted to different situations, including aerial bait distribution on islands with large inaccessible areas, which otherwise would have relied on a manual bait distribution. A priority list of eradications on islands, which was compiled ten years ago, has been met to a large extent, as rats have been successfully eradicated from many islands of great importance to breeding seabirds. Despite some cases of re-invasion occurring in early projects, advances in biosecurity measures have allowed for eradications on islands where this was previously considered unfeasible due to a high risk of re-invasion. This paper reports on black rat eradication work performed on Italian Mediterranean islands with small villages. We show biodiversity benefits of these programmes, but also qualitatively address socio-economic and health impacts on local communities. Eradication projects have faced new obstacles, due to recent changes in legislation which complicated the application of rodenticides and made it very difficult to get permission for aerial distribution of bait on some of the priority islands.

Keywords: biosecurity, cost-effectiveness, invasive alien species, *Rattus rattus*, reinvasion, shearwaters

INTRODUCTION

In recent years, a growing awareness about the importance of the threat posed by alien species on native ecosystems has driven an increasing number of interventions aimed at eliminating or mitigating their impacts. Much of the effort to restore native ecosystems has been directed towards islands, which represent ideal environments for implementing eradication actions, because the impact of alien species may be especially important (e.g. Manne, et al., 1999; Baillie, et al., 2004), and their natural isolation helps to maintain the benefits achieved.

On Italian islands, measures to eradicate rats have had great success. Since the late 1990s, rats have been eradicated, or locally controlled on many islands (Capizzi, et al., 2016), with the EU Life programme providing important financial support, making it possible to achieve significant conservation objectives.

Rat eradications were carried out over the years on islands with different characteristics, and experience built up in selecting context-sensitive materials, techniques and strategies. Indeed, activities were carried out on islands small and large, uninhabited or with small residential areas, flat or with very rough terrain and with significant differences in the presence of non-target species.

Although there have been successes over the years, some mistakes have also been made. In our opinion, a critical review encompassing the activities so far carried out, along with the results achieved, can help to effectively plan future eradications.

In this paper, we review the rat eradication actions carried out in past years as well as those currently implemented, highlighting the progress, problems and constraints experienced so far, and analyse the strengths and weaknesses of the solutions adopted. Our aim is to show that a review of past experiences can have a positive influence on planning for future eradication attempts.

Evolution of techniques and targets

Priority list

Since resources for conservation actions are limited, priority setting is considered a key aspect in defining conservation strategies (Hughey, et al., 2003; Joseph, et al., 2009), including those involving invasive alien species management (e.g. Gallardo & Aldridge, 2013). Capizzi, et al. (2010) established a priority list of islands for rat eradication on Italian islands, considering the optimal allocation of available resources. This prioritisation considered the number of shearwater pairs and the monetary costs of rat eradication on each island, as well as the risk of reinvasion. To date, all the islands in the top five (and seven in the top ten) were included in eradication projects performed or still ongoing (Table 1). Furthermore, recent advances in biosecurity measures have allowed the carrying out or planning of eradications on islands that were previously not included on the priority list because of a high risk of reinvasion, such as Linosa (eradication ongoing) and Ventotene (eradication ongoing).

Island size

Since our eradication projects began, the number of rat free islands has increased considerably. This was possible due to increased experience and confirmation that these interventions bring substantial benefits to birds (see below).

The first eradications in 1999–2000 were carried out on islands of a few hectares (Table 2), but since 2005 rats have been eradicated from islands with an area of over 100 hectares (Zannone, Giannutri, Molara). Since 2012, islands with over 1,000 hectares have also been attempted (success declared in 2014 for Montecristo, ongoing actions on Tavolara and Pianosa).

Field techniques

Bait delivery

In the first eradication programmes, rodenticide baits were placed inside bait stations, at a relatively high density (about 10 stations/ha). In subsequent eradications, involving islands larger than 100 ha (i.e. Giannutri and Zannone, between 2005 and 2006), bait station density was reduced to an average of 4/ha. On Zannone, given its relatively rough terrain, bait distribution, in some inaccessible areas, was carried out by hand-broadcasting from a helicopter, using rodenticide bait blocks, which were secured inside biodegradable dispensers (sections

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Table 1 List of islands prioritised for rat eradication (from Capizzi, et al., 2010) and status of eradication interventions. Crosses indicate the presence of the two shearwater species on the various islands.

		Scopoli's shearwater (*Calonectris diomedea*)	Yelkouan shearwater (*Puffinus yelkouan*)	
1	Tavolara	X	X	Eradication planned in 2017
2	Palmarola	X	X	Eradication planned in 2018
3	Montecristo	X	X	Eradication in 2012
4	Pianosa Group (La Scola and Pianosa)	X		Eradication in La Scola (2000), and Pianosa (2017)
5	Giannutri	X	X	Eradication in 2005
6	Santa Maria Group (14 islands)	X	X	No action
7	Molara		X	Eradication in 2009
8	Zannone	X	X	Eradication in 2006
9	Spargi	X	X	No action
10	Soffi Group (four islands)	X		No action

of bamboo trunk). On larger, mainly inaccessible islands, aerial distribution was carried out on the whole island. Bait, in the form of pellets, was distributed using helicopters, with an automated distributor (bucket) purchased in 2008 and used by all projects since then.

Optimisation of active ingredients

In the first eradication projects (e.g. those in 1999–2000 on small islands), both bromadiolone and brodifacoum were used, regardless of the presence of non-target animals. In subsequent years, on larger (> 100 ha) islands (i.e. Giannutri, Zannone and Molara), we relied solely on brodifacoum, which was judged, on the basis of published data, to be the most effective and the most used active ingredient (e.g. Howald, et al., 2007; Buckle & Eason, 2015). However, when dealing with inhabited islands with pets (e.g. Linosa and Ventotene, where eradication is ongoing) and livestock (Pianosa, Tavolara), we chose to perform a two-stage bait distribution, with a different active ingredient. In the first phase (first two distributions), when rat populations were still at a high level, a bait containing an active ingredient less toxic for non-target species was used (e.g. bromadiolone or difenacoum, e.g. Capizzi & Santini, 2007; Buckle & Eason, 2015), thereby reducing the risks of secondary poisoning for animals that could eat dead or dying rats. The use of brodifacoum was limited to the last two applications (second phase), when the population of rats was expected to have been decimated by previous baiting campaigns, and therefore the risk of a poisoned rat (or mouse) being eaten by a non-target species was much lower.

Biosecurity issues

Rat reinvasion following an eradication programme is a real threat (Russell & Clout, 2005; Russell & Clout, 2007), wasting a great deal of time and monetary effort. In recent years, rats have reinvaded some of the islands where they had been previously eradicated (Table 3). Reinvasion occurred as rats swam from neighbouring islands or the mainland (maximum distance of reinvaded islands: 320 m, average distance: 218.6 ± 102.7 m). In the case of Molara, the hypothesis of an unsuccessful eradication was not supported by evidence, as genetic analyses have shown that the reinvading rats were different from the eradicated ones (Ragionieri, et al., 2013). The distance of Molara from other neighbouring islands and the mainland (1,400 m), plus the simultaneous appearance of rabbits, suggests that they have been transported by boat. However, recent progress in the understanding of biosecurity measures, i.e. a better understanding of rat swimming abilities as well as of effective quarantine measures (Russell, et al., 2008; Oppel, et al., 2011), allowed us to plan and complete eradication programmes on islands where there is a boat service and on islands with small villages. Therefore, in 2016, rat eradication was achieved on Linosa, which has a small village of about 500 people, and has just started (January 2018) on Ventotene, which has about 700 residents. If the Ventotene rat eradication is successful, it will be the largest inhabited island in the Mediterranean cleared of rats.

Ecological and socio-economic benefits

Benefits for shearwaters

The detrimental impact of rats on nesting shearwaters has been well documented on several islands, both oceanic and Mediterranean. In the Mediterranean, observed population declines of burrowing seabirds such as Scopoli's shearwater (*Calonectris diomedea*), yelkouan shearwater (*Puffinus yelkouan*), Balearic shearwater (*P. mauretanicus*) and storm petrel (*Hydrobates pelagicus*) was mainly attributed to alien predators, especially rats (e.g. Penloup, et al., 1997; Martin, et al., 2000; Igual, et al., 2006; Baccetti, et al., 2009). Detailed surveys on Italian islands (for survey methods see Baccetti, et al., 2009) corroborated the evidence, showing a large difference in terms of breeding success between islands with or without rats; the latter included both islands where rats had never been present and where they had been eradicated (Capizzi, et al., 2016, Fig. 1). Pooled data from both Scopoli's shearwater and yelkouan shearwater indicated that breeding pairs on islands without rats had much higher breeding success (0.78±0.17, n=15) than those breeding on islands with rats (0.14±0.25, n=11). Rat removal also affected the size of shearwater colonies. At La Scola, ten years after rat eradication, the colony of Scopoli's shearwater increased from 60–100 pairs in 2001 to 150–250 pairs in 2010. At Zannone, after rat eradication (2007) there was an increase in the Scopoli's shearwater colony from 27 pairs in 2007 to 80 pairs in 2016.

The completed rat eradications have rendered over 1,500 ha rat-free, and ongoing or planned projects will likely increase this surface area to 4,500 ha (Fig. 2). Currently,

Table 2 Summary table showing the Italian islands where rat eradication was completed in the period 1999–2017, and those where the intervention is scheduled in coming months, with details on the islands, the interventions and project details. Success (i.e. successful eradication) was established two years after the last sign of rats.

Year	Island	Region	Area (ha)	Distance (m)	Active ingredient	Bait method	Responsible (funding)	Outcome
1999	Isolotto di Porto Ercole	Tuscany	6.5	320	Bromadiolone, brodifacoum	bait station	National Park	successful, reinvaded
1999	Isola dei Topi	Tuscany	1.3	300	Bromadiolone, brodifacoum	bait station	National Park	successful, reinvaded
1999	Peraiola	Tuscany	1	30	Bromadiolone, brodifacoum	bait station	National Park	successful
1999	Palmaiola	Tuscany	7.2	2,950	Bromadiolone, brodifacoum	bait station	National Park	successful
1999	Gemini Alta	Tuscany	1.9	48	Bromadiolone, brodifacoum	bait station	National Park	successful, reinvaded
1999	Gemini Bassa	Tuscany	1.6	120	Bromadiolone, brodifacoum	bait station	National Park	successful, reinvaded
2001	La Scola	Tuscany	1.6	242	Bromadiolone, brodifacoum	bait station	National Park	successful, new incursions (3) promptly eradicated
2006	Giannutri	Tuscany	239.4	11,471	Brodifacoum	bait station	National Park	successful
2007	Zannone	Latium	104.7	5,700	Brodifacoum	bait station	Circeo National Park	successful
2008	Molara	Sardinia	347.9	1,400	Brodifacoum	aerial	MPA	successful, reinvaded in 2010
2008	Proratora	Sardinia	4.5	200	Brodifacoum	bait station	MPA	successful, immediately reinvaded, eradicated 2010, reinvaded in 2010
2010	Isola Piana	Sardinia	13.6	551	Brodifacoum	bait station	MPA	successful
2010	Isola dei Cavalli	Sardinia	2.2	300	Brodifacoum	bait station	MPA	successful, new incursions (2) promptly eradicated
2012	Montecristo	Tuscany	1071	29,410	Brodifacoum	aerial	National Park	successful
2016–2017	Linosa	Sicily	545.1	43,000	Difenacoum & brodifacoum	bait station	Sicily Region (LIFE)	to be confirmed
2017	Pianosa	Tuscany	1026	13,300	Bromadiolone & brodifacoum	bait station	National Park	to be confirmed
2017	Tavolara	Sardinia	602.0	1,150	Brodifacoum	aerial	Municipality of Olbia (LIFE)	started in autumn 2017
2018	Palmarola	Latium	125.1	7,300	Brodifacoum	bait station	Latium Region (LIFE)	started in January 2018
2018	Ventotene	Latium	143.6	43,000	Bromadiolone & brodifacoum	bait station	Latium Region (LIFE)	started in January 2018

Distance = from mainland or other islands in metres. National Park = National Park of Tuscan Archipelago (LIFE). MPA = Marine Protected Area of Tavolara – Punta Coda Cavallo

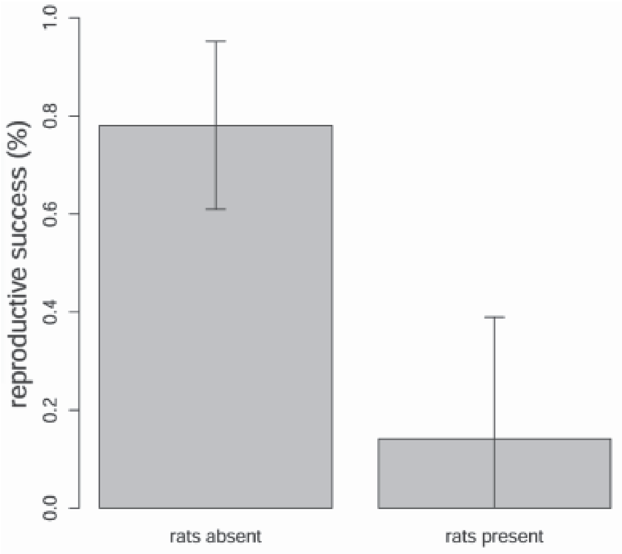

Fig. 1 Boxplot showing mean and standard deviation of breeding success (in terms of percent chick survival) of both shearwater species on Italian islands with (n=11) and without (n=15) rats.

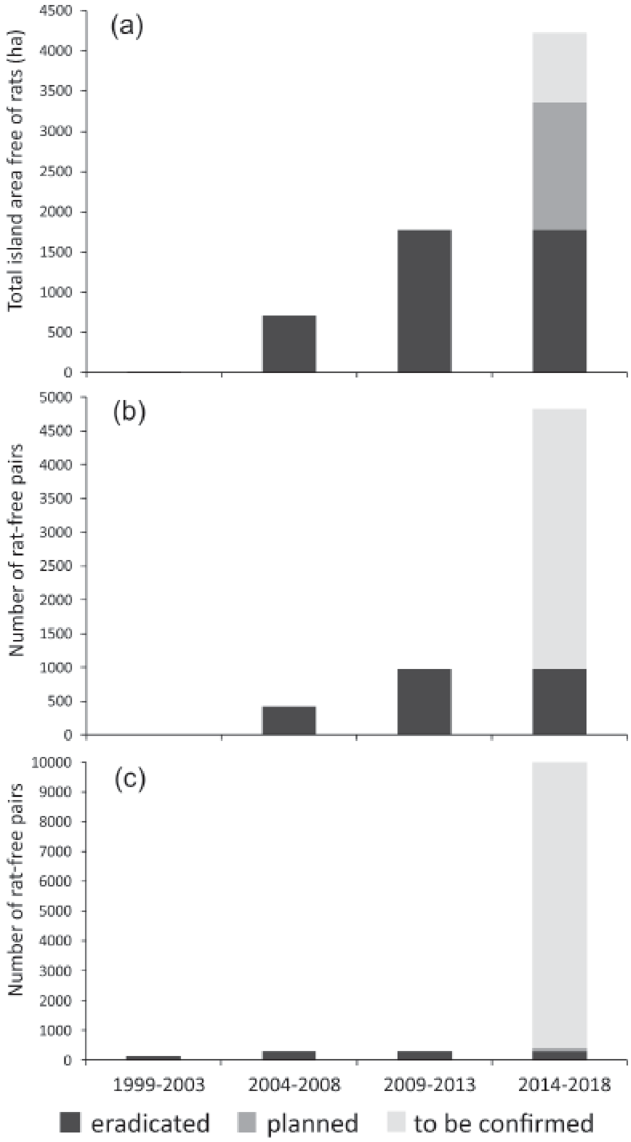

Fig. 2 Results of rat eradications on Italian islands in terms of pest free area and native species recovery since 1999 in five-year intervals a) total island surface area (ha) freed of rats, b) number of pairs of yelkouan shearwater (*Puffinus yelkouan*) and of c) Scopoli's shearwater (*Calonectris diomedea*) released from rat predation. The graphs also include eradications where the outcome is still to be confirmed, as well as those planned in the coming months.

15% of the Italian population of shearwater pairs (both *Calonectris diomedea* and *Puffinus yelkouan* computed as the geometric mean of minimum and maximum estimates, data from Baccetti, et al., 2009, updated when necessary) have been released from rat predation (Fig. 3). Increased benefits to the Italian population will occur with ongoing and planned eradications (i.e. Linosa for Scopoli's shearwater and Tavolara for yelkouan shearwater).

Socioeconomic and public health issues

Islands where rats have been eradicated are uninhabited or host just a few houses. Recently, the possibility of conducting rat eradication programmes on islands with small villages (Linosa, 500 residents, and Ventotene, 700) also provides significant socio-economic and health benefits for residents and tourists (see below). As an example, in Ventotene (120 ha, 700 inhabitants, rat eradication funded within Life PonDerat project), we ran a preliminary survey (performed through interviews to residents, which is still ongoing) to estimate the economic benefits when removing rats. First, in terms of prevented management costs, we estimated the current yearly quantity of rodenticides used to protect crops from rat damage at about 100 kg, corresponding to a yearly overall cost of about €5000. Also, the municipality runs its own pest control activities in public areas, hiring the service of a pest control company at an annual cost of about €3000. Second, rat eradication brings biodiversity benefits. As bait is generally used improperly, by using the most toxic active ingredients (usually brodifacoum) and by distributing baits indiscriminately, the risk to non-target species is apparent. Eradication would reduce these non-target effects. Third, direct damage costs are prevented because a certain amount of crop damage still occurs despite the current use of rodenticides which would also be prevented if rats were eradicated.

Lastly, rat eradication brings health benefits. For example, we recorded a 15.5% prevalence of *Leishmania infantum* in *Rattus rattus* from Montecristo, an island far from the mainland without carnivores (except the sporadic presence of dogs), leading us to identify rats as possible reservoirs and vectors of this protozoan (Zanet, et al., 2014). On inhabited islands (e.g. Ventotene and Linosa), it is likely that rat removal will bring health benefits by reducing the impact of rodent borne diseases, although social costs associated with rodent-borne diseases are difficult to quantify (e.g. World Bank, 2010). On Ventotene, the challenge is to obtain an overall estimate of the benefits of eradication, both ecological and socio-economic (García-Llorente, et al., 2008).

Therefore, the associated economic benefits should also be considered when evaluating the cost-effectiveness of these conservation efforts, as they may confer an added value that can help with public acceptance of this type of project.

Impact on non-target species

Conservationists, researchers and land managers can look pragmatically at the possible loss of individual non-target species, by comparing them with the increased benefits to native species and ecosystems (e.g. Ogden & Gilbert, 2009; Capizzi, et al., 2010; Gillespie & Bennett,

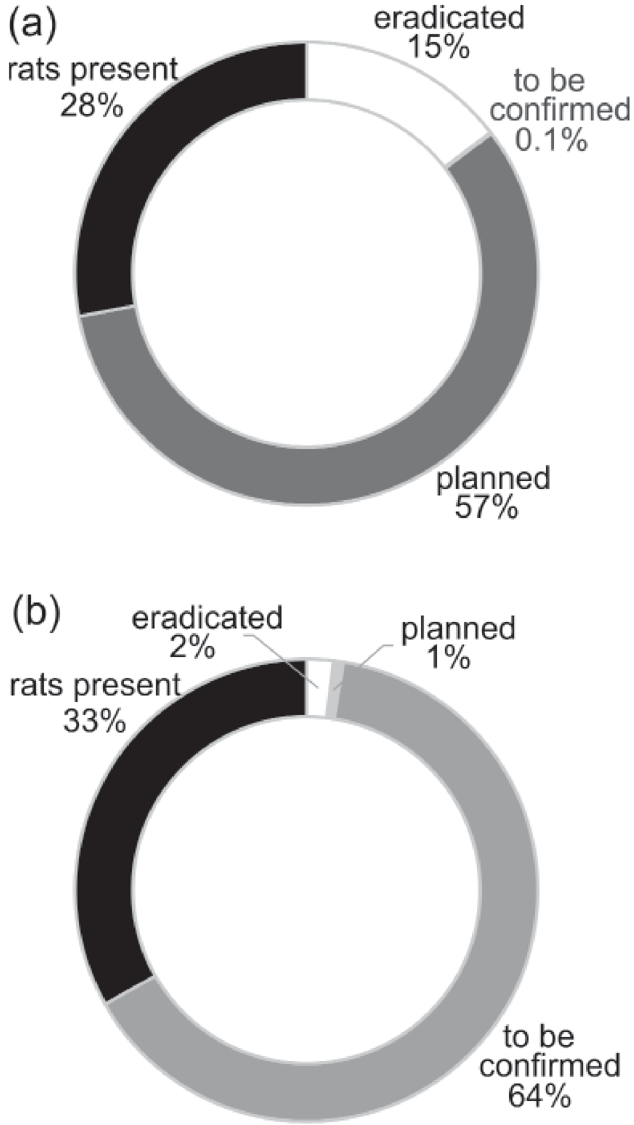

Fig. 3 Percentage of the Italian population of yelkouan shearwater (a) and Scopoli's shearwater (b) nesting pairs released from rat predation.

2017). However, minimal impacts on non-target species are often crucial to the acceptance of the project by the general public and are a significant factor in obtaining authorisation from public authorities. Indeed, much of the concerns of the public and public authorities were around the impact on non-target species, which has been demonstrated to be almost negligible (Capizzi, et al., 2016). In a few cases, the actual non-target impact involved species that, following rat removal, would have become extinct anyway, i.e. a few pairs of nocturnal raptors (barn owl, *Tyto alba*). We did not observe any impact on other rat predators, such as snakes (green whip snake, *Hierophis viridiflavus* and the asp viper, *Vipera aspis*), or birds of prey (kestrel, *Falco tinnunculus* and peregrine falcon, *Falco peregrinus*).

In most cases, populations of lizards (both *Podarcis sicula* and *P. muralis*) and native geckos have increased since rat eradication. The populations of wild or feral ungulates (mouflons and goats, in most cases alien species themselves) did not experience significant impacts, despite some losses of goats on Montecristo. Finally, no impact on pets (dogs and cats), poultry or livestock has been recorded so far.

Unsolved problems and lessons learnt

Authorisation and legal aspects

Limitations resulting from the application of EU Biocide Regulation 528/2012 represent a major obstacle to running eradication programmes, even though this Regulation explicitly accommodates a derogation on the use of rodenticides (Article 43), including aspects relating to the protection of the environment. Italian authorities interpreted the European regulation on biocides to mean that they should only be distributed inside bait stations, thus implicitly forbidding aerial distribution. This has led to legal disputes during the eradication on Montecristo, which were resolved but will cause problems for many eradications to come. For instance, the derogation for aerial distribution on Tavolara (which hosts the largest colony of *Puffinus yelkouan* in the world) was only obtained more than one year after the original request, thereby risking the loss of funding and compromising the outcome of the project.

Dealing with stakeholders

It is well known that communication and information aspects are very important in projects involving the suppression or removal of invasive species to favour native species or ecosystems (e.g. Larson, et al., 2011; Adriaens, et al., 2015). In the case of island communities, the main issue is that, if not properly communicated, actions may be perceived as an intrusion by outsiders. On-site meetings with island inhabitants do not always receive good feedback. In our experience, ensuring a constant presence in the area and establishing positive relationships with locals are paramount to raising public awareness on relevant conservation topics, as well as gaining project acceptance. Public approval is indeed a key factor for rat eradication success on islands (Epanchin-Niell, et al., 2010).

It is also vital to establish a constructive dialogue with port authorities and ship owners, to allow boats and harbours to be monitored, so that rats cannot be transported with the possibility of them being distributed across the island. This is especially important on islands served by regular ship visits, such as those hosting small villages (e.g. projects ongoing on Linosa and Ventotene).

Learning from failures

As mentioned above, the analysis of recolonisations following eradications has allowed us to conclude that islands closely neighbouring other rat-inhabited islands present a high risk of re-invasion after a successful eradication operation. The case of La Scola Island is representative, with three reinvasions in about fifteen years. The eradication of rats from the nearby (320 m) island of Pianosa will solve the problem permanently. Rat eradication on Molara represents a different case of reinvasion. The island was reinvaded a few months after an apparently successful rat eradication, but invading rats were genetically different from the eradicated rats (Ragionieri, et al., 2013). We strongly suspect that this recolonisation event represents a case of sabotage, possibly caused by the hostility of some people towards the project: the simultaneous appearance of rabbits on the island corroborated this hypothesis. This confirms the importance of properly addressing community opinions (Genovesi & Bertolino, 2001) and trying to highlight critical issues that may otherwise compromise the outcome of the project. To avoid the voluntary release of rats on rat-free islands, it is crucial to implement long term biosecurity and provide the necessary human resources for continuous awareness-raising.

REFERENCES

Adriaens, T., Baert, K., Breyne, P., Casaer, J., Devisscher, S., Onkelinx, T., Pieters, S., Stuyck, J., (2015). 'Successful eradication of a suburban Pallas's squirrel *Callosciurus erythraeus* (Pallas 1779) (Rodentia, Sciuridae) population in Flanders (northern Belgium)'. *Biological Invasions* 17: 2517–2526.

Baccetti, N., Capizzi, D., Corbi, F., Massa, B., Nissardi, S., Spano, G. and Sposimo, P. (2009). 'Breeding shearwater on Italian islands: population size, island selection and co-existence with their main alien predator'. *Rivista Italiana di Ornitologia* 78: 83–99.

Baillie, J.E.M., Hilton-Taylor, C. and Stuart, S.N. (2004). *2004 IUCN Red List of Threatened Species: A Global Species Assessment*. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Buckle, A.P. and Eason, C.T. (2015). 'Control Methods: Chemical'. In: A.P. Buckle and R.H. Smith (eds.) *Rodent Pests and their Control,* pp. 123–154. Wallingford, UK: CABI.

Capizzi, D. and Santini, L. (2007). *I Roditori Italiani: Ecologia, Impatto sulle Attività Umane e sugli Ecosistemi, Gestione delle Popolazioni*. Roma: Antonio Delfino Editore.

Capizzi, D., Baccetti, N. and Sposimo, P. (2010). 'Prioritizing rat eradication on islands by cost and effectiveness to protect nesting seabirds'. *Biological Conservation* 14: 1716–1727.

Capizzi, D., Baccetti, N. and Sposimo, P. (2016). 'Fifteen Years of Rat Eradication on Italian Islands'. In: F.M. Angelici (ed.) *Problematic Wildlife*, pp. 205–227. Switzerland: Springer International Publishing.

Epanchin-Niell, R.S., Hufford, M.B., Aslan, C.E., Sexton, J.P., Port, J.D. and Waring, T.M. (2010). 'Controlling invasive species in complex social landscapes'. *Frontiers in Ecology and the Environment* 8: 210–216.

Gallardo, B. and Aldridge, D.C. (2013). 'Priority setting for invasive species management: risk assessment of Ponto-Caspian invasive species into Great Britain'. *Ecological Applications* 23: 352–364.

García-Llorente, M., Martín-López, B., González, J.A., Alcorlo, P. and Montes, C. (2008). 'Social perceptions of the impacts and benefits of invasive alien species: implications for management'. *Biological Conservation* 141: 2969–2983.

Genovesi, P. and Bertolino, S. (2001). 'Human Dimension Aspects in Invasive Alien Species Issues: The Case of the Failure of the Grey Squirrel Eradication Project in Italy'. In: McNeely, J. (ed.) *The Great Reshuffling: Human Dimensions of Invasive Alien Species*, pp. 113–119. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Gillespie, R. and Bennett, J. (2017). 'Costs and benefits of rodent eradication on Lord Howe Island, Australia'. *Ecological Economics* 140: 215–224.

Howald, G., Donlan, C., Galván, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Hughey, K.F., Cullen, R. and Moran, E. (2003). 'Integrating economics into priority setting and evaluation in conservation management'. *Conservation Biology* 17: 93–103.

Igual, J.M., Forero, M.G., Gomez, T., Orueta, J.F. and Oro, D. (2006). 'Rat control and breeding performance in Cory's shearwater (*Calonectris diomedea*): Effects of poisoning effort and habitat features'. *Animal Conservation* 9: 59–65.

Joseph, L.N., Maloney, R.F. and Possingham, H.P. (2009). 'Optimal allocation of resources among threatened species: A project prioritization protocol'. *Conservation Biology* 23: 328–338.

Larson, D.L., Phillips-Mao, L., Quiram, G., Sharpe, L., Stark, R., Sugita, S. and Weiler, A. (2011). 'A framework for sustainable invasive species management: environmental, social, and economic objectives'. *Journal of Environmental Management* 92: 14–22.

Manne, L.L., Brooks, T.M. and Pimm, S.L. (1999). 'Relative risk of extinction of passerine birds on continents and islands'. *Nature* 399: 258–261.

Martin, J.L., Thibault, J.C. and Bretagnolle, V. (2000). 'Black rats, island characteristics, and colonial nesting birds in the Mediterranean: consequences of an ancient introduction'. *Conservation Biology* 14: 1452–1466.

Ogden, J. and Gilbert, J. (2009). 'Prospects for the eradication of rats from a large inhabited island: Community based ecosystem studies on Great Barrier Island, New Zealand'. *Biological Invasions* 11: 1705–1717.

Oppel, S., Beaven, B.M., Bolton, M., Vickery, J. and Bodey, T.W. (2011). 'Eradication of invasive mammals on islands inhabited by humans and domestic animals'. *Conservation Biology* 25: 232–240.

Penloup, A., Martin, J.L., Gory, G., Brunstein, D. and Bretagnolle, V. (1997). 'Distribution and breeding success of pallid swift, *Apus pallidus*, on Mediterranean islands: Nest predation by the roof rat, *Rattus rattus*, and nest quality'. *Oikos* 80: 78–88.

Ragionieri, L., Cutuli, G., Sposimo, P., Spano, G., Navone, A., Capizzi, D., Baccetti, N. and Fratini, S. (2013). 'Establishing the eradication unit of Molara Island: A case of study from Sardinia, Italy'. *Biological Invasions* 15: 2731–2742.

Russell, J.C. and Clout, M.N. (2005). 'Rodent Incursions on New Zealand islands'. In: J. Parkes, M. Statham and G. Edwards (eds.) *Proceedings of the 13th Australasian Vertebrate Pest Conference*, pp. 324–330. Lincoln, New Zealand: Landcare Research.

Russell, J.C. and Clout, M.N. (2007). 'Early Detection and Eradication of Invading Rats'. In: W.C. Witmer, K.A. Pitt and K.A. Fagerstone (eds.) *Managing Vertebrate Invasive Species*, pp. 268–272. Fort Collins, CO: USDA/APHIS/WS, National Wildlife Research Center.

Russell, J.C., Beaven, B.M., MacKay, J.W.B., Towns, D.R. and Clout, M.N. (2008). 'Testing island biosecurity systems for invasive rats'. *Wildlife Research* 35: 215–221.

World Bank (2010). *People, Pathogens and our Planet*, *Volume 1: Towards a One Health Approach for Controlling Zoonotic Diseases*. Washington: The World Bank.

Zanet, S., Sposimo, P., Trisciuoglio, A., Giannini, F., Strumia, F. and Ferroglio, E. (2014). 'Epidemiology of *Leishmania infantum*, *Toxoplasma gondii*, and *Neospora caninum* in *Rattus rattus* in absence of domestic reservoir and definitive hosts'. *Veterinary Parasitology* 199: 247–249.

Control of house mice preying on adult albatrosses at Midway Atoll National Wildlife Refuge

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Abstract Sand Island, Midway Atoll National Wildlife Refuge (MANWR), is home to 21% of all nesting black-footed albatross (*Phoebastria nigripes*) and 47% of all nesting Laysan albatross (*P. immutabilis*) worldwide. During the 2015–2016 nesting season predation and disturbance by non-native house mice (*Mus musculus*), here documented for the first time, resulted in 70 abandoned nests, 42 adult birds killed and 480 wounded. In the following nesting season the affected area increased, resulting in 242 dead adults, 1,218 injured birds and 994 abandoned nests. Mouse predation activities triggered a mouse control response to reduce mouse densities in the affected areas using multi-catch live traps, kill traps, and limited use of anticoagulant rodenticides in bait stations. In 2016–2017 we applied a pelleted cholecalciferol rodenticide, $AGRID_3$ (Bell Laboratories, Madison, WI), at a rate of 20 kg/ha in all affected areas. The purpose of this study was to evaluate the efficacy of using $AGRID_3$ to reduce mouse density and rate of mouse attacks on nesting albatrosses on Sand Island. Mouse attacks decreased and mouse abundance was reduced following rodenticide applications in the plots treated in December but changes in attack rates in the plots treated in January were not detectable and mouse abundance increased subsequent to treatment. The plots in the December treatments were much larger than those used in January and rainfall rate increased after December. A minimum size of treatment area may be necessary to achieve a reduction in injury rates in albatrosses. No deleterious effects were observed in non-target organisms. The casualties resulting from mouse predation (mostly Laysan albatross) represent a small proportion of the 360,000 pairs nesting on Sand Island. However, the risk to adult breeding albatrosses representing such a large fraction of the global population prompted the United States Fish & Wildlife Service to prioritise mouse control efforts.

Keywords: cholecalciferol, non-target species, Pacific, rodent, seabirds, tropical

INTRODUCTION

Midway Atoll National Wildlife Refuge (MANWR) is home to over three million birds representing 29 species including species of conservation concern and the largest albatross breeding colony in the world. MANWR supports 36% of the earth's black-footed albatross (*Phoebastria nigripes*) and 73% of all Laysan albatross (*P. immutabilis*). Of the three islands that make up the refuge, Sand Island is the largest and provides habitat to approximately 360,000 breeding pairs of Laysan albatross, making it a globally significant colony. House mice (*Mus musculus*) were introduced to Sand Island more than 75 years ago and persisted after black rats (*Rattus rattus*) were eradicated in 1996. Until recently, these non-native mammals appeared to co-exist with the refuge's large seabird populations without harm.

This changed in December 2015 when, shortly after the initiation of the albatross breeding season, severe wounds were discovered on the dorsa of several incubating albatrosses on Sand Island and images from motion-sensing cameras revealed that the source of the wounds were mice (Fig. 1). This was the first time house mice had been observed attacking adult albatrosses and the first documentation of mice preying on albatross in the Northern hemisphere. House mice had not been considered a threat to seabird populations until 2001 when they were found preying on albatross chicks as well as other seabird species at two sites in the Southern hemisphere (Cuthbert & Hilton, 2004; Angel, et al., 2009; Jones & Ryan, 2009).

The discovery of attacks by mice on Sand Island caused immediate concern for wildlife managers at the refuge. Adult mortality has the strongest effect on population growth rates in species such as albatrosses with low fecundity, longevity, high age at first breeding, and prolonged parental care. The loss of the breeding adult is compounded by the loss of its egg or chick, and also reduces the fecundity of its surviving mate, as it often takes more than a year for a widowed bird to find a new mate. In response to the attacks first discovered in December 2015, emergency control efforts were immediately initiated at a 5 m grid resolution over attack areas using a combination of available methods; live traps, kill traps, and difethialone rodenticide applied in bait stations near structures.

When albatrosses returned to Sand Island in the autumn of 2016, surveys were initiated to look for signs of mouse attack and it quickly became clear that mice were attacking the albatross again. Moreover, the rate at which birds were being killed or injured suggested that the 2016–2017 outbreak might be much greater than during the previous year. This time, however, United States Fish & Wildlife Service (USFWS) staff had a plan and were prepared to address the situation. Research had suggested that $AGRID_3$

Fig. 1 Introduced house mouse attacking adult Laysan albatross as it incubates. As captured by a Reconyx trail camera.

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(Bell Laboratories, Madison, WI), a cholecalciferol rodenticide, might provide an effective tool for reducing the number of mice in areas where they were attacking albatross, thereby reducing the impacts to the nesting birds. A plan was developed for applying the rodenticide in affected areas and also for measuring the effects of the treatments on both mice and nesting albatross.

In this paper we describe the mouse predation on albatross that occurred on Sand Island during the 2016–2017 breeding season and the actions taken to abate the threat they imposed on the albatross population there: specifically, a broadcast application of AGRID$_3$ in the areas in which we observed mouse predation on albatrosses. We also describe the monitoring that was undertaken to measure both the direct effects of the rodenticide on the mouse population and the indirect effects that this treatment had on reducing albatross death, injury, and nest abandonment.

MATERIALS AND METHODS

Study area

MANWR is located at the north-west end of the Hawaiian Islands archipelago, 1,930 km from Honolulu, Hawaii at 28.208° N; -177.379° W. One of the oldest atoll formations in the world, MANWR consists of three islands within an 8 km diameter fringing reef. MANWR is classified as a tropical wet/dry savannah with an average annual rainfall of 1,104 mm (43.5 in). MANWR has had a relatively continuous human presence since 1904 when a station was built to support the construction of a trans-Pacific telegraph cable. From 1941 until 1997, Midway Atoll was used by the United States Military during which time both black rats and house mice were introduced. As a consequence, the atoll's ecosystems are highly altered. In 2015 there were 190 species of plants observed, 24 (13%) native and 166 (87%) non-native (Starr & Starr, 2015). The largest, and only, mouse infested island is Sand at 460 ha. MANWR currently supports a resident human community of 50 people along with an operational runway, Henderson Airfield. In 1988 the natural habitats of Midway Atoll began to be managed as part of the National Wildlife Refuge system. Its conservation importance is reflected in its designation as a UNESCO World Heritage Site and its inclusion within the Papahānaumokuākea Marine National Monument.

Baiting methods

During 2016–2017, AGRID$_3$ (Bell Laboratories, Madison, WI), a cholecalciferol rodenticide, was hand-broadcast in all affected areas to reduce mouse populations more effectively and with less disturbance to other wildlife species compared to trapping. AGRID$_3$ pellets contain 0.075% cholecalciferol (non-anticoagulant), which acts by disrupting calcium (Ca) homeostasis through increasing Ca absorption from the small intestine, mobilisation of Ca from the bones into the blood stream, and decreasing Ca excretion by the kidneys (Marshall, 1984). Cholecalciferol has been proven to be toxic and effective at controlling rodents, yet relatively safe to non-target species when used according to label specifications. Due to cholecalciferol's unique mode of action, target specificity, no taste aversion, and delayed toxic effect, it has been successfully used in commensal and agriculture field rodent control situations (Hix, et al., 2012). These attributes make it ideal for use as an interim control measure in the event that eradication is subsequently preferred and approved. The registered use of AGRID$_3$ in the United States has only been for agriculture purposes in the past. The USFWS collaborated with Bell Laboratories, Inc. to develop a supplemental label to be attached to AGRID$_3$ Pelleted Bait (EPA REG. NO. 12455-117-3240). This supplemental label specifically for use by USFWS to control house mice on MANWR was approved by the Environmental Protection Agency for use in a wildland setting.

We hand-broadcast AGRID$_3$ pelleted bait along a 5 m grid (one application within each 25 m^2 square grid cell) over every mouse attack area on Sand Island, as well as a 10 m buffer zone on the periphery of the area, on December 17–18, 2016. Previous experimental bait uptake trials using the protocol described in Pott, et al., (2015), in which we applied placebo bait at 40 kg/ha, marked pellets, and measured pellets taken over a four-day period, led to the selection of 20 kg/ha as an effective application rate under average conditions and 35 kg/ha when mouse density was very high. Following bait application, we surveyed treatment areas to document any sick or injured non-target species or instances of non-target species foraging on bait pellets. We repeated the application at the same rate of 20 kg/ha on 20 January 2017. Over the course of the season from 17 December 2016 to 20 January 2017 we applied 721 kg of AGRID$_3$ to the treatment areas. Areas receiving only a single application included the control plot and impact areas identified after the December application such as Plots 4 and 5. Each application took approximately 440 person-hours to complete.

Non-targets

In order to reduce house mouse predation on incubating albatrosses while minimising the effects (mortality and disturbance) to non-target species, including Laysan ducks and migratory shorebirds, managers treated albatross attack areas where dead adults or abandoned nests were found on Sand Island, MANWR, with AGRID$_3$. AGRID$_3$ was chosen specifically because of its minimal potential effects on non-target species, specifically endangered Laysan ducks (*Anas laysanensis*; listed under the United States Endangered Species Act of 1973) and shorebirds which are protected under the Migratory Bird Treaty Act, particularly bristle-thighed curlews (*Numenius tahitiensis*), Pacific golden plovers (*Pluvialis fulva*), and ruddy turnstones (*Arenaria interpres*). These species were present in large numbers on Sand Island, MANWR, during the mouse attacks and are known to have ingested rodenticide pellets or insects that have consumed bait at other sites where rodent eradication has been implemented. Eason, et al. (2000) documented that mallard ducks fed cholecalciferol at a rate of 2,000 milligrams/kilogram were not affected and concluded that ducks would have to consume 2,000 g (4.4 lbs) of bait with this concentration to receive a lethal dose. Smaller Laysan ducks may consume some bait; however, it is unlikely the ducks would consume enough to cause injury and would need to ingest more than twice their body weight in pellets to experience lethal effects.

Study design and monitoring methods

Starting in December 2016, when most albatrosses had laid their eggs, observers trained to detect mouse-injured albatrosses again searched for, documented, and mapped birds showing signs of mouse attack as well as areas that had an unusually high occurrence of abandoned eggs in nest cups across Sand Island. To avoid double counting, they marked dead adult albatrosses. Nests belonging to injured birds (typically bite wounds, sometimes resulting in severe infection) and abandoned eggs were also marked every three days in the intensive monitoring area (Plot 1). Once the majority of mouse attack areas had been identified, three baiting plots (Plots 1 [16,493 m^2], 2 [15,119 m^2], and 3 [11,740 m^2]) and a control [6,031 m^2] that was not treated with rodenticide were established and monitored for changes in mouse abundance in all plots prior to rodenticide applications on 17 December 2016 and for two weeks afterwards. Two additional baiting plots (Plots 4 [1,900 m^2] and 5 [4,725 m^2]) were added later and monitored for dead adults and abandoned eggs one day before and once at six days and once at 10 days after a second bait application that began on 18 January 2017. The plots were all of different sizes because we chose entire discrete areas in which dead albatrosses and abandoned nests were found to label as attack areas. The control area was smaller than the treatment plots because our priority was to implement a management action as quickly as possible in as much of the colony as possible. General surveillance for signs of mouse attacks continued after hatching in early February and throughout the rest of the chick rearing period.

All nests in Plot 1 were monitored to determine reproductive success, defined as number of nests with an incubating adult present at the beginning of February divided by the total number of nests with eggs present at the start of the study. The reproductive success in Plot 1 was compared to data from plots unaffected by mouse predation that were part of a long-term albatross demography project being conducted at MANWR for the same time interval.

We measured mouse relative abundance in all five plots and the control area two days before rodenticide treatment, one day a week later and one day two weeks after application. We used six multi-catch mouse traps (Trapper 24/7 Bell Laboratories) per treatment area, baited with peanut butter, and summed the number of mice captured over one night for each plot. The traps were centred within the plot ca.10 m from each other. To detect any change in number of mice at each plot, we conducted a one-tailed, paired t-test comparing the mean number of captures prior to bait application with the number of captures two weeks post-treatment ($\alpha = 0.05$). In addition, for Plots 1, 2, 3 and the control area we walked a 150 m transect and counted all mice seen within 2.5 m of the path on either side between 7:30 and 10:00 p.m. the night immediately before the bait application and then one night one week after broadcast and one night two weeks after the broadcast.

We used weather data measured daily at Henderson Airfield weather station located on Sand Island and available from the U.S. National Climate Data Center, <(https://www7.ncdc.noaa.gov/CDO/cdopoemain.cmd?datasetabbv=DS3505&countryabbv=&georegionabbv=&resolution=40>) to evaluate fluctuations in mouse relative abundance over time in the context of rainfall and aid in our interpretation of results.

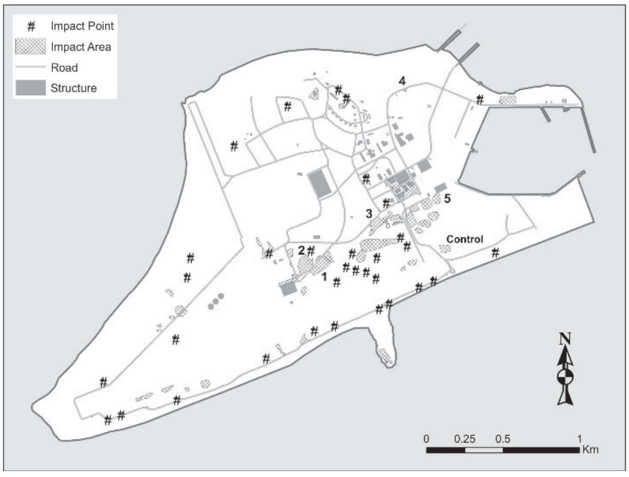

Fig. 2 Areas in which mouse attacks (dead adults, wounded adults, abandoned eggs) were detected in the 2016–2017 albatross breeding season.

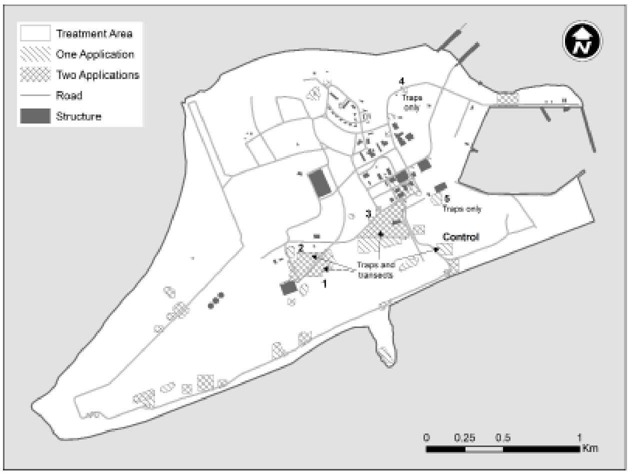

Fig. 3 Areas treated with AGRID$_3$ at a rate of 20 kg/ha December 2016 through January 2017.

RESULTS

Over the course of the 2015–2016 breeding season, mice killed at least 42 adult albatross, wounded an additional 480 birds, and resulted in 70 abandoned nests in three distinct areas, totalling 1.65 ha of Sand Island. During the breeding season of 2016–17 mouse predation was first observed on 4 December, 2016. Numbers of injured and dead adult albatross and abandoned nests increased dramatically in comparison with the previous breeding season. The number of affected areas in the colony increased from three to 50 and the total affected area increased from 1.65 ha to 11 ha (Fig. 2). Albatrosses nest on all of the 460 ha of Sand Island except where they are excluded by active runway paving or structures, so the area affected is still a relatively small proportion of all the albatrosses at Midway. All areas where albatross mortality was detected in 2015–2016 also had mouse predation in 2016–2017. By mid-February there were 242 dead adults, 1218 injured birds, and 994 abandoned nests. This represented a 7-fold increase in mortality, more than double the rate of injury and a more than 10-fold rate of nest abandonment compared to the previous year. The majority of birds found injured and dead were Laysan albatrosses; few black-footed albatrosses were affected. Six carcasses recovered fresh from the area were sent to the USGS Wildlife Health Laboratory in Honolulu in January 2016. Analysis of the specimens revealed that the birds were in excellent body condition with no cause of death evident other than the large wounds on their necks, backs or flanks. Study of the wound sites confirmed the rodent bites occurred before death.

There were no confirmed instances of mouse predation after February 6, 2017, about the time that most eggs started hatching. Most identified mouse attack areas were baited twice before predation stopped in February (Fig. 3). There were no observations of any non-target organism such as shorebirds or Laysan ducks interacting with bait pellets in the field or being found sick or dead in the baited areas.

The number of newly deceased adults and abandoned nests diminished after both bait applications in Plot 1 (Fig. 4) where we were able to conduct more intensive mortality and nest abandonment monitoring every three days. In contrast, during December, the number of abandoned eggs more than doubled in the control area from 10 to 23 but no dead adults were recorded in that area. In January, Plot 4 showed a decrease in the number of abandoned nests after the AGRID$_3$ application but Plot 5 continued to have relatively steady counts of newly dead adults and abandoned nests (Fig. 5). Reproductive success (number of eggs in early February / number of eggs in mid-December)

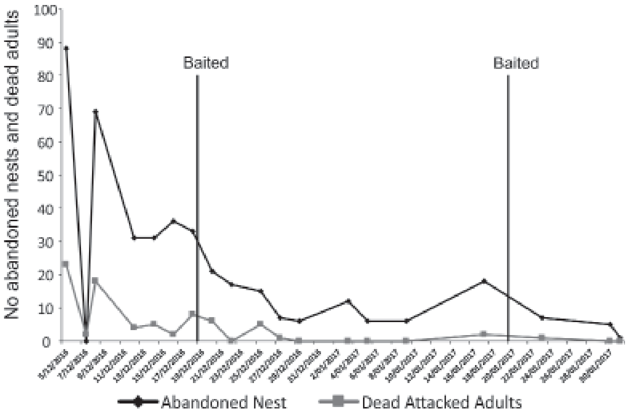

Fig. 4 Absolute counts of new detections of abandoned eggs and dead adults surveyed approximately every three days in Plot 1 throughout the incubation period of breeding Laysan albatrosses at Midway.

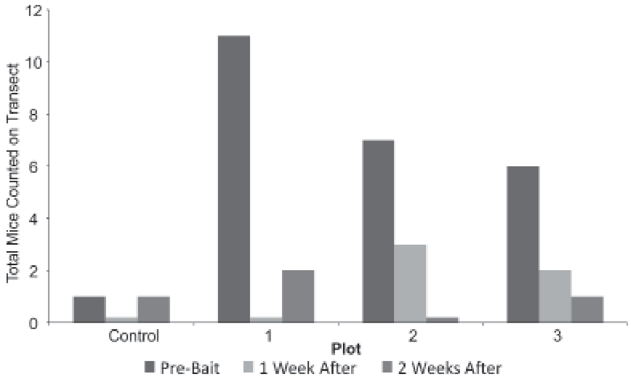

Fig. 7 Total mice counted at night on a 150 m transect (2.5 m to each side) prior to and one and two weeks after application of cholecalciferol bait in Plots 1–3 and a control site on Sand Island during December 2016.

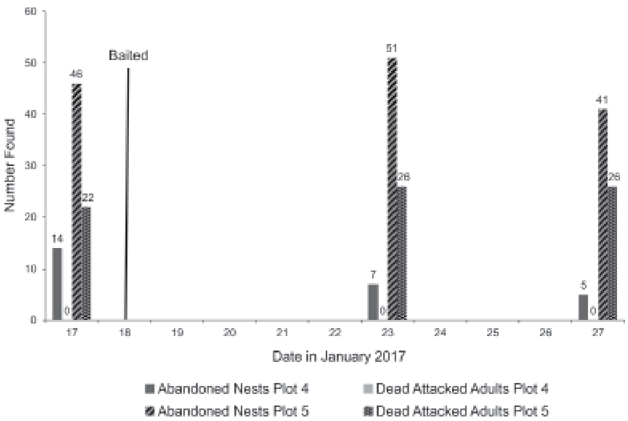

Fig. 5 Count of new detections of abandoned eggs and dead adults in Plots 4 and 5 immediately before and one week and two weeks after baiting.

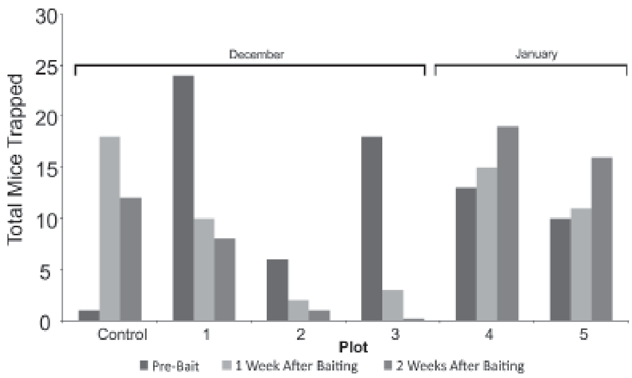

Fig. 6 Number of mice trapped in 6 multi-catch traps per plot, one day before and one and two weeks subsequent to applications of cholecalciferol rodent bait in Plots 1–5, and a control site at Sand Island during December 2016 and January 2017. Only 1 application was done in areas 4 and 5.

in Plot 1 was six percent lower than in the unaffected long-term demography plots.

After the December rodenticide application, the number of mice trapped in Plots 1, 2, and 3 dropped (Fig. 6) (Plot 1 $t(5) = 2.46$ $P = 0.03$; Plot 2 $t(5) = 0.8$ $P = 0.23$; Plot 3 $t(5) = 2.18$ $P = 0.04$). Over the same time period the control site showed an increase in mice trapped ($t(5) = -2.63$ $P = 0.02$). Trapping in Plots 4 and 5, done a month later in January, showed a different pattern with mouse numbers increasing after treatment (Plot 4 $t(5) = -0.99$ $P = 0.18$; Plot 5 $t(5) = -1.66$ $P = 0.08$). Mouse detections on the 150 m transect in Plots 1, 2, and 3 showed a decline after the application of $AGRID_3$ while detections remained much the same in the control plot (Fig. 7).

DISCUSSION

The exposure of a non-negligible proportion of the world's Laysan and black-footed albatrosses to a threat of adult mortality stimulated the management team at MANWR to seek short-term and long-term solutions. The application of a pelleted cholecalciferol rodenticide, $AGRID_3$, in a wildland setting at MANWR, where many non-target species are present, shows promise as a management tool to limit house mouse predation on breeding seabirds without causing harm to the non-target shorebird and duck species that inhabit Sand Island. $AGRID_3$ measurably reduced mouse predation on nesting albatross in the areas where injured and dead albatrosses and abandoned nests were being detected.

While this study was limited in scope and sample size due to the prioritisation of rodent management for the purposes of protecting nesting albatrosses, the larger plots studied during the December application of rodenticide showed decreases in the attacks by mice on albatross as well as some reduction in mouse abundance. The results from the January trial were less promising, showing an increase in mouse density and ambiguous effects on albatross mortality and nest abandonment counts.

There were two differences between the December and January trials that might explain the contrasting outcomes. First, the plots baited in December were much larger in area than the plots baited in January. In a food-limited environment, mice may have been attracted by the bait into the smaller plots elevating the mouse density thus offsetting mortality and mouse population reduction. In an experimental application of cholecalciferol over a much larger area of 100 ha in New Zealand Hix, et al. (2012) observed a 100% reduction of mouse numbers. Second, rainfall increased dramatically over the two months of the study. The increase in rain between December and January might have increased the amount of natural rodent foods within the study area while also leading to higher rates of pellet degradation due to the moister conditions, thus reducing bait availability. There was no control plot established in the January trials so changes in mouse behaviour or abundance cannot be evaluated but Plot 1 continued to show a decrease in mouse attacks throughout the January trial period leading to the possible conclusion that the results in Plots 4 and 5 were due to the smaller plot size. During future efforts to control rodent populations in targeted areas using a broadcast of rodenticide, control areas of at least 1.2 ha should be considered to ensure sufficient coverage to compensate for edge effects.

The decision to apply $AGRID_3$ prior to the albatross breeding season in any particular year may be informed by the likelihood that conditions will trigger mouse predation. Hypotheses about the conditions on Sand Island that may have triggered the emergence of house mouse attacks include population fluctuations of mice and a shift in mouse behaviour due to habitat changes and food availability. Golden crownbeard, *Verbesina encelioides,* an introduced sunflower-relative, was once dominant across the island with coverage now reduced to less than one percent due to control measures ongoing since 2011. We have no evidence that *Verbesina* is consumed by mice, and it is considered a poisonous plant to ungulates (Keeler, et al., 1992) and is allelopathic, thus inhibiting all other vegetation (Inderjit, et al., 2000). *Verbesina* distribution and density was much reduced several years before mice were documented killing albatrosses at Midway. Changes in seasonality of rainfall patterns observed during the 2015–16 and 2016–17 El Niño event may have shifted the timing of normal population fluctuations in the mouse population of Sand Island, in which drying conditions reduce forage and subsequently cause mass-starvation. In 2015–16 and 2016–17 this crash occurred just as albatrosses began the vulnerable incubation period when the adult birds are reluctant to leave their eggs. Rodent populations are well known to fluctuate with rainfall (Jaksic, et al., 1997) and climate change may increase the frequency of El Niño–Southern Oscillation events (Timmermann, et al., 1999), exacerbating the risk to albatrosses in the future. The question of whether there was cultural transmission of albatross predation behaviour in the mice at Sand Island remains open. During 2016–17 the behaviour arose almost simultaneously over much of the island so it seems unlikely.

Preparations for a proposed mouse eradication attempt at Sand Island, MANWR, are underway and the proposed toxicant is brodifacoum. $AGRID_3$, being a cholecalciferol-based rodenticide may be advantageous for control operations prior to a possible eradication to reduce the chance of mice developing aversion or resistance to the type of bait products and toxicants that might be used in an actual eradication operation.

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REFERENCES

Angel, A., Wanless, R.M. and Cooper, J. (2009). 'Review of impacts of the introduced house mouse on islands in the Southern Ocean: Are mice equivalent to rats?' *Biological Invasions* 11: 1743–1754.

Cuthbert, R. and Hilton, G. (2004). 'Introduced house mice *Mus musculus*: A significant predator of threatened and endemic birds on Gough Island, South Atlantic Ocean?' *Biological Conservation* 117: 483–489.

Eason, C.T., Wickstrom, M., Henderson, R., Milne, L. and Arthur, D. (2000). 'Non-target and secondary poisoning risks associated with cholecalciferol'. *New Zealand Plant Protection* 53: 299–304.

Hix, S., Aylett, P. Shapiro, L., MacMorran, D., Eason, C.T. Sam, S., Ross, J.G., Miller, A., Ogilvie, S.C. (2012). 'Low-dose cholecalciferol bait for possum and rodent control.' *New Zealand Journal of Agricultural Research* 55: 207–215.

Inderjit, Asakawa, C. and Dakshini K.M.M. (2000). 'Allelopathic potential of *Verbesina encelioides* root leachate in soil.' *Canadian Journal of Botany* 77(10): 1419–1424.

Jaksic, F.M., Silva, S.I., Meserve, P.L. and Gutierrez, J.R. (1997). 'A long-term study of vertebrate predator responses to an El Niño (ENSO) disturbance in western South America'. *Oikos* 78: 341–354.

Jones, M.G.W. and Ryan, P.G. (2009). 'Evidence of mouse attacks on albatross chicks on sub-Antarctic Marion Island'. *Antarctic Science* 22: 39–42.

Keeler, R.F., Baker, D.C., Panter, K.E. 1992. 'Concentration of galegine in *Verbesina encelioides* and *Galega oficinalis* and the toxic and pathologic effects induced by the plants.' *Journal of Environmental Pathology, Toxicology and Oncology* 11(2): 11–17

Marshall, E.F. (1984). 'Cholecalciferol: A Unique Toxicant for Rodent Control'. In: D.O. Clark (ed.) *Proceedings of the Eleventh Vertebrate Pest Conference*, pp. 95–98. Davis, CA: University of California.

Pott, M., Wegmann A.S. Griffiths, R., Samaniego-Herrera, A., Cuthbert, R., Brooke, M. de L., Pitt, W.C., Berentsen, A.R., Holmes, N.D., Howald, G.R., Ramos-Rendón, K. and Russell, J.C. (2015). 'Improving the odds: Assessing bait availability before rodent eradications to aid in selecting bait application rates.' *Biological Conservation* 185: DOI: 10.1016/j.biocon.2014.09.049.

Starr, F. and Starr, K. 2015. *Botanical Survey of Midway Atoll*. Unpublished Report. Prepared for United States Fish and Wildlife Service. Midway Atoll National Wildlife Refuge.

Timmermann, A., Oberhuber, J., Bacher, A., Esch, M., Latif, M. and Roeckner, E. (1999). 'Increased El Niño frequency in a climate model forced by future greenhouse warming.' *Nature* 398: 694–697.

Eradicating black rats from the Chagos – working towards the whole archipelago

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Abstract The Chagos Archipelago comprises some 58 islands covering 5,000 ha in the centre of the Indian Ocean. Black rats (*Rattus rattus*) were introduced about 230 years ago and have likely had a severe impact on the native terrestrial fauna, which is dominated by seabirds and land crabs. Most of the archipelago's terrestrial land mass is vegetated with old coconut plantations, with over 75% of the native forest cleared for coconut from 26 of the largest islands. Likely as a result of this colonisation and clearance, at least 30 islands have rats present (95.3% of the Chagos landmass) along with feral cats (*Felis catus*) on 62%, which suppresses the recovery of native fauna and flora. Efforts at rat eradication include the failed attempt on Eagle Island (252 ha) in the northern Chagos Archipelago in 2006 and the recent success of a ground-based eradication on Île Vache Marine in 2014, where two applications of brodifacoum poison were hand-spread at a rate of 18 kg/ha. Two islets on the nearby Salomon atoll were also cleared of black rats during the same operation with single bait applications. The 2014 operation was successful on what are regarded as difficult islands for rat eradication, being 'wet' tropical islands with land crabs and coconut plantations present, and has engendered confidence to proceed with additional rat eradications on other northern Chagos islands.

Keywords: atoll, *Birgus*, Chagos, eradication, hand-broadcast, *Rattus*, seabirds, tropical

INTRODUCTION

Invasive species have caused 75% of terrestrial vertebrate extinctions on islands (McCreless, et al., 2016) and of these species' rats are probably the most pervasive, having been introduced to more than 85% of oceanic islands and archipelagos (Harper & Bunbury, 2015). Rats have been responsible for some 40–60% of all bird and reptile extinctions (Howald, et al., 2007). Rats prey upon and compete with animals and can consume all parts of plants, which disrupts ecosystem function and can cause direct or indirect cascades of collapse, through interruption of pollination and nutrient pathways, seed predation, and in some cases leading to forest collapse (Towns, et al., 2006; Athens, 2009; Towns, 2009; Hilton & Cuthbert, 2010).

Black rats (*Rattus rattus*) have been present on the Chagos Archipelago, in the mid-Indian Ocean, since the late 1700s when the archipelago was settled (Wenban-Smith & Carter, 2016). Diego Garcia is in the southern Chagos Archipelago and is the largest (~2,900 ha) and only inhabited island, with a transient population associated with a military base. It has rats and cats (*Felis catus*) present and there are no current plans for rat eradication. In the northern Chagos Archipelago (~2,100 ha total combined area), 26 of the 55 islands are known or suspected to have black rats present (Carr & Harper, 2015). These rat-infested islands comprise some 1,700 ha in combined total land area or some 47% of the islands in the group. Only 4.7% of the entire Chagos terrestrial space is regarded as mammalian predator free and seabird population density is approximately 20 times higher on rat-free islands (Hilton & Cuthbert, 2010). (Fig. 1).

Low-lying, remote and geologically young (49 mMYA, Duncan & Hargraves, 1990), the Chagos Archipelago has not had the speciation that has developed on similarly isolated elevated archipelagos such as Hawaii and the Mascarene Islands. The atolls of the Chagos Archipelago are largely formed from marine sand deposits with some raised rock formations. Many islands had their native forest removed during settlement and replaced with a dense monoculture of coconut palms (*Cocos nucifera*). As several seabird species preferentially nest in native trees, this destruction of nesting habitat was probably the first major impact on the previously large seabird colonies that existed there (Carr, 2013). This was followed by direct persecution by man and other introduced predators; principally this was rats, cats, dogs and pigs. By the start of the 1900s, the vast seabird colonies now indicated by guano deposits had disappeared (Carr, 2011).

Fig. 1 Map of the Chagos Archipelago showing location of islands mentioned in the text.

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On less anthropogenic-impacted islands the architecture of the native oceanic rain forest allows arboreal nesting by lesser noddy (*Anous tenuirostris*) and red-footed booby (*Sula sula*), whereas the open areas are used by species such as brown booby (*Sula leucogaster*), brown noddy (*Anous stolidus*), sooty tern (*Onychoprion fuscatus*) and the tropical shearwater (*Puffinus bailloni*). Two introduced birds, the domestic chicken (*Gallus gallus*), and Madagascan fody (*Foudia madagascariensis*) are the only land birds resident in the northern Chagos, the former found on only a few islands. Fodies are found on most of the vegetated islands. Land crabs are the dominant invertebrates, with the coconut crab (*Birgus latro*) being the most obvious. Smaller hermit crab species, the burrowing land crab (*Cardisoma carniflex*) and other land crab species are present (Stoddart, 1971a; PC pers. obs.). There is a reviving population of green turtles (*Chelonia mydas*) and hawksbill turtles (*Eretmochelys imbricata*) that nest on some islands (Mortimer & Day, 2009). No native mammals, including bats, exist on the islands.

Rat eradication planning

The Chagos Conservation Trust is championing the eradication of rats from the archipelago, to provide an environment for populations of existing native species to recover and to restore the ecosystem to a state prior to rat invasion (<https://chagos-trust.org/about/vision-and-mission>). This endeavour is in concert with The British Indian Ocean Territory Interim Conservation Management Framework of 2014 (<https://biot.gov.io/biot-interim-conservation-management-framework-september-2014.pdf>), which has an overarching vision: "To maintain and, where possible, enhance the biodiversity and ecological integrity of the British Indian Ocean Territory (BIOT)".

All eradication attempts require comprehensive planning before implementation and this was particularly true for a rat eradication programme on a highly isolated tropical island, which presented a novel suite of problems. Invasive mammal eradication work in the Chagos Archipelago faces both logistical and ecological challenges due to the archipelago's remoteness and inaccessibility, along with the wet climate and the vegetation composition with the significant component of coconut 'chaos'. Île Vache Marine was selected for the initial rat eradication because it was: deemed a realistic and manageable size for a start-up operation; the risk of reinvasion was considered negligible due to its distance from other islands; there were no susceptible non-target species; it nestled in amongst five confirmed or proposed IUCN classified Important Bird Areas (Carr, 2011); there was some anecdotal evidence that shearwaters had once bred on the island and, if successful, the probability of re-colonisation by marine avifauna was likely.

Île Vache Marine (12.4 ha, 2 m elevation) is situated on the southern rim of the Peros Banhos atoll (05°25› S, 71°49› E, Fig. 1). It is a typical tropical low-lying oceanic coralline island and the vegetation comprises a shoreline perimeter of *Scaevola taccada* on the exposed southern coast with introduced coconut and the occasional *Guettarda speciosa* and *Morinda citrifolia* on the coast facing the atoll. The mean annual rainfall for the Peros Banhos atoll (data from 1950–1966) is approximately 4,000 mm, distributed bi-modally, with a slightly drier period through the austral winter (Stoddart, 1971b). Île Vache Marine was never inhabited, but was visited until 1974 for coconut harvesting. The plantation workers would have come from Île du Coin (126 ha), some 6 km distant, the former plantation headquarters and likely source of rats. In 2014 there were very limited numbers of seabirds present.

Previous rat eradication attempts in the Chagos

There was an attempt to eradicate black rats from Eagle Island (252 ha) in 2006. A team of 11 established 2,864 bait stations on a 30 m × 30 m grid of cut tracks starting in early February. The bait stations were loaded with Talon™ wax blocks (0.05 g/kg brodifacoum with bitrex) that was maintained in the stations until the team departed in late April (Meier, 2006). Later checks revealed that the operation had failed.

The Île Vache Marine eradication served two purposes. Firstly, it was an opportunity to undertake a rat eradication operation, albeit small, as proof that the method could be successful in the northern Chagos islands and engender confidence in the technique as a management tool for biodiversity gains in the region. Secondly, it added Île Vache Marine to a string of rat-free islands in eastern Peros Banhos, which were situated amongst Important Bird Areas and within an area designated as a Strict Nature Reserve under BIOT Law.

Rat eradications on tropical islands have a higher failure rate than temperate islands for a variety of reasons, including the presence of coconuts, land crabs as bait competitors, and on 'wet' tropical islands, in particular (Russell & Holmes, 2015; Holmes, et al., 2015). Ground-based operations also have had a higher failure rate than aerial bait applications but are usually cheaper to undertake on small islands, with less logistical and technical input required. Hence, for the rat eradications on the northern Chagos a ground-based eradication was planned for cost and logistical reasons but needed to be very cognisant of the risk factors associated with the islands. A successful outcome for the eradication operation in the face of these impediments would promote confidence in the technique for tropical islands with similar characteristics.

As an adjunct to the planned eradications, an additional bait trial was carried out on Diego Garcia in order to measure bait-take by rats at a measured rat population density and refine bait application rates for future rat eradications on Chagos atolls (Harper & Carr, 2015).

METHODS

Île Vache Marine rat eradication, Peros Banhos atoll

Parallel lines were cut at 25 m intervals across the island in June 2014. This was undertaken by volunteers from the British Forces stationed on Diego Garcia. The timing was important, in that it needed to be done long enough before the operation so that any disturbance did not affect rat behaviour but not too early as re-growth was rapid, particularly in *S. taccada* thickets.

August was chosen as the month for bait application as it was one of the driest months of the year (Stoddart, 1971b) and when a vessel was available. The eradication operation staff assembled the equipment and supplies in Diego Garcia on 31 July 2014 prior to departure on 1 August. The team, GH, PC and members of the British Forces on Diego Garcia, landed on Île Vache Marine early on 2 August to allow passage over the coral reef at high tide. The cut lines were checked and, where required, were either re-cut or additional lines slotted in between existing lines. Sites for bait throwing were marked at 25 m along the cut lines and black plastic bait stations (Protecta LP, Bell Labs, USA) laid at these sites for post-broadcast bait deployment. The bait stations were raised 40 mm off the ground with wooden blocks to reduce interference by hermit crabs. By the end of the first day there was a 25 m × 25 m grid of 154 sites across the entire island. The island size was also reconfirmed at 12 ha by walking the coast of the island with a GPS unit (Garmin 62S).

Bait application trials and eradications on other Indian Ocean islands showed that bait could be spread at a rate of >15 kg/ha and be available to all rats for four nights (Merton, et al., 2002, Harper & van Dinther, 2014). Bait was spread on Île Vache Marine on 3–4 August. Pollard pellet bait (Bell Labs 25W) was hand spread at a rate of 18 kg/ha by GH and PC. This involved hand-throwing bait at each of the grid sites. Bait (280 g) was thrown in four directions at right angles to each other, such that it reached about 10–12 m, along with 280 g spread at the throwing point. Bait spreading by the two operators began at each end of the island and lines were traversed such that the operators were converging on each other. Bait coverage was almost completed on the first day, except for a strip of about 2 ha in the centre of the island. This was covered the next morning and a little additional bait was spread above the high tide mark around the coast of the island where hermit crabs were abundant. All the equipment and empty poison bait containers were removed by the end of the morning. The team departed for Diego Garcia shortly thereafter.

A second bait application was undertaken 11 days later. This was to ensure all rats had access to bait, particularly if breeding was occurring and suckling mothers or young animals may have been missed in the first bait application. The island was revisited on 14–15 August and poison bait pellets (Pestoff 20R) were hand laid at a rate of 18 kg/ha. Differing bait types were used for the two applications as ship rats have been observed with distinct preferences for one or other bait, thus circumventing possible bait avoidance (Harper & van Dinther, 2014). Several recently dead rats were located during the second bait application, suggesting rats had readily consumed the poison bait laid in the first application. The bait stations were also then loaded with wax-based poison baits (Ditrac™ 0.005% w/w brodifacoum, Bell Labs) at a rate of three bait blocks (150 g) secured inside each station. This was to ensure that if heavy rain degraded the bait post-departure, or any rats missed the hand-laid bait, then poison bait was still available for several weeks after the operation. The team departed Île Vache Marine on 15 August at midday. It did not rain during either of the bait deployments.

Bird counts on Île Vache Marine had been undertaken by PC since 2009. Counts in 2014 revealed fewer than five pairs of brown noddy and white tern (*Gygis alba*) and ten pairs of great crested terns (*Thalasseus bergii*), were breeding on the island. About 15 pairs of the one introduced passerine, the Madagascar fody (*Foudia madagascariensis*) were present.

In April 2015, the bait stations were removed by PC and a Connect Chagos graduate (a Zoological Society/CCT project with funding from the UK FCO) during a different expedition.

The eradication phase of Îles du Sel and Jacobin, Salomon Islands atoll

Additional poison bait intended as a contingency for the Île Vache Marine operation was deployed on two islets, Îles du Sel (2.2 ha) and Île Jacobin (1.6 ha), in the Salomon atoll, some 25 km east of the Peros Banhos. These two islands were selected for their small size and their relative isolation from other islands. This meant there was a lower probability of re-invasion by rats than other islands in the area and a single application of the remaining bait was deemed practical. On arrival, a quick survey was carried out immediately before each operation to assess the likelihood of success. Both islands were dominated by coconut, with varying amounts of native forest present, with few other factors that would limit the probability of success, as identified by Holmes, et al. (2015). Of note was the lack of large seeds or seedlings of native trees.

Bait was deployed on Île du Sel and Île Jacobin on 16 August 2014. The islands were circumnavigated and waypoints marked at 25 m intervals on each side of the islands using a GPS unit (Garmin 62S). The operators (GH & PC) then walked from the first waypoint on one side to the corresponding waypoint on the opposite side of each island without cutting the vegetation. Pellet bait (Pestoff 20R) was broadcast at 25 m intervals, using the same method as on Île Vache Marine. Bait was spread at a higher rate of 20 kg/ha on Île du Sel and 25 kg/ha on Île Jacobin as it was a single application. The difference in application rate was due to slightly more bait remaining after the first island was treated.

There were opportunities for post-eradication monitoring on Île Vache Marine by PC as part of other expeditions. The first check was made seven months later in April 2015 with a Connect Chagos graduate during daylight, and during an overnight stay, and no sign of rats was seen. A second daytime check was made by PC in February 2016 and again no rat sign was recorded but signs of vegetative recovery were noted (Table 1). An opportunity for both GH and PC to undertake a more comprehensive survey of the island became available over 9–10 April 2017, when 45 rat snap-traps and wax tags were deployed over a 24-hour period. In addition, coconuts were cut open and placed on the ground near the campsite and searches were made for gnawed seeds/coconuts, rat tracks/caches etc. Additional searches were conducted at night by torchlight to detect rats.

During the same expedition, surveys were made at Île du Sel and Île Jacobin on 15 April and 15 rat detection devices (snap traps, wax tags, secured portions of coconut flesh) were deployed overnight on each island. Searches

Table 1 Initial checks of Île Vache Marine for rat sign.

Date	Event	Results	Responsible
24–25/03/2015	Rodent survey including: a. 50 × snap-traps deployed overnight b. Check for rat gnawing on fallen fruit and flowers c. Check for rat excrement d. Daytime visual inspection of island e. Nocturnal inspection of island (overnight stay)	No sign of rat presence	P Carr C Narina J Schlayer
09/02/2016	Rodent survey including: a. Check for rat gnawing on fallen fruit and flowers b. Check for rat excrement c. Daytime visual inspection of island	No sign of rat presence. Obvious signs of native tree seed germination especially *Guittarda speciosa and* extensive flowering of *Scaevola taccada*	P Carr

were made for signs of rats similar to the operation on Île Vache Marine. The islands were revisited the following morning and detection devices recovered and further searches made.

Diego Garcia bait trial

Two 1 ha plots were set out 200 m apart in disused coconut plantation forest some 2 km west of the small township on western Diego Garcia. The plots were divided into a 5 × 5 grid at 25 m intervals. Within the plots an internal trapping grid of 15 Victor snap-traps was established on an interval of 25 m × 12.5 m. The internal grid was centrally located so that there was a 25 m buffer from the plot perimeter.

Poison bait (Pestoff 25R pellets, Animal Control Products, NZ) was hand-spread on both 1 ha plots at a rate of 15 kg/ha on 7 August 2014. The bait had been dyed with Rhodamine-B, which fluoresces under UV light. After one night to allow rats to access the bait the snap-traps were baited with coconut and peanut butter and set. Trapped rats were collected morning and evening for the next three days. The rats were dissected and their gut cavities examined under UV light for evidence that the dyed bait had been consumed.

To give a simple estimate of rat population density, the number of rats caught was divided by the effective trapping area (ETA). To estimate ETA for rats, a boundary strip was added to the edge of the trapping grids (Dice, 1938). The width of the boundary strip was set by adding the average radius (15 m) of a home range of ship rats from mangrove forest on Aldabra Atoll and forest on Juan de Nova and Europa (Harper, et al., 2015, Ringler, et al., 2014).

RESULTS

Rat eradications in northern Chagos Islands

None of the various indicators used to detect rats during the overnight stay on Île Vache Marine on 9–10 April 2017 showed that rats remained on the island. Prior to the eradication rats had been seen on every previous inspection and were easily trapped both diurnally and nocturnally. Moreover, there had been an increase in breeding pairs of seabird species for pre- and post-eradication counts, including a significant increase in numbers of white tern ($T_1 = -2.32$, d.f. = 6, $p = 0.03$), which are vulnerable to rats, and great crested terns ($T_1 = -4.73$, d.f. = 3, $p = 0.009$).

Similarly, none of the indicators for detecting rats on Îles du Sel and Jacobin showed sign of any rats. Many seeds of the large native tree *Intsia bijuga* had germinated and there was a carpet of 300 mm high seedlings on the forest floor of both islands, along with many untouched seeds. These large seeds appear to be a favoured food of rats, as the seeds and seedlings are rarely found on rat-infested islands.

Diego Garcia bait trial

Sixty rats were removed from traps over the three days; 30 from each plot. There was significant interference with, and removal of, trapped rats by land crabs so this is highly likely to be a minimum number of rats trapped. Of the 60 rats, 59 (98.3%) had eaten dyed bait. The one rat that had not consumed bait was an adult female that was trapped in the first morning after the bait application, so bait had been available for a little over 36 hours. Some bait was still present on the last day of trapping.

Of the trapped rats, only two were juveniles (both female) and there was a slight sex bias towards males (32:28). Several adult male rats were in poor condition, whilst some rats were in good condition with substantial amounts of mesenteric fat. Of the 26 adult female rats trapped, two were pregnant.

The trapping grids within the bait grids were 25 m in diameter and adding a 15 m boundary strip gave a total radius of the ETA of 40 m, for an area of 0.5 ha. At least 30 rats were caught on each trapping grid, which translates to a minimum population density of 60 rats/ha.

DISCUSSION

Rats were eradicated on three small ‘wet tropical’ islands in the northern Chagos with two hand-spread application rates of 18 kg/ha each on the larger Île Vache Marine and single applications of 20 and 25 kg/ha respectively, on the smaller islets Îles du Sel and Jacobin.

Of particular interest was the success of the rat eradications on the very small islets, considering that only one bait application, albeit at a higher initial rate but cumulatively less than on Île Vache Marine, was made on each. Best practice suggests two applications, although it is generally acknowledged that the second application acts as an insurance policy against unforeseen confounding factors, such as heavy rain ruining bait (Keitt, et al., 2015), and because rats can breed year-round in the wet tropics (Russell & Holmes, 2015). In this case the small size of the islands, selection of the driest period of year and well planned rapid bait deployment by a small team is likely to have assisted with operational success as the factors associated with eradication failure on tropical islands were reduced (Holmes, et al., 2015).

Of crucial importance were the parallel and well-cut lines cut in the thick vegetation on Île Vache Marine, such that there were no gaps in bait coverage due to converging tracks. The bait application took 1.5 days, at a rate of about 5 ha/person/day. The bait applications began at both ends of the island simultaneously and a gap in bait coverage was left for one night in both cases, which did not affect the operational success. It is not known whether the bait was degraded by any heavy rain in the days immediately after bait deployment as the team left the area shortly after both applications. Although there were several land crab species, including hermit crabs, present on the island, coconut crabs that can outcompete rats for bait were absent.

It can be concluded that rats can be eradicated from small Chagos Archipelago islands with a minimum toxic bait application of 20 kg/ha, and the trials on Diego Garcia indicate that a 15 kg/ha application rate is too low. This suggests that on similar small islands at least, single applications of poison can successfully remove rats and should be considered in appropriate circumstances. A single bait application has advantages in reduced logistics, cost and possible impact on the environment. Where possible, further bait trials will be undertaken on islands in the northern Chagos Archipelago, to gain more confidence with the amount of bait and bait presentation required. These trials are of particular importance on islands largely dominated by coconut crabs, and with burrowing crab species present (Holmes, et al., 2015; GH & PC, pers. obs.) and where mangrove or *Pemphis acidula* is present at periodically flooded sites (Harper, et al., 2015).

This operation has provided evidence that rats can be eradicated from small wet tropical islands that contain large populations of land crabs and coconut forest that has previously been deemed difficult to achieve (Holmes, et al., 2015). We demonstrate that careful assessment and planning prior to the operation can result in a successful outcome (Keitt, et al., 2015). Given the success of ground-based rat eradication operations on the three small islands in the Chagos Archipelago, an eradication is being planned for larger islands in the near future, such as Île Yéyé (61 ha), which is the only remaining rat-infested island in the eastern Peros Banhos Strict Nature Reserve. If this is successful a larger operation to eradicate rats from all of the northern Chagos Archipelago is likely to be pursued.

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REFERENCES

Athens, J.S. (2009). '*Rattus exulans* and the catastrophic disappearance of Hawai'i's native lowland forest'. *Biological Invasions* 11: 1489–1501.

Carr, P. (2011). A guide to the birds of the British Indian Ocean Territory. *Information Press, Oxford, UK.*

Carr, P. (2013). 'Factors Influencing Breeding Island Selection of Red-footed Booby *Sula sula* (Linn. 1766) in the Chagos Archipelago, Central Indian Ocean, and the Implications for Future Island Management Plans'. MSc thesis. UK: Warwick University.

Carr, P. and Harper G.A. (2015). 'The distribution of ship rat *Rattus rattus* in the Chagos Archipelago'. *Chagos News* 47: 20–31.

Dice, L.R. (1938). 'Some census methods for mammals'. *Journal of Wildlife Management* 2: 119–130.

Duncan, R.A. and Hargraves, R.B. (1990). '^{40}AR/^{39}AR Geochronology of Basement Rocks from the Mascarene Plateau, the Chagos Bank, and the Maldives'. In: R. A. Duncan, J. Backman, L.C. Peterson, et al. (eds.) *Proceedings of the Ocean Drilling Program, Scientific Results* Vol. 115, pp. 43–51. Texas: College Station.

Harper, G.A. and van Dinther, M. (2014). *Bait and Trap Trials for Invasive Rats and Cats on Aldabra Atoll and Implications for Their Eradication*. Unpublished report for the Seychelles Islands Foundation, Republic of Seychelles.

Harper, G.A., van Dinther, M. and Bunbury, N. (2015). 'The response of black rats (*Rattus rattus*) to evergreen and seasonally arid habitats: informing eradication planning on a tropical island'. *Biological Conservation* 185: 66–74.

Harper, G.A. and Bunbury, N. (2015). 'Invasive rats on tropical islands: Their population biology and impacts on native species'. *Global Ecology and Conservation* 3: 607–627.

Harper, G.A. and Carr, P. (2015). 'An initial trial to determine an effective rat bait application rate at Diego Garcia, British Indian Ocean Territory'. *Chagos News* 47: 17-19.

Hilton, G.M. and Cuthbert, R.J. (2010). 'The catastrophic impact of invasive mammalian predators on birds of the UK Overseas Territories: A review and synthesis'. *Ibis* 152: 443–458.

Holmes, N. D., Griffiths,R., Pott, M., Alifano, A., Will, D., Wegmann, A.S. and Russell, J.C. (2015). 'Factors associated with rodent eradication failure'. *Biological Conservation* 185: 8–16.

Howald, G., C. J. Donlan, C.J., Galván, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

McCreless, E.E., Huff, D.D. Croll, D.A., Tershy, B.R., Spatz, D.R., Holmes, N.D., Butchart S.H.M. and Wilcox, C. (2016). 'Past and estimated future impact of invasive alien mammals on insular threatened vertebrate populations'. *Nature Communications* 7: 12488.

Meier, G.G. (2006). 'Rat eradication on Eagle Island'. *Chagos News* 28: 2–4.

Merton, D.V., Climo, G., Laboudallon, V., Robert, S. and Mander, C. (2002). 'Alien mammal eradication and quarantine on inhabited islands in the Seychelles'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 182–198. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK .

Mortimer, J.A. and Day. M. (2009). 'Sea Turtle Populations and Habitats in the Chagos Archipelago'. In: C.R.C. Sheppard and M.R.D. Seaward (eds.) *Ecology of the Chagos Archipelago.* Westbury, England: Linnean Society Occasional Publications 2.

Ringler, D., Russell, J., Jaeger, A., Pinet, P., Bastien, M. and Le Corre, M. (2014). 'Invasive rat space use on tropical islands: Implications for bait broadcast'. *Basic Applied Ecology* 15: 179–186.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Stoddart, D.R. (1971a). 'Terrestrial fauna of Diego Garcia and other Chagos atolls'. *Atoll Research Bulletin* 149: 163–170.

Stoddart, D.R. (1971b). 'Rainfall on Indian Ocean coral islands'. *Atoll Research Bulletin* 147: 1–21.

Towns, D.R., Atkinson I.A.E. and C.H. Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?' *Biological Invasions* 8: 863–891.

Towns, D.R. (2009). 'Eradications as reverse invasions: lessons from Pacific rat (*Rattus exulans*) removals on New Zealand islands'. *Biological Invasions* 11: 1719–1733.

Wenban-Smith, N. and Carter, M. (2016). *Chagos: A History; Exploration, Exploitation, Expulsion*. London: Chagos Conservation Trust.

Bridging the research-management gap: using knowledge exchange and stakeholder engagement to aid decision-making in invasive rat management

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Abstract The world is facing a biodiversity crisis. Nowhere is that more apparent than on oceanic islands where invasive species are a major threat for island biodiversity. Rats are one of the most detrimental of these and have been the target of numerous eradication programmes; a well-established conservation tool for island systems. For at-risk native species inhabiting large, populated islands, where rat eradication is not an option, control of rat populations has been conducted but this requires continuous management and therefore its long-term viability (and that of the at-risk native species which the project aims to protect) can be uncertain. Large-scale rat management areas or 'mainland islands' have been successfully developed in New Zealand. However, large-scale management is a long-term investment with huge financial implications and committing to such an investment can be met with reluctance. This reluctance, and its subsequent hindrance to decision-making, can be caused by uncertainty relating to species conservation outcomes, and the multiple objectives of stakeholders. We address the issue of uncertainty and the importance of communication between all stakeholder parties in relation to the Mauritius olive white-eye (*Zosterops chloronothos*), a critically endangered passerine endemic to Mauritius and highly threatened by invasive rats. Specifically, we illustrate how the combination of scientific research and communication, knowledge exchange, and stakeholder workshops, can address some of the barriers of decision-making, helping to bridge the research-management gap, and enable the timely expansion of existing rat management for the benefit of this highly threatened bird.

Keywords: mainland islands, Mauritius, rat control, uncertainty, *Zosterops chloronothos*

INTRODUCTION

The world is facing a biodiversity crisis and nowhere is that more apparent than on oceanic islands where invasive species are a major threat (Jones & Merton, 2012; Rodrigues, et al., 2014). Recent research has identified islands as conservation priority areas for evolutionary distinct and globally endangered (EDGE) species, increasing the importance of conservation for island endemics from areas such as Hawaii, New Zealand, the Mascarenes and the West Indies where there are high extinction rates (Diamond, 1989; Jetz, et al., 2014). A major cause of extinction for island birds has been invasive species and rats are the most detrimental; having reached around 90% of all islands they have been identified as a massive threat to ecosystems (Atkinson, 1985; Towns, et al., 2006; Blackburn, et al., 2014).

The eradication of invasive rats from islands is a well-established conservation tool with 474 successful eradications of *Rattus rattus* and *R.norvegicus* (black rat and brown rat) between 1951 and 2014 (Towns & Broome, 2003; DIISE, 2015). However, for species inhabiting large, populated islands, where eradication is not an option, localised rat control has to be conducted. However, this is not a long-term solution for many species of conservation interest as the areas of control can be too small to create viable populations and rat reinvasion rates can be too high. An alternative are large-scale rat management areas or 'mainland islands' which have been successfully developed in New Zealand (Saunders & Norton, 2001; Butler, et al., 2014). However, large-scale management is a long-term investment with huge financial implications and in a world of limited resources and accountability, committing to such an investment can be met with reluctance (Cullen, et al., 2001; Burns, et al., 2012; McCarthy, 2014; Smith, et al., 2015). This reluctance, caused by uncertainty, could hinder decision-making and result in projects maintaining inadequate small-scale management which does not ensure species survival.

Here we address this issue of outcome uncertainty and the importance of communication between scientists, project managers and stakeholders concerning the Mauritius olive white-eye (*Zosterops chloronothos*), a critically endangered passerine endemic to Mauritius and highly threatened by invasive rats (Maggs, et al., 2015; Birdlife International, 2016). The olive white-eye is part of an ancient Indian Ocean white-eye lineage and is in the top 10% of the EDGE bird species list based on their high level of endemism and evolutionary distinctiveness (Warren, et al., 2006; Jetz, et al., 2014). Research has identified rats (black and brown) as a major limiting factor for olive white-eye, preying on nests and causing an estimated annual population decline of 14%; however, rat management can mitigate this threat and ensure population persistence (Maggs, et al., 2015). Based on these findings, small-scale management has been implemented over remnant olive white-eye breeding territories around the Combo region of the Black River Gorges National Park (BRGNP), Mauritius (Fig. 1; Ferrière, et al., 2016). However, small-scale rat management is not adequate enough to enable olive white-eye population viability in the long-term, highlighting the need for large-scale management in the form of a mainland island (Maggs, 2017).

Here we illustrate how a collaborative approach to conservation management can aid decision-making through communication between scientists, managers, and project stakeholders which can facilitate scaling up small-scale rat control to the implementation of a mainland island. For highly threatened species, such as the olive white-eye, this approach ensures the timely implementation of evidence-based decisions and bridges the gap between research and management.

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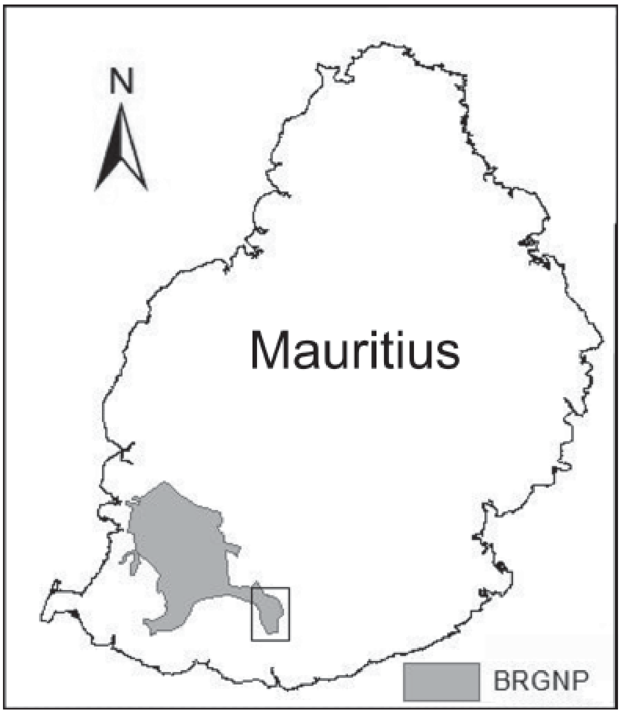

Fig. 1 Mauritius, illustrating the location of the Combo region (black rectangle) within the Black River Gorges National Park (BRGNP).

METHODS

To combat uncertainty, two tools were used; knowledge exchange and stakeholder workshops, in combination with scientific research, to break down some of the barriers to decision-making.

Knowledge exchange

When scaling-up invasive species management there are many logistical and financial considerations. For conservation programmes which have never established such large-scale management, accounting for these considerations and identifying limitations is difficult. Methods and costings of mainland islands have been published (Clapperton & Day, 2001; Gillies, 2002; Gillies, et al., 2006; Scofield, et al., 2011; Burns, et al., 2012; Norbury, et al., 2014; Carter, et al., 2016), but detailed information regularly remains in undocumented individual staff experiences or when data is gathered it remains in inaccessible forms and grey literature. This compounds information inaccessibility resulting in personnel within programmes making decisions based on limited experience rather than evidence (Sutherland, et al., 2004; Pullin & Knight, 2005; Kapos, et al., 2008). Gaining first-hand experience can enable a nuanced understanding of both short and long-term management, which for robust and realistic costing is vital.

To identify the considerations which should be made and gain first-hand information a knowledge exchange was conducted with rat control/eradication experts and conservation managers in the field across New Zealand during April–May 2015. These individuals were identified either through the Hihi Recovery Group, which works closely with numerous mainland island managers, or identifying people through published literature. Using a ‘boundary organisation’ approach (Cook, et al., 2013; Cvitanovic, et al., 2015) scientific researchers facilitated knowledge exchange between experts across New Zealand and project managers in Mauritius. Grey literature and expert knowledge were gathered, identifying potential management techniques and the demands and practicalities involved which aided scientific research.

The sites visited across New Zealand varied in management type and size but all targeted invasive rat species (black and brown). Meetings with the experts and managers were standardised by discussing the same topics, these included:

- Management history. Have other management techniques been previously used, if so, what was the scale of the management and why did it change?
- Identifying mainland island area. What process was used to identify locations, what were the constraints and benefits considered, how were topography and river courses tackled and what was the conservation objective of the mainland island?
- Management technique. What rat management technique is currently used, over what area, and how long has it been in place, is there a buffer zone, how many staff and volunteers work on the site, have additional techniques been trialled and if so what were the outcomes?
- Maintenance. How often are the traps/stations/fence checked or re-baited, how long does this take and how many staff members does this require, what maintenance demands are there, how often does equipment need replacing and how do weather conditions impact the management and work load?
- Management efficiency. Is rat abundance or presence monitored in the management site, if so, what is the rate of rat incursions or rat abundance and is there a response protocol and if so how quickly is this implemented?

Stakeholder workshops

Improving knowledge exchange between decision–makers and scientists is fundamental to support sustainable evidence-based management. However, despite evidence being available, in some cases decisions can still remain hindered due to multiple objectives from a mix of stakeholders with differing priorities, values or conflicting interests (Conroy, et al., 2002). Science can help overcome these obstacles by providing tools to inform decisions and aid stakeholders to make informed trade-offs if required.

An approach termed ‘interdependency’ recognises that all participants in knowledge exchange can contribute, emphasising the need for a two-way exchange between scientists and decision-makers (Contandriopoulos, et al., 2010; Cvitanovic, et al., 2015). This can increase understanding and stakeholder communication through access to the best scientific information, enabling science-based decision-making (Meek, et al., 2015). This process supports collaboration and bridges the research-implementation gap (Knight, et al., 2008), but requires the roles of participants to be outlined from the start to ensure clarity throughout the workshop process; identifying expert advisors, decision-makers or workshop facilitators to mediate between stakeholders.

To ensure collaboration between scientific researchers and decision-makers and avoid conflicting interests, a stakeholder workshop was held in the case of the olive white-eye. During this workshop there were three main objectives to be considered by decision-makers when tackling development from small-scale localised management to a large-scale mainland island: should a mainland island be established, what size it should be to enable population viability and management cost-effectiveness. The workshop was facilitated by scientific researchers, from the Zoological Society of London (ZSL) and University College London (UCL), who provided expert advice on these three objectives; this was accompanied by field staff providing first-hand information on the status of the species and the current management in place from the Mauritian Wildlife Foundation (MWF) (Ferrière, et al., 2016).

Scientific research on the olive white-eye has successfully developed decision-making tools identifying the mainland island area required to ensure population persistence and management cost-effectiveness (Maggs, 2017). These decision-making tools outline scenarios and assist in identifying informed, evidence-based management for the remnant olive white-eye population, ensuring population persistence and clear financial and logistical requirements over 50 years (see Maggs, 2017 for details). Using these tools, discussions were held between the expert advisors (scientific researchers and field staff) and the key decision-makers (project managers, organisation directors, project funders and government officials) where the scientific evidence was discussed, expert opinion shared and questions raised through open dialogue and in a transparent environment.

RESULTS

Knowledge exchange

In total, over four weeks, 30 individuals participated in the knowledge exchange including managers and volunteers from eight mainland island sites and experts from additional conservation companies, central government (Department of Conservation) and specialist groups across New Zealand (Fig. 2). The rat management techniques identified across these sites and discussed included trapping, ground-based poisoning, self-resetting traps and predator-proof fencing. The information gathered through the knowledge exchange was vital for the detailed long-term budgeting of a mainland island in Mauritius

Fig. 2 The distribution of the mainland islands visited during a knowledge exchange in April and May 2015 and the organisations who participated: Hihi Recovery Group, Biodiversity Restoration Specialists, (1) Shakespear Open Sanctuary (Auckland Council), (2) Tawharanui Open Sanctuary (Auckland Council), (3) Sanctuary Mountain Maungatautari, (4) Boundary Stream Mainland Island (Department of Conservation), (5) Rotokare Scenic Reserve Trust, (6) Bushy Park Sanctuary, (7) Zealandia, (8) Rotoiti Nature Recovery Project (Department of Conservation).

under each of the four management techniques, providing detail into the equipment and materials required and labour demands. This first-hand information was combined with existing literature and fed directly into scientific research conducting cost-effectiveness analysis for the four rat management techniques, accounting for the costs over 50 years. By accurately budgeting each management technique over 50 years the long-term cost-effectiveness of the four rat management techniques against olive white-eye population quasi-extinction risk were robustly illustrated; which acts as the effectiveness score of the rat management techniques (Table 1; see Maggs, 2017 for full details).

Stakeholder workshop

Eighteen delegates from six organisations participated in the stakeholder workshop; these included project management (MWF), organisation directors (MWF and Durrell Wildlife Conservation Trust), scientific researchers (ZSL and UCL), project funders (Chester Zoo) and government officials (National Parks and Conservation Service).

The olive white-eye is a priority species for conservation in Mauritius and it was decided, based on the scientific findings presented, that a mainland island should be established, aiming for the minimum area identified by Maggs (2017) of 275 ha; required at a low population density to ensure a 99% chance of population persistence over 50 years. Using the cost-effectiveness analysis conducted by Maggs (2017), and presented at the workshop, the rat management technique decided upon was Goodnature®A24 self-resetting traps based on their cost-effectiveness, specifically, their low labour requirements and competitive financial costs (Maggs, 2017). Although a relatively new technique, their long-term costs, maintenance and replacements, were accounted for based on manufacturer recommendations; the same long-term costs were accounted for all of the techniques discussed.

Trapping was considered too labour intensive even though it was highly cost-effective when considering equipment costs alone. Poison was ruled out based on the potential environmental impacts and the overall high cost of poison and associated labour. Predator-proof fencing was not considered as an option based on the huge initial setup cost and the long-term financial commitment, also the habitat loss associated with installing a predator-proof fence (at least 8m of forest would need to be cleared both sides of the fence to prevent mammals jumping over (Day, 2004); with highly threatened flora species within the BRGNP this cannot be justified at this time). Fencing is the most cost-effective technique when creating a mainland island over vast areas and could maintain zero rat densities, which the other techniques cannot achieve, but complete rat removal is not required for olive white-eye viability, merely reduced rat densities. The techniques combined were not discussed but could be in an additional option to consider in the future.

As well as the rat management technique it was also identified that the mainland island would have to take a 'multi-species/multi-threat' approach, targeting a number of invasive species until the impact of individual species is known in order to avoid the 'surprise factor' of secondary unexpected and undesired results (Alterio, et al., 1999; Saunders & Norton, 2001; Caut, et al., 2009; Carter, et al., 2016). This would involve targeting small Indian mongoose (*Urva auropunctata*), feral cats (*Felis domesticus*) and potentially crab eating macaques (*Macaca fascicularis*) as well as rats. This level of predator control would also benefit other highly threatened endemic species such as the Mauritius cuckoo-shrike (*Coracina typica*), echo parakeet

Table 1 The minimum area required for a mainland island to ensure a 99% chance of population persistence for the Mauritius olive white-eye over 50 years, the total cost of establishing and running a mainland island over 50 years, the establishment costs alone and the average annual costs; comparing trapping, ground based poisoning, self-resetting traps and predator-proof fencing (Maggs, 2017).

	Area (ha)	Total Cost (£ millions)	Establishment Costs (£)	Annual costs (£)
Trapping	275	2.9	186,700	37,908
Poisoning	300	7.9	40,925	157,913
Self-resetting traps	275	3.8	130,315	37,505
Predator-proof fencing	275	5.7	1,766,472	80,196

(*Psittacula eques*) and Mauritius pink pigeon (*Nesoenas mayeri*), which are found within the same regions.

Finally, it was suggested that, if possible, the site of a mainland island should be combined with existing conservation management areas (CMAs), which have been established on mainland Mauritius in the BRGNP to protect native vegetation communities by removing exotic flora (Cheke & Hume, 2008). The control of rats would encourage habitat regeneration and resources could be combined for both invasive fauna and flora control.

DISCUSSION

This case study aimed to illustrate how a collaborative approach to conservation management, through knowledge exchange and stakeholder workshops, can aid communication and decision-making. In this case, it was used to guide the timely expansion of rat management from existing small-scale control (32 ha) to a mainland island (275 ha), relatively quickly and effectively, which is vital for highly threatened and declining species, such as the olive white-eye.

A mainland island has never been established in Mauritius. The rat management techniques used for the olive white-eye have been limited to localised snap-trapping and ground-based poisoning (Maggs, et al., 2015). In the past, feasibility studies have been conducted for various techniques, including predator-proof fencing, but taking the step from localised to landscape scale management was not taken due to resource limitations and long-term financial and logistical uncertainty (Day, 2004).

Here we have tackled the barriers of logistical and financial uncertainty and decision-making through a 'co-production' approach with full cooperation between scientific researchers and decision-makers (Cvitanovic, et al., 2015; van Kerkhoff & Lebel, 2015). Conducting knowledge exchange allowed project managers to gain first-hand information and fill knowledge gaps from leading experts in the field of invasive species management. Incorporating this into a robust analysis of the financial and logistical requirements of a mainland island helped to minimise uncertainty, justify expenditure and identify the long-term financial support required from funders (Maggs, 2017). A stakeholder workshop then allowed scientific research to be fed directly back to all involved, successfully highlighting project priorities and enabling all participants to come to a unified decision on future management goals for the olive white-eye; guiding science-based conservation while maintaining transparency among stakeholders.

Through this collaborative approach, in just three years, long-term management goals have been identified to establish the first mainland island in Mauritius to protect the olive white-eye and ensure long-term population viability. Implementation of a mainland island within the national park has started in the Brise Fer CMA with the introduction of olive white-eye planned for 2021 if rat management can maintain adequately low rat densities over a prolonged period. The area of the mainland island will be increased with growth in capacity, aiming to reach the full mainland island size (275 ha) within 5–10 years. Without these processes, project decisions could have taken years longer to reach the same point if field trials were required (to test all potential rat management techniques), accurate long-term financial requirements were not known, open discussion was not had or scientific research was not fed back to decision-makers; delays which would have detrimental impacts on highly threatened and declining species like the olive white-eye.

The methods discussed here address ways to approach existing challenges, reduce uncertainty and enable evidence-based decision-making. The approaches taken, although case-specific, provide methods for researchers and managers to adopt and apply to different scenarios depending on the decision-making barriers and uncertainty being faced; bridging both the knowledge-action boundary and the research-management divide (Roux, et al., 2006; Cook, et al., 2013), which is rarely achieved in conservation.

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REFERENCES

Alterio, N., Moller, H. and Brown, K. (1999). 'Trappability and densities of stoats (*Mustela erminea*) and ship rats (*Rattus rattus*) in a South Island *Nothofagus* forest, New Zealand'. *New Zealand Journal of Ecology* 23: 95–100.

Atkinson, I.A.E. (1985). 'The Spread of Commensal Species of *Rattus* to Oceanic Islands and their Effects on Island Avifaunas'. In: P.J. Moors (ed.) *Conservation of Island Birds*, pp. 35–81. ICBP Technical Publication 3. Cambridge, UK: International Council for Bird Preservation (ICBP).

Birdlife International (2016). *The IUCN Red List of Threatened Species 2016*. <www.iucnrelist.org/details/22714005/0>.

Blackburn, T.M., Essl, F., Evans, T., Hulme, P.E., Jeschke, J.M., Kühn, I., Kumschick, S., Marková, Z., Mrugała, A., Nentwig, W., Pergl, J., Pyšek, P., Rabitsch, W., Ricciardi, A., Richardson, D.M., Sendek, A., Vilà, M., Wilson, J.R.U., Winter, M., Genovesi, P. and Bacher, S. (2014). 'A unified classification of alien species based on the magnitude of their environmental impacts'. *PLOS Biology* 12: 1–11.

Burns, B., Innes, J. and Day, T. (2012). 'The Use and Potential of Pest-proof Fencing for Ecosystem Restoration and Fauna Conservation in New Zealand'. In: M. Somers and M. Hayward (eds.) *Fencing for Conservation: Restriction of Evolutionary Potential or a Riposte to Threatening Processes?* pp. 1–320. Springer Science+Business Media.

Butler, D., Lindsay, T. and Hunt, J. (2014). *Paradise Saved*. Auckland, NZ: Random House New Zealand.

Carter, A., Barr, S., Bond, C., Paske, G., Peters, D. and van Dam, R. (2016). 'Controlling sympatric pest mammal populations in New Zealand with self-resetting, toxicant-free traps: a promising tool for invasive species management'. *Biological Invasions* 18: 1723–1736.

Caut, S., Angulo, E. and Courchamp, F. (2009). 'Avoiding surprise effects on Surprise Island: Alien species control in a multitrophic level perspective'. *Biological Invasions* 11: 1689–1703.

Cheke, A. and Hume, J. (2008). '*Lost Land of the Dodo*'. United Kingdom: T & AD Poyser.

Clapperton, B.K. and Day, T.D. (2001). 'Cost-effectiveness of exclusion fencing for stoat and other pest control compared with conventional control'. *DOC Science Internal Series* 14: 5–19.

Conroy, D.E., Willow, J.P. and Metzler, J.N. (2002). 'Multidimensional fear of failure measurement: The performance failure appraisal inventory'. *Journal of Applied Sport Psychology* 14: 76–90.

Contandriopoulos, D., Lemire, M., Denis, J.-L. and Tremblay, E. (2010). 'Knowledge exchange processes in organizations and policy arenas: A narrative systematic review of the literature'. *The Milbank Quarterly* 8: 444–483.

Cook, C.N., Mascia, M.B., Schwartz, M.W., Possingham, H.P. and Fuller, R.A. (2013). 'Achieving conservation science that bridges the knowledge-action boundary'. *Conservation Biology* 27: 669–678.

Cullen, R., Fairburn, G.A. and Hughey, K.F.D. (2001). 'Measuring the productivity of threatened-species programs'. *Ecology Economics* 39: 53–66.

Cvitanovic, C., Hobday, A.J., van Kerkhoff, L., Wilson, S.K., Dobbs, K. and Marshall, N.A. (2015). 'Improving knowledge exchange among scientists and decision-makers to facilitate the adaptive governance of marine resources: A review of knowledge and research needs'. *Ocean and Coastal Management* 112: 25–35.

Day, T. (2004). *Feasibility of pest proof fencing in Mauritius*. Cambridge, New Zealand: Xcluder™ Pest Proof Fencing Company.

Diamond, J. (1989). 'Overview of Recent Extinctions'. In: D. Western and M. Pearl (eds.) *Conservation for the Twenty-First Century*, pp. 37–41. New York: Oxford University Press.

DIISE. (2015). The Database of Island Invasive Species Eradications, developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand. <http://diise.islandconservation.org/>.

Ferrière, C., Page, J., Zuël, N., Tatayah, V. and Jones, C. (2016). *Mauritius Olive White-eye Recovery Programme Annual Report 2015–16*. Vacoas, Mauritius.

Gillies, C. (2002). 'Managing Rodents on the New Zealand Mainland – What Options are Currently Available?'. *DOC Scientific Internal Series* 47: 5–20.

Gillies, C., Styche, A., Bradfield, P., Chalmers, K., Leach, M., Murphy, E., Ward-Smith, T. and Warne, R. (2006). 'Diphacinone Bait for Ground Control of Rats on Mainland Conservation Land'. *Science for Conservation* 270: 5–20.

Jetz, W., Thomas, G.H., Joy, J.B., Redding, D.W., Hartmann, K. and Mooers, A.O. (2014). 'Global distribution and conservation of evolutionary distinctness in birds'. *Current Biology* 24: 919–930.

Jones, C.G. and Merton, D.V. (2012). 'A Tale of Two Islands: The Rescue and Recovery of Endemic Birds in New Zealand and Mauritius'. In: E.G. Ewen, D.P. Armstrong, K.A. Parker and P.J. Seddon (eds.) *Reintroduction Biology: Integrating Science and Management*, pp. 33–72. Wiley-Blackwell.

Kapos, V., Balmford, A., Aveling, R., Bubb, P., Carey, P., Entwistle, A., Hopkins, J., Mulliken, T., Safford, R., Stattersfield, A., Walpole, M. and Manica, A. (2008). 'Calibrating conservation: new tools for measuring success'. *Conservation Letters* 1: 155–164.

Knight, A.T., Cowling, R.M., Rouget, M., Balmford, A., Lombard, A.T. and Campbell, B.M. (2008). 'Knowing but not doing: Selecting priority conservation areas and the research-implementation gap'. *Conservation Biology* 22: 610–617.

Maggs, G., Nicoll, M., Zuël, N., White, P.J.C., Winfield, E., Poongavanan, S., Tatayah, V., Jones, C.G. and Norris, K. (2015). '*Rattus* management is essential for population persistence in a critically endangered passerine: Combining small-scale field experiments and population modelling'. *Biological Conservation* 191: 274–281.

Maggs, G. (2017). 'The Ecology and Conservation of Wild and Reintroduced Populations of the Critically Endangered Mauritius Olive White-eye (*Zosterops chloronothos*). PhD thesis: London, UK: University College London.

McCarthy, M.A. (2014). 'Contending with uncertainty in conservation management decisions'. *Annals of the New York Academy of Sciences* 1322: 77–91.

Meek, M.H., Wells, C., Tomalty, K.M., Ashander, J., Cole, E.M., Gille, D.A., Putman, B.J., Rose, J.P., Savoca, M.S., Yamane, L., Hull, J.M., Rogers, D.L., Rosenblum, E.B., Shogren, J.F., Swaisgood, R.R. and May, B. (2015). 'Fear of failure in conservation: The problem and potential solutions to aid conservation of extremely small populations'. *Biological Conservation* 184: 209–217.

Norbury, G., Hutcheon, A., Reardon, J. and Daigneault, A. (2014). 'Pest fencing or pest trapping: A bio-economic analysis of cost-effectiveness'. *Austral Ecology* 39: 795–807.

Pullin, A.S. and Knight, T.M. (2005). 'Assessing conservation management's evidence base: A survey of management-plan compilers in the United Kingdom and Australia'. *Conservation Biology* 19: 1989–1996.

Rodrigues, A.S.L., Brooks, T.M., Butchart, S.H.M., Chanson, J., Cox, N., Hoffmann, M. and Stuart, S.N. (2014). 'Spatially explicit trends in the global conservation status of vertebrates'. *PLOS ONE* 9: 1–18.

Roux, D.J., Rogers, K.H., Biggs, H.C., Ashton, P.J. and Sergeant, A. (2006). 'Bridging the science-management divide: Moving from unidirectional knowledge transfer to knowledge interfacing and sharing'. *Ecology and Society* 11(1): 4.

Saunders, A. and Norton, D.A. (2001). 'Ecological restoration at mainland islands in New Zealand'. *Biological Conservation* 99:109–119.

Scofield, R.P., Cullen, R. and Wang, M. (2011). 'Are predator-proof fences the answer to New Zealand's terrestrial faunal biodiversity crisis?' *New Zealand Journal of Ecology* 35: 312–317.

Smith, M., Wallace, K., Lewis, L. and Wagner, C. (2015). 'A structured elicitation method to identify key direct risk factors for the management of natural resources'. *Heliyon* 1: e00043.

Sutherland, W.J., Pullin, A.S., Dolman, P.M. and Knight, T.M. (2004). 'The need for evidence-based conservation'. *Trends in Ecology and Evolution* 19: 305–308.

Towns, D.R. and Broome, K.G. (2003). 'From small Maria to massive Campbell: Forty years of rat eradications from New Zealand islands'. *New Zealand Journal of Zoology* 30: 377–398.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?'. *Biological Invasions* 8: 863–891.

van Kerkhoff, L.E. and Lebel, L. (2015). 'Coproductive capacities: Rethinking science-governance relations in a diverse world'. *Ecology and Society* 20: 14.

Warren, B.H., Bermingham, E., Prys-Jones, R.P. and Thébaud, C. (2006). 'Immigration, species radiation and extinction in a highly diverse songbird lineage: White-eyes on Indian Ocean islands'. *Molecular Ecology* 15: 3769–3786.

Timing aerial baiting for rodent eradications on cool temperate islands: mice on Marion Island

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Abstract Aerial baiting from helicopters with a bait-sowing bucket and GPS to ensure coverage with anticoagulant toxins in cereal-based baits can reliably eradicate rodents on islands. Current best practice for temperate islands is to bait in winter when the rodents are not breeding, rodent numbers are lowest so competition for toxic baits is lowest, natural food is likely to be scarce, and many non-target species are absent from the island. However, short winter day lengths at high latitudes restrict the time helicopters can fly and poor weather in winter may increase risks of failure. This paper notes precedents from cool temperate islands where baiting was not conducted in winter and then uses the extensive data on mice on Marion Island to explore whether current recommendations for winter baiting based on breeding and natural food availability are important risk factors in determining time of year to bait. Marion Island mice do not breed between early May and late September, mouse densities reach a maximum in May and minimum in November, but the biomass of main natural food (invertebrates) does not fluctuate greatly over the year. This means the per capita food availability is least in autumn and increases through winter to most in spring and summer. The weight of the stomach contents of mice is also highest in winter. Based on this per capita food parameter, mice are likely to be most hungry between about March and May suggesting baiting would be more effective in this period (perhaps towards the end of it when breeding stops) than in the more traditional winter season.

Keywords: food availability, rodent abundance, seasonality

INTRODUCTION

Successful attempts to eradicate one or more (up to four) species of rodent by sowing toxic baits from an aircraft have been made on at least 166 islands in 13 countries (DIISE, 2018; J. Parkes, unpubl. data) since the first use of this method in 1985 against Norway rats (*Rattus norvegicus*) on Whale Island (143 ha) in New Zealand (Imber, et al., 2000). Most of these islands are at latitudes with temperate climates (n = 96) biased by the large sample from New Zealand, or in tropical latitudes (n = 64) biased by those in the Montebello Group of islands in Western Australia. Few islands are at latitudes with cold climates similar to Marion Island (n = 6). Aerial baiting is currently the only practical option to eradicate rodents on large or topographically difficult islands and has a high success rate when modern best practice is followed (Parkes, et al., 2011; Parkes, 2016). The cost of operational failure is high, especially for large, remote islands, both in the money invested (Holmes, et al., 2016) and if failures discourage risk-averse funders from attempting further projects. Therefore, careful planning and application of best practice based on precedence and analysis of the particular constraints and risks for each project is essential.

Pest eradications achieved by reduction of the target population to zero by a sequence of removal events (e.g. by shooting, trapping or by deployment and re-baiting of bait stations) provide information (e.g. catch per unit effort, kill locations, trends in rates of bait-take across seasons and years) as the population is reduced (e.g. Thomas & Taylor, 2002). Under this strategy, the 'start rules' are not critical as managers can (and should) adapt actions as information accrues during the project, e.g. to allow a change in tactics to account for animals that might avoid one control method (Parkes, et al., 2010). In these projects knowing when to stop and declare success is the more critical issue – at least in terms of efficiency and risk management (Ramsey, et al., 2011).

In contrast, the use of aerial baiting provides little information on likely success or failure from the control itself, other than bait coverage if GPS technology is used. Under this strategy everything has to 'go right on the day' and 'start rules' with meticulous planning are critical (Cromarty, et al., 2002; Springer, 2016). One key 'start rule' is to identify the optimal time of year (or at least avoid sub-optimal times) to conduct the baiting. Broome, et al. (2014) suggest that winter to early spring is the preferred time of year to aerially bait rodents on New Zealand's temperate islands because it is supposed that (a) rodents are often not breeding and so young individuals that might not be exposed to bait because of possible lack of dependence or subordinate behaviours are at lowest levels of abundance, (b) rodent densities are likely to be lowest and so competition for baits least, (c) natural foods are likely to be least abundant, the rodents most hungry and so 100% are likely to eat the baits, and (d) some potential non-target animals such as seabirds are not present in this season. Most rodent eradication projects have followed this advice by baiting in winter for temperate islands. However, these factors are not always mutually independent (least food and fewest rodents), and other factors (weather or logistics) may constrain decisions. The parameter around food availability we are really seeking is the time of year when there is least per capita food, which may or may not be when there is least food or fewest rodents and may or may not be what drives any breeding season. Managers are probably wise to stick with precedence and bait in winter or early spring (or during dry seasons in the tropics) in the absence of any data on the seasonality of food, rodent dynamics or breeding seasonality.

However, for a variety of reasons a few rodent eradications on cool temperate islands using aerial baiting have been conducted in the summer. Mice (*Mus musculus*) were eradicated on Enderby Island (710 ha at 50ºS) in January 1993 because that was when the primary target species, the rabbits (*Oryctolagus cuniculus*), were not breeding (Torr, 2002). Norway rats and house mice were eradicated on the subantarctic island of South Georgia (103,000 ha and 4,900 ha, respectively, at 54ºS) between late February and late May (mostly in March–April) in phases between 2011, 2013 and 2015 because weather conditions and persistent snow cover made a winter operation impossible (Anon., 2016; Martin & Richardson, 2017). Timing and other operational details of aerial baiting on several islands in the French Southern Territories appear to have been determined by the timing of the supply ship, the *Marion Dufrense*. Rabbits and ship rats (*Rattus rattus*), but not mice were eradicated from Saint Paul Island (900 ha at 38ºS) in January–February (Micol & Jouventin, 2002). Attempts to eradicate rodents from some of the islands in the Golfe du Morbihan in the Kerguelen group (49ºS) have

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been made during summer months when the supply ship visits the region. Ship rats and mice were eradicated from Île Château (220 ha) (Anon., 2006) and ship rats but not mice from Île Australia (2337 ha). Attempts to eradicate mice on Île Stoll (60 ha) and ship rats and mice on Île Moules (500 ha) failed (Anon., 2006; DIISE, 2018).

So, maybe we are unnecessarily constraining ourselves to times of year with the worst weather and shortest days on islands at high latitudes by baiting in winter. This paper explores this seasonality question by describing the process used to inform decision-makers of a proposed eradication of mice on Marion Island, a place where the long history of research by South African scientists has provided most of the information to answer the question.

RESULTS AND DISCUSSION

Marion Island

Marion Island (29,000 ha) and the adjacent Prince Edward Island are South Africa's only offshore islands. They lie on the sub-Antarctic convergence at 46º54′S in the south Indian Ocean. Apart from a meteorological station on Marion Island, the islands are uninhabited. Marion is an active volcano rising to 1,230 m a.s.l. (Fig. 1). The climate is cool, wet and temperate with only a few degrees seasonal variation between coldest and warmest months (mean annual temperature is 6.4ºC and mean annual precipitation is about 200 cm). The physical and biotic characters of Marion Island are described in detail in Chown & Froneman (2008) and the impacts and history of the introduced flora and fauna by Angel & Cooper (2011) and Greve, et al. (2017).

Mice were introduced accidentally some 200 years ago, probably with sealers, and are having a significant impact on the native biota (Angel & Cooper, 2011; Dilley, et al., 2016) such that the South African government is considering whether they might be eradicated (Parkes, 2016). Cats (*Felis catus*) were introduced in 1948 in an attempt to control mice at the meteorological station but soon spread as feral animals over the island, killing mice as primary prey and an estimated 450,000 seabirds per year (Dilley, et al., 2017). The cats were eradicated between 1977 and 1991 (Bester, et al., 2002).

Breeding season

Mice can breed all year if high quality food is available, e.g. during beech (*Nothofagus* spp.) mast events in New Zealand winters (Ruscoe, et al., 2005). However, mice

Fig. 1 Vegetated lava (foreground) and swamp habitat (middle background), Marion Island (Photo by John Parkes, April 2016).

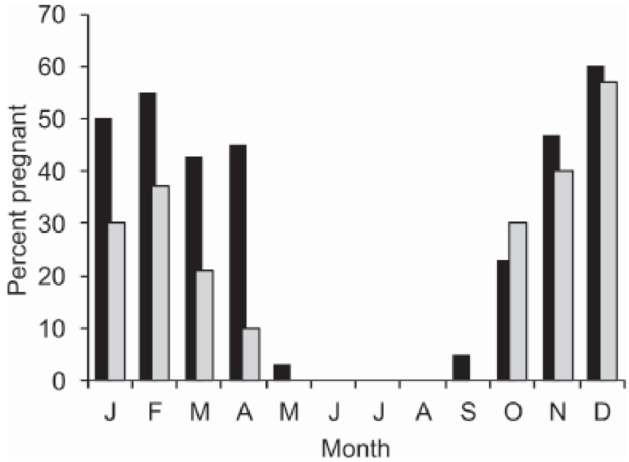

Fig. 2 Monthly pregnancy rates of adult mice, Marion Island in 1991/92 (after Matthewson, et al. (1994) black bars) and 1992/93 (after Avenant & Smith (2004) grey bars).

have a distinct breeding season on Marion Island with no pregnant animals present between early May and late September (Fig. 2). However, this is not a universal rule on all cool temperate islands as 16% of mice sampled during August/September 2012 on Steeple Jason Island (51ºS in the Falkland Islands) were pregnant (Rexer-Huber, et al., 2013).

Density of mice and competition for bait

This breeding season is reflected in the monthly abundance of mice on Marion Island with increasing numbers from the start of breeding in late spring and declining numbers once breeding ends in late autumn (Fig. 3), resulting in lowest densities (at the favoured habitats) at the start of the breeding season (43/ha) in spring and highest (242/ha) in early winter before the decline (Avenant & Smith, 2004).

Baiting during low rodent densities is recommended by Broome, et al. (2014) in part to ensure there are plenty of baits such that all mice, irrespective of their social status, have access to baits. Bait sowing rates in high-density rodent populations of 8 kg/ha in an initial sowing followed by a second sowing of 6 kg/ha about eight days later would result in 7,000 baits/ha – or even in the highest density mouse habitats of Marion Island of 23 baits per mouse. This seems more than adequate to overcome any potential between-mouse competition given each bait contains a lethal dose.

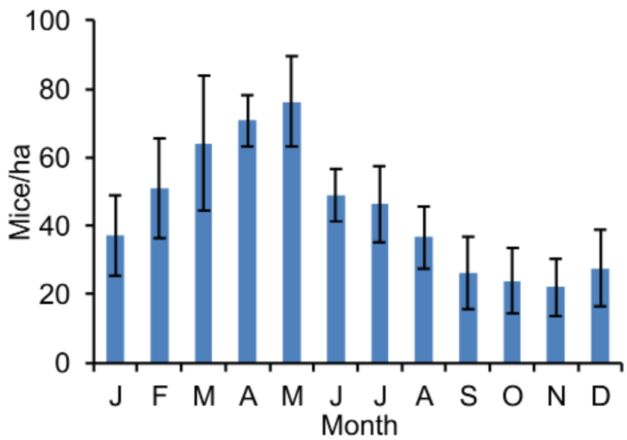

Fig. 3 Seasonal abundance of mice (minimum number known to be alive/ha) averaged across three main habitat types, Marion Island (after Ferreira, et al., 2006).

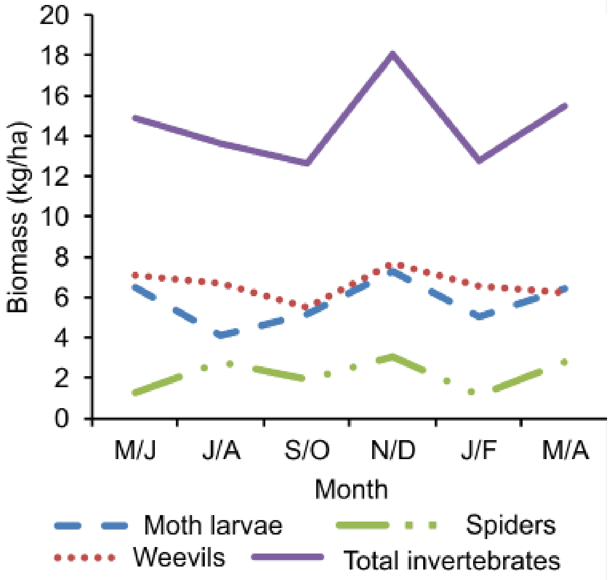

Fig. 4 Seasonal biomass of main invertebrate prey of mice (after Gleeson & van Rensburg (1982)). Total invertebrate biomass (top solid line), weevils (dotted line), moth larvae (dashed line), spiders (lower broken line).

Seasonal variation in per capita natural food

The decades of detailed research conducted on Marion Island (Chown & Froneman (2008) have included studies on the seasonal diet of mice and on the seasonal biomass of their prey. Invertebrates form the bulk of mice diet (depending on habitat) with the larvae and adults of the flightless keystone moth (*Pringleophaga marioni*) (between 13 and 64% by volume) and weevils (*Ectemnorhinus* spp.) (between 11 and 32% by volume) being the most important, followed by earthworms (*Microscolex kerguelarum*) (between 1 and 9% by volume). Plant material, mostly grass and sedge seeds was important, between 16 and 48% by volume (Smith, et al., 2002).

There appears to be only small seasonal variation in the abundance of the main invertebrate fauna favoured by mice (Fig. 4) and Avenant & Smith (2004) found no significant summer–winter differences in invertebrate biomass in the habitat most favoured by mice – apart from spiders which were actually more abundant in winter. The preferred prey for mice, larvae, pupae and adults of *Pringleophaga marioni*, has a long-life cycle of between two and five years (Haupt, et al., 2014) so the absence of seasonal fluctuations is not unexpected given also the small seasonal differences in climate on Marion Island (le Roux & McGeoch, 2008).

The absence of strong seasonal changes in invertebrate prey abundance mean that there is least food per mouse when mouse density is at a maximum, i.e. between March and May, and most food per mouse over winter and spring. For example, the per capita food availability is an order of magnitude lower in early winter when mice are at maximum densities than in early summer when they are at lowest densities. The weight of stomach contents of mice also increases during winter to reflect this (Fig. 5), and mice begin to scavenge or prey upon other mice in autumn and winter (Smith, et al., 2002).

Seasonal absence of non-target species

Most cool temperate islands have a mix of permanent resident bird and seasonally present nesting or moulting seabirds. Unacceptable risks to the former from toxic baiting and secondary poisoning have to be mitigated, e.g. by holding individuals in safe captivity (Rexer-Huber & Parker, 2011), but risks to the latter have to be accepted (e.g. as on Macquarie Island; Parkes, 2016; Springer, 2016) or avoided by baiting when the birds are least common on the island. Marion Island has only two terrestrial birds at risk – the kelp gull (*Larus dominicanus*) and lesser sheathbill (*Chionis minor*) while among the 26 nesting seabird species only three (sub-Antarctic skua (*Catharacta antarctica*), southern giant petrel (*Macronectes giganteus*) and northern giant petrel (*M. hallii*) are at low to modest risk if the baiting was conducted in mid-winter (Parkes, 2016; Springer, 2016).

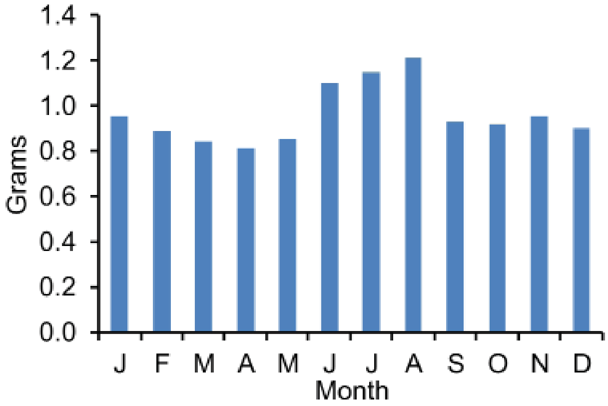

Fig. 5 Monthly changes in the weight of stomach contents adjusted for body weights of Marion Island mice (after Matthewson, et al. (1994) reported in Smith, et al. (2002)).

CONCLUSIONS

Optimal timing of aerial baiting on Marion Island depends on whether the non-breeding season is more or less important than the period with minimum per capita food availability for the mice. Neither hypothesis has been tested. If the latter is most important then a March–May baiting is indicated, but if the former then a May–September baiting is indicated – May at least being a month of overlap. Of course, an earlier timing in late summer is better than a later one in winter, when short days, snow and gales limit flying time.

It is not clear whether the lack of large changes in seasonal abundance and biomass of invertebrates seen on Marion Island is normal for all cool temperate islands. Most studies on other islands lack the year-round data on changes in invertebrate biomass available for Marion Island. However, mice on other cool temperate islands also show a lack of strong seasonality in the occurrence of invertebrates (the bulk of their diet) in their diet, e.g. on Macquarie Island (Copson, 1986) and Île Guillou (Le Roux et al., 2002). This suggests the multi-year life cycles of the invertebrate species on Marion Island may also apply on similar islands and the per capita food supply depends on seasonal changes in mouse density rather than on food abundance. Therefore, mice are likely to be hungriest when they are at maximum densities and not during the winter when they are likely to be least hungry and perhaps less likely to eat artificial food such as baits. An aerial baiting project between March and May is indicated on this condition. Of course, the other considerations mooted by Broome, et al. (2014) might constrain such a choice, as might weather, day length, logistics of ship and helicopter availability as with other projects noted in the introduction.

However, there are several caveats. First, the comparisons between mouse and food abundance are derived across several studies over several decades. This may not be a problem except that the whole ecosystem around mice on Marion Island is highly dynamic. Second, the biomass of invertebrates has collapsed by about 90% since the mid-1970s (Table 1 in Parkes, 2016 and references therein), despite which mouse densities have increased (between 1990 and 2002; Ferreira, et al. (2006) and well after cats were eradicated; the climate is warming (le Roux & McGeoch, 2008); and mice are switching their primary prey from moths to weevils and earthworms (Chown & Smith, 1993) and learning to eat albatross chicks (Dilley, et al., 2016).

Finally, if natural food availability is a problem limiting bait acceptance by rodents (i.e. the proportion of a population that eat the bait) as suspected for some recent failures on tropical islands (Parkes, et al., 2011; Keitt, et al., 2014), and such food competition cannot be predicted or avoided, then one solution is to increase the palatability of the bait relative to natural foods by adding lures or attractants.

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REFERENCES

Angel, A. and Cooper, J. (2011). *A review of the Impacts of the House Mouse* Mus musculus *on sub-Antarctic Marion Island, Prince Edward Islands.* Report to the Prince Edward Islands Management Committee, South African National Antarctic Programme. Core Initiatives.

Anon. (2006). *Status of Measures Implemented by France in the French Southern and Antarctic lands (TAAF) Relating to the Conservation of Petrels and Albatrosses. Agreement on the Conservation of Albatrosses and Petrels.* Second meeting of advisory committee, Brasilia, Brazil 5–8 June 2006.

Anon. (2016). *The South Georgia Habitat Restoration Project: Final Report*. Dundee, Scotland: South Georgia Heritage Trust.

Avenant, N.L. and Smith, V.R. (2004). 'Seasonal changes in age class structure and reproductive status of house mice on Marion island (sub-Antarctic)'. *Polar Biology 27*: 99–111.

Bester, M.N., Bloomer, J.P., van Aarde, R.J., Erasmus, B.H., van Rensburg, P.J.J., Skinner, J.D., Howell, P.G. and Naude, T.W. (2002). 'A review of the successful eradication of feral cats from sun-Antarctic Marion Island, southern Indian ocean'. *South African Journal of Wildlife Research* 32: 65–73.

Broome, K., Cox, A., Golding, C., Cromarty, P., Bell, P. and McClelland P. (2014). *Rat Eradication Using Aerial Baiting. Current Agreed Best Practice used in New Zealand (Version 3.0).* Wellington, New Zealand: New Zealand Department of Conservation.

Chown, S.L. and Smith, V.R. (1993). 'Climate change and the short-term impact of feral house mice at the sub-Antarctic Prince Edward Islands'. *Oecologia* 96: 508–516.

Chown, S.L. and Froneman, P.W. (eds.) (2008). *The Prince Edward Islands. Land-sea Interactions in a Changing Ecosystem.* Stellenbosch, South Africa: Sun Press.

Copson, G.R. (1986). 'The diet of the introduced rodents *Mus musculus* L. and *Rattus rattus* L. on subantarctic Macquarie Island'. *Australian Wildlife Research* 13: 441–445.

Cromarty, P.L., Broome, K.G., Cox, A., Empson, R.A., Hutchinson, W.M. and McFadden I. (2002). 'Eradication planning for invasive alien animal species on islands – the approach developed by the New Zealand Department of Conservation'. In: C.R. Veitch and M.N. Clout (eds.). *Turning the tide: the eradication of invasive species,* pp. 85–91. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

DIISE (2018). 'The Database of Island Invasive Species Eradications, developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group'. University of Auckland and Landcare Research New Zealand. <http://diise.islandconservation.org>.

Dilley, B.J., Schoombie, S., Schoombie, J., and Ryan, P.G. (2016). '"Scalping" of albatross fledglings by introduced mice spreads rapidly at Marion Island'. *Antarctic Science* 28: 73–80.

Dilley, B.J., Schramm, M. and Ryan, P.G. (2017). 'Modest increases in densities of burrow-nesting petrels following the removal of cats (*Felis catus*) from Marion Island'. *Polar Biology* 40: 625–637.

Ferreira, S.M., van Aarde, R.J. and Wassenaar, T.D. (2006). 'Demographic responses of house mice to density and temperature on sub-Antarctic Marion Island'. *Polar Biology* 30: 83–94.

Gleeson, J.P. and van Rensburg, P.J.J. (1982). 'Feeding ecology of the house mouse *Mus musculus* on Marion Island'. *South African Journal of Antarctic Research 12*: 34–39.

Greve, M., Mathakutha, R., Steyn, C. and Chown, S.L. (2017). 'Terrestrial invasions on sub-Antarctic Marion and Prince Edward Islands'. *Bothalia* 47(2): a2143.https://doi.org/10.4102/abc.v4712.2143.

Haupt, T.M., Crafford, J.E. and Chown, S.L. (2014). 'Solving the puzzle of *Pringleophaga* – threatened, keystone detritivores in the sub-Antarctic'. *Insect Conservation and Diversity* 7: 308–313.

Holmes, N.D., Campbell, K.J., Keitt, B.S., Griffiths, R., Beek, J., Donlan, C.J. and Broome, K. (2016). 'Correction: reporting costs for invasive vertebrate eradications'. *Biological Invasions* 18: 2801–2807.

Imber, M., Harrison, M. and Harrison, J. (2000). 'Interactions between petrels, rats and rabbits on Whale Island, and effects of rat and rabbit eradication'. *New Zealand Journal of Ecology* 24: 153–160.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2014). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

Le Roux, V., Chapuis, J-L., Frenot, Y. and Vernon, P. (2002). 'Diet of the house mouse (*Mus musculus*) on Guillou Island, Kerguelen archipelago, Subantarctic'. *Polar Biology* 25: 49–57.

le Roux, P.C. and McGeoch, M.A. (2008). 'Changes in climate extremes, variability and signature on sub-Antarctic Marion Island'. *Climate Change* 86*:* 309–329.

Martin, A.R., and Richardson, M.G. (2017). 'Rodent eradication scaled up: clearing rats and mice from South Georgia.' *Oryx*: 1-9. doi.org/10.1017/S003060531700028X

Matthewson, D.C., van Aarde, R.J. and Skinner, J.D. (1994). 'Population biology of house mice (*Mus musculus* L.) on sub-Antarctic Marion Island'. *South African Journal of Zoology* 29: 99–106.

Micol, T. and Jouventin, P. (2002). 'Eradication of rats and rabbits from Saint-Paul Island, French Southern Territories'. In: C.R. Veitch and M.N. Clout (eds.). *Turning the tide: the eradication of invasive species*, pp. 199–205. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Parkes, J.P., Ramsey, D.S.L., Macdonald, N., Walker, K., McKnight, S., Cohen, B.S. and Morrison, S.A. (2010). 'Rapid eradication of feral pigs (*Sus scrofa*) from Santa Cruz Island, California'. *Biological Conservation* 143: 634–641.

Parkes, J., Fisher, P. and Forrester, G. (2011). 'Diagnosing the cause of failure to eradicate introduced rodents on islands: brodifacoum versus diphacinone and method of bait delivery'. *Conservation Evidence* 8: 100–106.

Parkes, J. (2016). 'Eradication of House Mice *Mus musculus* from Marion Island: A Review of Feasibility, Constraints and Risks'. In: R.M. Wanless (ed.) *Birdlife South Africa Occasional Report Series 1*. Johannesburg, South Africa: Birdlife South Africa.

Ramsey, D.S.L., Parkes, J.P., Will, D., Hanson, C.C. and Campbell, K.J. (2011). 'Quantifying the success of feral cat eradication, San Nicolas Island, California'. *New Zealand Journal of Ecology* 35: 163–173.

Rexer-Huber, K. and Parker, G. (2011). *Captive Husbandry of the Gough Island Bunting and Moorhen*. RSPB Research Report No. 42. Sandy, UK: Royal Society for the Protection of Birds.

Rexer-Huber, K., Parker, G.C., Reeves, M., Stanworth, A.J. and Cuthbert, R.J. (2013). 'Winter ecology of house mice and the prospects for their eradication from Steeple Jason (Falkland Islands)'. *Polar Biology*36: 1791–1797.

Ruscoe, W.A., Elkington, J.S., Choquenot, D. and Allen, R.B. (2005). 'Predation of beech seed by mice: effects of numerical and functional responses'. *Journal of Animal Ecology* 74: 1005–1019.

SGHT (2016). *The South Georgia Habitat Restoration Project: Final Report*. Dundee, UK: South Georgia Heritage Trust.

Smith, V.R., Avenant, N.K. and Chown, S.L. (2002). 'The diet and impact of house mice on a sub-Antarctic island'. *Polar Biology* 25: 703–715.

Springer, K. (2016). 'Methodology and challenges of a complex multi-species eradication in the sub-Antarctic and immediate effects of invasive species removal'. *New Zealand Journal of Ecology* 40: 273–278.

Thomas, B.W. and Taylor, R.H. (2002). 'A History of Ground-based Rodent Eradication Techniques Developed in New Zealand, 1959–1993'. In: C.R. Veitch and M.N. Clout (eds.). *Turning the tide: the eradication of invasive species*, pp. 301–310. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Torr, N. (2002). 'Eradication of Rabbits and Mice from Subantarctic Enderby and Rose Islands'. In: C.R. Veitch and M.N Clout (eds.). *Turning the tide: the eradication of invasive species*, pp. 319–328. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

South Africa works towards eradicating introduced house mice from sub-Antarctic Marion Island: the largest island yet attempted for mice

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Abstract House mice (*Mus musculus*) were introduced to South Africa's sub-Antarctic Marion Island, the larger of the two Prince Edward Islands, by sealers in the early 19th century. Over the last two centuries they have greatly reduced the abundance of native invertebrates. Domestic cats (*Felis catus*) taken to the island in 1948 to control mice at the South African weather station soon turned feral, killing large numbers of breeding seabirds. An eradication programme finally removed cats from the island by 1991, in what is still the largest island area cleared of cats at 290 km^2. Removal of the cats, coupled with the warmer and drier climate on the island over the last half century, has seen increasing densities of mice accumulating each summer. As resources run out in late summer, the mice seek alternative food sources. Marion is home to globally important seabird populations and since the early 2000s mice have resorted to attacking seabird chicks. Since 2015 c. 5% of summer-breeding albatross fledglings have been killed each year, as well as some winter-breeding petrel and albatross chicks. As a Special Nature Reserve, the Prince Edward Islands are afforded the highest degree of protection under South African environmental legislation. A recent feasibility plan suggests that mice can be eradicated using aerial baiting. The South African Department of Environmental Affairs is planning to mount an eradication attempt in the winter of 2021, following a partnership with the Royal Society for the Protection of Birds to eradicate mice on Gough Island in the winter of 2020. The eradication programme on Marion Island will be spearheaded by the South African Working for Water programme – Africa's biggest conservation programme focusing on the control of invasive species –which is already driving eradication projects against nine other invasive species on Marion Island.

Keywords: albatross, climate change, eradication, *Felis catus*, invasive species, *Mus musculus*, petrel, predation

INTRODUCTION

In the late 18th and early 19th century humans travelled far and wide in the southern oceans to exploit marine wildlife (Trathan & Reid, 2009). An unfortunate consequence of this travel was the deliberate or incidental introduction of alien animal and plant species to distant environments, causing extensive changes in biological communities (Mooney & Cleland, 2001). The effects of invasive species on biodiversity have been described as "immense, insidious and usually irreversible" (IUCN, 2000). Island ecosystems are highly susceptible to change and introduced species are the main cause of species extinctions on islands (Manne, et al., 1999; Chapin, et al., 2000).

Many seabirds nest on isolated islands that lack land mammals and consequently one of the major threats to oceanic seabird species is the introduction of mammalian predators such as rats (*Rattus* spp.), domestic cats (*Felis catus*) and house mice (*Mus musculus*) onto their breeding islands (Croxall, et al., 2012). The devastating impact of rats on seabird populations breeding on oceanic islands has been well documented (Atkinson, 1985; Jones, et al., 2008). However, mice have been introduced to even more oceanic islands than have rats and, although their impacts on sub-Antarctic island biota are legion (Angel, et al., 2009), until recently they were considered to have little impact on seabird populations (Wanless, et al., 2007; Jones, et al., 2008).

Sub-Antarctic Marion Island (290 km^2, 46°54´S, 37°45´E) is the larger of the two South African Prince Edward Islands which lie c.2,300 km south-east of Cape Town in the south-western Indian Ocean (Fig. 1). As a Special Nature Reserve, established in 1995, the Prince Edward Islands are afforded the highest degree of protection under South African environmental legislation (de Villiers & Cooper, 2008). They also have been a Wetland of International Importance in terms of the Ramsar Convention since 2007 (de Villiers, et al., 2011) and are surrounded by a large (180,000 km^2) Marine Protected Area, declared in 2013, that reaches in places to the edges of South Africa's Exclusive Economic Zone (Lombard, et al., 2007; Nel & Omardien, 2008).A revised management plan adopted in 2014 guides and controls activities at the island group, including biosecurity protocols to avoid alien introductions (DST-NRF Centre of Excellence for Invasion Biology, 2014).

The Prince Edward Islands currently support breeding populations of 29 species of birds, all but two of which probably breed on Marion Island (Ryan & Bester, 2008; Peter Ryan, FitzPatrick Institute unpubl. data; Table 1). Eight bird species of the order Procellariiformes that breed on Marion are listed by the International Agreement on the Conservation of Albatrosses and Petrels, to which South Africa is a founding signatory (Cooper, et al., 2006).These four albatross and four petrel (*Macronectes* and *Procellaria*)

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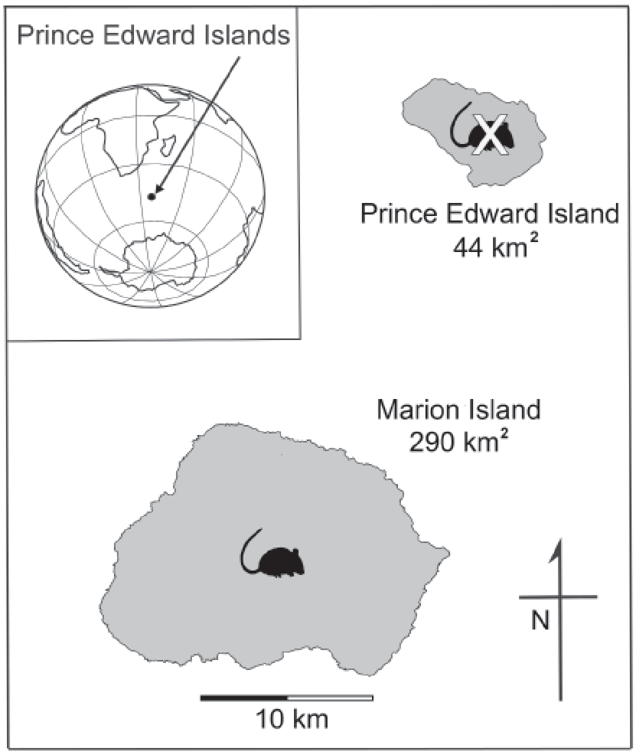

Fig. 1 South Africa's Prince Edward Islands (46°54´S, 37°45´E) lie 2,300 km south-east of Cape Town in the south-western Indian Ocean. Marion Island has introduced house mice, but Prince Edward Island, 22 km to the north-east, remains free of introduced mammals.

species are at risk at sea to bycatch by commercial fisheries, especially longlining, and are considered threatened or near threatened at regional (Taylor, et al., 2015) and global (BirdLife International, 2017) levels. Marion Island supports about 25% of the world's breeding population of wandering albatrosses *Diomedea exulans* (globally and regionally Vulnerable), 12% of sooty (*Phoebetria fusca*) and 7% of grey-headed (*Thalassarche chrysostoma*) albatrosses (both globally and regionally Endangered) and smaller percentages of light-mantled albatrosses (*P. palpebrata*) (globally and regionally Near Threatened) and grey petrels (*Procellaria cinerea*) (globally Near Threatened and regionally Vulnerable).

We show that these five ACAP-listed species, along with the regionally Near Threatened great-winged petrel (*Pterodroma macroptera*), are at serious risk to predation from introduced mice on Marion Island. Mice were accidentally introduced to Marion Island during the sealing era sometime before 1818 and were the sole introduced mammal until 1948 when five domestic cats were introduced to control mice at the newly-established weather station (Watkins & Cooper, 1986). However, even in the 1950s, little was known about the potential harmful effects of invasive species on islands. Rand (1954) was the biologist on the Eighth South African Expedition to Marion Island over 1951/52 and noted how "a few domestic cats have gone feral and prey on the smaller petrels or mice that are widespread over the coastal plain" (p. 178) and "mice often burrow into the [albatross] nest cone but do no appreciable damage" (p. 189). Unfortunately, the cats preferred to eat the island's native birds, especially the burrow-nesting petrels, and by the 1970s more than 2,000 cats were killing some 450,000 birds each year (van Aarde, 1980). As a result, at least one species, the common diving petrel (*Pelecanoides urinatrix*), disappeared from the island and all the other burrowing petrels became far less common than at nearby predator-free Prince Edward Island. A sustained eradication programme that commenced in the mid-1970s had finally eradicated cats from the island by 1991 (Bester, et al., 2002), in what until recently was the largest island area cleared of cats.

We give an overview of the adverse impacts of mice on Marion Island's biota and ecosystem and discuss the mouse eradication attempt planned for the austral winter of 2020.

A syndrome of adverse factors

House mice have been present on Marion Island for nearly 200 years (Berry, et al., 1978), significantly disrupting terrestrial ecosystem functioning (Chown & Smith, 1993). The mice may be seen as part of a syndrome of interacting factors (Parkes, 2016) having adverse impacts on native invertebrates, plants and seabirds (e.g. Phiri, et al., 2009; Angel & Cooper, 2011). The mice have changed the state of Marion Island's ecosystems compared with the near-pristine condition of neighbouring Prince Edward Island (45 km^2, see Fig. 1).

For more than 30 years the burrowing petrel populations on Marion Island were impacted by cats (top predators) and mice (mesopredators). Whereas mice target eggs and chicks (Fugler, et al., 1987; Dilley, et al., 2015; Dilley, et al., 2018), reducing reproductive success, cat predation was far more detrimental because they killed chicks and adults, affecting both reproduction and adult survival (Le Corre, 2008). Removal of the top predator on Marion Island has benefited adult survival but may have triggered a 'mesopredator release' effect (Zavaleta, et al., 2001; Le Corre, 2008), whereby mouse numbers expanded, increasing their impact on petrel populations (Rayner, et al., 2007). The dramatic decrease in burrowing petrel populations at Marion Island caused by the cats is again presumed to have adversely affected key ecological processes driven by burrowing petrels such as soil turn-over and marine nutrient imports (Caut, et al., 2012).

Mouse numbers cycle seasonally on Marion Island, linked partly to changes in the abundance of invertebrates and seeds. Mouse densities are highest in autumn, when breeding ceases, and are lowest in early summer, before breeding resumes. Invertebrate biomass also changes seasonally, but not to the same extent, so that the *per capita* food supply (from macro invertebrates as the primary food of the mice) was estimated to be 3.4 kg/ha and 3.6 kg/ha in early summer but only 0.4 kg/ha and 0.2 kg/ha in early winter in the Biotic and Mire habitats favoured by mice, respectively (Avenant & Smith, 2003).

Peak mouse densities occur in April–May, and have increased between 1990 and 2008, driven in part by a warmer, drier climate (Ferreira, et al., 2006; le Roux & McGeoch, 2008; McClelland, et al., 2018). By comparison, invertebrate biomass has decreased >80% since the late 1970s (McClelland, et al., 2018). Since 2015, there has been a marked increase in the frequency of mice attacking surface-breeding seabird chicks (Dilley, et al., 2016a) and if invertebrate biomass continues to decline, the impact of mouse predation on Marion's seabird chicks is likely to become even more serious.

Overview of mice attacking seabirds at Marion Island

The first signs of mouse attacks on seabirds at Marion Island were recorded in 2003, when wandering albatross chicks were observed with rump wounds typical of those inflicted by mice on Tristan albatross (*D. dabbenena*) chicks on Gough Island (Jones & Ryan, 2010; Table 2). The first attacks on summer-breeding albatross chicks were

Table 1 Estimated risk of local extirpation of bird species currently known or thought to breed on Marion Island if the mice are not eradicated.

Species	Estimated numbers of breeding pairs	Known or considered vulnerable to predation	Estimated years to local extirpation
Grey-backed storm petrel *Garrodia nereis**	? [1]	yes	possibly locally extirpated
Black-bellied storm petrel *Fregetta tropica**	? [1]	yes	possibly locally extirpated
Grey petrel *Procellaria cinerea*	800[2]	yes	30
Cape petrel *Daption capense*	<5[2]	yes	30
Kerguelen petrel *Lugensa brevirostris*	5,000[2]	yes	50
South Georgian diving petrel *Pelecanoides georgicus*	1,000[1]	yes	50
Common diving petrel *Pelecanoides urinatrix*	50–100[2]	yes	50
Great-winged petrel *Pterodroma macroptera*	14,000[2]	yes	50–100
Light-mantled albatross *Phoebetria palpebrata*	300[3]	yes	50–100
Sooty albatross *Phoebetria fusca*	1,465[3]	yes	50–100
Grey-headed albatross *Thalassarche chrysostoma*	7,900[1]	yes	50–100
Wandering albatross *Diomedea exulans*	1,800[1]	yes	50–100
Fairy prion *Pachyptila turtur*	1,000[1]	yes	50–100
Salvin's prion *Pachyptila salvini*	150,000[2]	yes	50–100
Blue petrel *Halobaena caerulea*	145,000[4]	yes	50–100
Soft-plumaged petrel *Pterodroma mollis*	5,000[1]	yes	50–100
White-chinned petrel *Procellaria aequinoctialis*	24,000[5]	yes	50–100
Antarctic tern *Sterna vittata*	25[1]	yes	50–100
Kerguelen tern *Sterna virgata*	50[1]	yes	50–100
Southern giant petrel *Macronectes giganteus*	1,750[1]	uncertain	
Northern giant petrel *Macronectes halli*	400[1]	uncertain	
Crozet shag *Leucocarbo melanogenis*	270[1]	uncertain	
Brown skua *Catharacta antarctica*	300[6]	uncertain	
Kelp gull *Larus dominicanus*	100[1]	uncertain	
Lesser sheathbill *Chionis minor*	700[1]	uncertain	
King penguin *Aptenodytes patagonicus*	220,000[1]	no	
Gentoo penguin *Pygoscelis papua*	900[1]	no	
Macaroni penguin *Eudyptes chrysolophus*	370,000[1]	no	
Southern rockhopper penguin *Eudyptes chrysocome*	67,000[1]	no	

*Current breeding not proven but suspected
Data sources: [1]Ryan & Bester (2008); [2]FitzPatrick unpubl. data; [3]Schoombie et al., (2016); [4]Dilley et al., (2017); [5]Ryan et al., (2012); [6]Ryan et al., (2009)

recorded in April 2009 when sooty albatross fledglings at two colonies were found 'scalped' with raw, bleeding crowns and necks (Jones & Ryan, 2010; Fig. 2). Mice were suspected of being responsible for these wounds (Jones & Ryan, 2010), even though summer-breeding albatross chicks are seldom attacked by mice on Gough Island (Cuthbert, et al., 2013). Another sooty albatross fledgling was attacked in 2010 at one of the colonies where scalpings occurred in 2009 (Ben Dilley, FitzPatrick Institute unpubl. data), but no further attacks were recorded until 2015, when mice attacked large chicks of all three albatross species that fledge in autumn: grey-headed (Fig. 3), sooty and light-mantled albatrosses (Dilley, et al., 2016a, Table 2). Filming at night confirmed that mice were responsible for these wounds, with most affected chicks dying within a few days of being attacked (Dilley, et al., 2016a). Attacks started independently in small pockets all around the island's 70 km coastline, separated by distances hundreds of times greater than mouse home ranges (Wanless, et al., 2008; Dilley, et al., 2016a; Fig. 2). In 2015, three of the six mouse-injured wandering albatross chicks had head wounds ('scalpings', see Fig. 4). In 2016, 2017 and 2018 mouse attacks continued on summer-breeding albatross fledglings, indicating that the sudden increase in 2015 was not a one-off event.

With cats having been eradicated from Marion Island by 1991, we expected burrowing petrel populations to

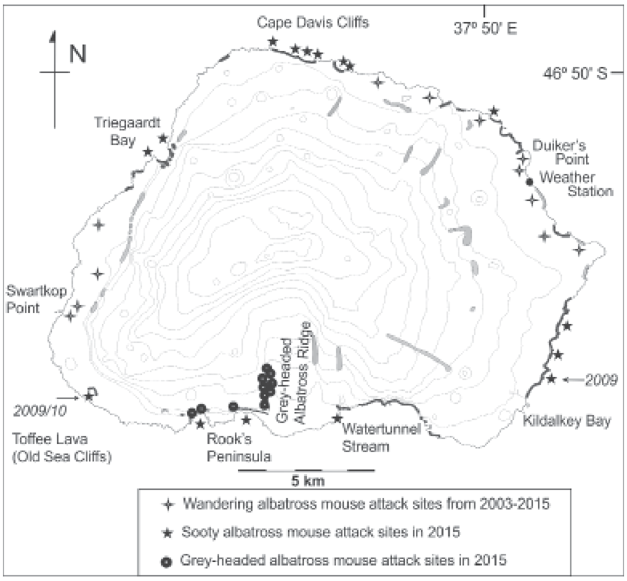

Fig. 2 Marion Island showing the locations of albatross breeding colonies (sooty albatross = dark grey shade; light-mantled albatross = light grey shade; all grey-headed albatross colonies are along Grey-headed Albatross Ridge and Rook's Peninsula) and mouse-attack sites from 2009–2015 (adapted from Dilley, et al., 2016a). Contour lines indicate 100 m.

Fig. 3 Grey-headed albatross chicks showing distinctive 'scalping' wounds inflicted by mice on Marion Island in 2015 (photo Ben Dilley).

have recovered by two decades later. Initial indications were positive; following the removal of cats there were marked increases in the breeding success of burrowing petrels, especially great-winged petrels that breed in winter when cat predation pressure was most severe (Cooper & Fourie, 1991; Cooper, et al., 1995). However, the post-cat recovery of burrowing petrel numbers on Marion has been much slower than anticipated, especially for smaller species (Dilley, et al., 2016b). Of the nine species of burrowing petrels breeding on Marion Island, the two smallest species (black-bellied (*Fregetta tropica*) and grey-backed storm petrels (*Garrodia nereis*)) are now very uncommon and are likely locally extirpated on the island due to mice (Dilley, et al., 2016b). Recent evidence from a repeat survey of burrow densities (Dilley, et al., 2016b) and from analyses of brown skua *Catharacta antarctica* prey remains (Cerfonteyn & Ryan, 2016) both suggest there has been little or no recovery of burrowing petrel populations at Marion since cats were eradicated.

Predation by mice is the most likely explanation for the limited recovery of Marion's petrel populations (Dilley, et al., 2016b). Recent evidence from breeding success studies shows that mice are suppressing the recovery of burrowing petrel populations, especially those that breed in winter, through predation on eggs and chicks (Dilley, et al., 2018). Winter breeders had lower breeding success than did summer breeders, with most fatalities being of small chicks <14 days old. Mice were filmed attacking and killing chicks of two winter-breeding species: grey petrel (three of 18 nests filmed; <https://youtu.be/Og1d6a2cmXQ>) and great-winged petrel (one of 19; <https://youtu.be/D9vPoFsjvgs>, Dilley, et al., 2018). Grey petrel chicks, which had the highest mortality rate, hatch in early winter when mouse densities are still fairly high, but food availability is low, resulting in the lowest seasonal *per capita* food availability for mice (Dilley, et al., 2018). Most grey petrel mortalities occurred when chicks were <7 days old, and were likely due to mouse predation (Dilley, et al.,2018).

We conclude that mice are currently suppressing the recovery of burrowing petrel populations on Marion Island, especially those that breed in winter, through predation on eggs and chicks. The widespread increase in mouse predation on albatross chicks at Marion in 2015 is also a cause for concern. Left uncontrolled, it is feared that 18 of the 28 species breeding on Marion Island may be vulnerable to local extirpation (see Table 1), should the mice not be eradicated.

PLAN OF ACTION

The Prince Edward Islands are recognised as a Special Nature Reserve, which affords the highest degree of protection under South African environmental legislation, and the islands' management plan aims to eradicate alien plants and animals as far as possible (DST-NRF Centre of Excellence for Invasion Biology, 2014). As summarised above, the structure of Marion Island's terrestrial ecosystem has been radically transformed by introduced mice, which are now threatening the island's globally important seabird

Table 2 Summary of mouse attacks on surface-nesting seabirds breeding on Marion Island (from Dilley, et al., 2016a and FitzPatrick Institute unpubl. data).

Species	Year of first attack	Maximum number attacked	% of annual production
Wandering albatross	2003	6	0.8%
Sooty albatross	2009	45	4.3%
Light-mantled albatross	2015	1	4.0%
Grey-headed albatross	2015	102	4.6%

Fig. 4 A Wandering albatross chick being scalped by a mouse on Marion Island in the winter of 2015 (photo Stefan Schoombie).

populations. Given the island's importance as a breeding site for threatened albatrosses and other seabird species that are being killed by mice, there is an urgent need to eradicate mice from the island. A detailed feasibility plan (Parkes, 2016) suggests that mice can be eradicated using aerial baiting. This follows the now well-established approach of using helicopters fitted with GPS guidance systems and under slung bait-distribution buckets to spread brodifacoum-laced pellets across the entire island over a relatively short period, to ensure that all rodents have access to the poison bait. Such operations, pioneered on New Zealand's offshore islands, have a good track record in recent years with 21 of 22 operations around the world targeting mice being successful in the last decade (DIISE, 2015). However, the operation on Marion Island will be an order of magnitude larger than any previous island eradication targeting mice only (cf. Springer, 2016; Martin & Richardson, 2017). This will require the deployment of poison bait with a high level of accuracy given the small home ranges of mice relative to rats (Parkes, 2016).

The South African Department of Environmental Affairs is planning to mount an eradication attempt on Marion Island in the austral winter of 2021. This is timed to follow a planned eradication of mice on Gough Island led by the United Kingdom's Royal Society for the Protection of Birds in the winter of 2020. Gough Island, part of the UK Overseas Territory of St Helena, Ascension and Tristan da Cunha, is one of the world's most important seabird breeding islands. It is the site where mice were first appreciated to pose a significant risk to breeding seabirds (Cuthbert & Hilton 2004; Wanless, et al., 2007), and experiences very high levels of chick mortality in several species, including the Tristan albatross (globally Critically Endangered), Atlantic petrel (*Pterodroma incerta*) (Endangered) and Macgillivray's prion (*Pachyptila macgillivrayi*) (Endangered) (Davies, et al., 2015; Dilley, et al., 2015). Despite these impacts, the island still supports some 12 million breeding seabirds of 22 species and is regarded as a top-priority island for rodent eradication world-wide (Hilton & Cuthbert, 2010).

At 65 km^2, Gough will be the largest island where an eradication has been attempted targeting mice alone (mice were eradicated from 129 km^2 Macquarie Island (Australia), but they occurred at lower densities than on Marion due to the presence of black rats (*R. rattus*) on the island (Springer 2016; <http://www.parks.tas.gov.au/?base=13013>). Planning for the Gough Island eradication has involved more than a decade of research to ensure the highest probability of success (e.g. Angel & Cooper, 2006; Brown, 2007; Parkes, 2008; Wanless, et al., 2009; Cuthbert, et al., 2011a; Cuthbert, et al., 2011b; Cuthbert, et al., 2014; Cuthbert, et al., 2016). At 290 km^2, Marion Island is almost five times larger than Gough Island, but the terrain is less rugged, and the presence of a largely un-vegetated interior above 800 m with few, if any, mice in winter makes an eradication attempt at Marion less challenging in some regards (Parkes, 2016). The intention is to commence the operation during early winter, when mouse numbers are falling due to lack of food and cold conditions, increasing the likelihood of all animals consuming bait (see Parkes, 2019, for further details on the crucial decision of 'when to bait' on Marion). Mice also cease breeding on Marion from late May to August, reducing the chances of semi-independent young in the den failing to encounter bait (Parkes, 2016). Winter also coincides with the period of lowest numbers of brown skuas and giant petrels (*Macronectes* spp.) present on the island, which might be killed accidentally by either primary or secondary poisoning.

Mitigation plans will be needed to reduce the impacts on resident scavenging species (Wanless, et al., 2010). At this stage, the intention is to keep approximately 100 lesser sheathbills (*Chionis minor*) in captivity during the eradication attempt, given the moderate level of mortality of snowy sheathbills (*C. albus*) during the rodent eradication at South Georgia (Martin & Richardson, 2017). The Prince Edward Islands are home to an endemic subspecies of sheathbill *C. m. marionensis*, but nearby Prince Edward Island houses a substantial population of this subspecies and could be used to re-establish birds on Marion Island. Kelp gulls (*Larus dominicanus*) also are resident scavengers at Marion Island, but they may be less susceptible to non-target poisoning (Martin & Richardson, 2017). Given the small population size (Table 1) and difficulty of catching and maintaining captive birds, there is currently no plan to mitigate impacts on this species. Gulls are thought to move freely between Marion and Prince Edward Island, so immigration should aid the recovery of the Marion population after the eradication.

The eradication on Marion Island was stimulated by the donation of US$100,000 and the three helicopters used in the South Georgia rodent eradication by the Mamont Foundation to the South African Department of Environmental Affairs in early 2017. South Africa has a weather station on Gough Island, and will assist this eradication effort through the provision of accommodation (including possible refurbishments on the island), the hosting of the eradication team from its Cape Town harbour, and assistance with transportation. In return, the equipment used and expertise developed during the Gough eradication will be transferred to South Africa for use in the planned Marion eradication. The programme on Marion Island will be spearheaded by the Department of Environmental Affairs' Working for Water programme – Africa's biggest conservation programme focusing on the control of invasive species. Working for Water is already managing eradication projects against eight invasive vascular plant species on Marion Island, and the possible eradication of one introduced invertebrate, the rough woodlouse (*Porcellio scaber*), is being assessed (D. Muir). The South African Government is budgeting for this programme (with an initial budget of about US$2.2 million). It will seek to raise co-funding, including through a crowd-funding initiative being led by BirdLife South Africa, a non-governmental organisation.

Eradicating rodents from islands is an effective, long-term conservation management action, provided robust biosecurity measures are put in place to minimise the likelihood of any reintroductions. The South African National Antarctic Programme has imposed stringent quarantine measures on all vessels and materials going to the Prince Edward Islands (and Gough Island) since the early 1990s. These include fumigation of the resupply vessel prior to each voyage, use of rat guards on all hawsers when in harbour, placement of rodenticide baits at strategic points throughout the ship, and inspection of all cargo before being opened ashore (DST-NRF Centre of Excellence for Invasion Biology, 2014).

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REFERENCES

Angel, A., Wanless, R.M. and Cooper, J. (2009). 'Review of the impacts of house mice on islands'. *Biological Invasions* 11: 1743–1754.

Angel, A. and Cooper, J. (2006). *A Review of the Impacts of Introduced Rodents on the Islands of Tristan da Cunha and Gough*. Sandy, UK: Royal Society for the Protection of Birds.

Angel, A. and Cooper, J. (2011). *A Review of the Impacts of the House Mouse* Mus musculus *on Sub-Antarctic Marion Island, Prince Edward Islands*. Report to the Prince Edward Islands Management Committee, South African National Antarctic Programme.

Atkinson, I.A.E. (1985). 'The Spread of Commensal Species of *Rattus* to Oceanic Islands and their Effects on Island Avifaunas'. In: P.J. Moors (ed.) *Conservation of Island Birds*, pp. 35–81. ICBP Technical Publication 3. Cambridge, UK: International Council for Bird Preservation.

Avenant, N.L. and Smith, V.R. (2003). 'The microenvironment of house mice on Marion Island (sub-Antarctic)'. *Polar Biology* 26: 129–141.

Berry, R.J., Peters, J. and van Aarde, R.J. (1978). 'Sub-antarctic house mice: colonization, survival and selection'. *Journal of Zoology, London* 184: 127–141.

Bester, M.N., Bloomer, J.P., van Aarde. R.J., Erasmus, B.H., van Rensburg, P.J.J., Skinner, J.D., Howell, P.G. and Naude, T.W. (2002). 'A review of the successful eradication of feral cats from sub-Antarctic Marion Island, southern Indian Ocean'. *South African Journal of Wildlife Research* 32: 65–73.

BirdLife International (2017). 'IUCN Red List for Birds'. <http://www.birdlife.org>. Accessed 3 August 2017.

Brown, D. (2007). *A Feasibility Study for the Eradication of Rodents from Tristan da Cunha*. Sandy, UK: Royal Society for the Protection of Birds.

Caut, S., Angulo, E., Pisanu, B., Ruffino, L., Faulquier, L., Lorvelec, O., Chapuis, J.-L., Pascal, M., Vidal, E. and Courchamp, F. (2012). 'Seabird modulations of isotopic nitrogen on islands'. *PLOS ONE* 7: e39125.

Cerfonteyn, M. and Ryan, P.G. (2016). 'Have burrowing petrels recovered on Marion Island two decades after cats were eradicated? Evidence from sub-Antarctic skua prey remains'. *Antarctic Science* 28: 51–57.

Chapin, F.S.; Zavaleta, E.S., Viner, V.T., Naylor, R.L., Vitousek, P.M., Sala, O.E., Reynolds, H.L., Hooper, D.U., Mack, M., Diaz, S.E., Hobbie, S.E. and Lavorel, S. (2000). 'Consequences of changing biodiversity'. *Nature* 405: 234–242.

Chown, S.L. and Smith, V.R. (1993). 'Climate change and the short-term impact of feral house mice at the sub-Antarctic Prince Edward Islands'. *Oecologia* 96: 508–516.

Cooper, J. and Fourie, A. (1991). 'Improved breeding success of great-winged petrels *Pterodroma macroptera* following control of feral cats *Felis catus* at sub-Antarctic Marion Island'. *Bird Conservation International* 1: 171–175.

Cooper, J., Marais, A.v.N., Bloomer, J.P. and Bester, M.N. (1995). 'A success story: breeding of burrowing petrels (Procellariidae) before and after the extinction of feral cats *Felis catus* at sub-Antarctic Marion Island'. *Marine Ornithology* 23: 33–37.

Cooper, J., Baker, G.B., Double, M.C., Gales, R., Papworth, W., Tasker, M. and Waugh, S.M. (2006). 'The Agreement on the Conservation of Albatrosses and Petrels: Rationale, history, progress and the way forward'. *Marine Ornithology* 34: 1–5.

Croxall, J.P., Butchart, S.H., Lascelles, B., Stattersfield, A.J., Sullivan, B., Symes, A. and Taylor, P. (2012). 'Seabird conservation status, threats and priority actions: a global assessment'. *Bird Conservation International* 22: 1–34.

Cuthbert, R. and Hilton, G. (2004). 'Introduced house mice *Mus musculus*: a significant predator of threatened and endemic birds on Gough Island, South Atlantic Ocean?' *Biological Conservation* 117: 483–489.

Cuthbert, R.J., Visser, P., Louw, H. and Ryan, P.G. (2011a). 'Palatability and efficacy of rodent baits for eradicating house mice (*Mus musculus*) from Gough Island, Tristan da Cunha'. *Wildlife Research* 38: 196–203.

Cuthbert, R.J., Parker, G., Louw, H., Rexer-Huber, K., Parker, G. and Ryan, P.G. (2011b). 'Preparations for the eradication of mice from Gough Island: results of bait acceptance trials above ground and around cave systems'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 47–50. Occasional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Cuthbert, R.J., Louw, H., Parker, G., Rexer-Huber, K. and Visser, P. (2013). 'Observations of mice predation on dark-mantled sooty albatross and Atlantic yellow-nosed albatross chicks at Gough Island'. *Antarctic Science* 25: 763–766.

Cuthbert, R.J., Broome, K., Bradley, J. and Ryan, P.G. (2014). 'Evaluating the effectiveness of aerial baiting operations for rodent eradications on cliffs on Gough Island, Tristan da Cunha'. *Conservation Evidence* 11: 25–28.

Cuthbert, R.J., Wanless, R.M., Angel, A., Burlé, M.-H., Hilton, G.M., Louw, H., Visser, P., Wilson, J. and Ryan, P.G. (2016). 'Drivers of predatory behaviour and extreme body size in house mice *Mus musculus* on Gough Island'. *Journal of Mammalogy* 97: 533–544.

Davies, D., Dilley, B.J., Bond, A.L., Cuthbert, R.J. and Ryan, P.G. (2015). 'Trends and tactics of mouse predation on Tristan Albatross *Diomedea dabbenena* chicks at Gough Island, South Atlantic Ocean'. *Avian Conservation and Ecology* 10(1): 5 (doi: 10.5751/ACE-00738-100105).

de Villiers, M.S. and Cooper, J. (2008). 'Conservation and Management'. In: S.L Chown and P.W. Froneman (eds). *The Prince Edward Islands: Land–Sea Interactions in a Changing Ecosystem,* pp. 113–131. Stellenbosch, South Africa: SunPress.

de Villiers, M.S., Chown, S.L. and Cooper, J. (2011). *Prince Edward Islands Conservation Handbook. Keeping your Footprint at the Islands Small*. Stellenbosch, South Africa: SunPress.

Dilley, B.J., Davies, D., Bond, A.L. and Ryan, P.G. (2015). 'Effects of mouse predation on burrowing petrel chicks at Gough Island'. *Antarctic Science* 27: 543–553.

Dilley, B.J., Schoombie, S., Schoombie, J. and Ryan, P.G. (2016a). ''Scalping' of albatross fledglings by introduced mice spreads rapidly at Marion Island'. *Antarctic Science* 28: 73–80.

Dilley, B.J., Schramm, M. and Ryan, P.G. (2016b). 'Modest increases in densities of burrow-nesting petrels following the removal of cats *Felis catus* from Marion Island'. *Polar Biology* 40: 25–637.

Dilley, B.J., Davies, D., Schramm, M., Connan, M. and Ryan, P.G. (2017). 'The distribution and abundance of Blue Petrels (*Halobaena caerulea*) breeding at subantarctic Marion Island, *Emu– Austral Ornithology* 117: 222–232.

Dilley, B.J., Schoombie, S., Stevens, K., Davies, D., Perold, V., Osborne, A., Schoombie, J., Brink, C.W., Carpenter-Kling, T. and Ryan, P.G. (2018). 'Mouse predation affects breeding success of burrow-nesting petrels at sub-Antarctic Marion Island'. *Antarctic Science* 30: 93–104.

DIISE (2015). 'The Database of Island Invasive Species Eradications, developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand'. <http://diise.islandconservation.org>. Accessed 3 August 2017.

DST-NRF Centre of Excellence for Invasion Biology (2014). *Prince Edward Islands Management Plan Version 0.2*. Pretoria, South Africa: Department of Environmental Affairs.

Ferreira, S.M., van Aarde, R.J. and Wassenaar, T.D. (2006). 'Demographic responses of house mice to density and temperature on sub-Antarctic Marion Island'. *Polar Biology* 30: 83–94.

Fugler, S.R., Hunter, S., Newton, I.P. and Steele, W.K. (1987). 'Breeding biology of blue petrels *Halobaena caerulea* at the Prince Edward Islands'. *Emu* 87: 103–110.

Hilton, G.M. and Cuthbert, R.J. (2010). 'The catastrophic impact of invasive mammalian predators on birds of the UK Overseas Territories: a review and synthesis'. *Ibis* 152: 443–458.

IUCN (2000). *Guidelines for the Prevention of Biodiversity Loss Due to Biological Invasion*. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Jones, H.P., Tershy, B.R., Zavaleta, E.S., Croll, D.A., Keitt, B.S., Finkelstein, M.E. and Howald, G.R. (2008). 'Severity of the effects of invasive rats on seabirds: a global review'. *Conservation Biology* 22: 16–26.

Jones, M.G.W. and Ryan, P.G. (2010). 'Evidence of mouse attacks on albatross chicks on sub-Antarctic Marion Island'. *Antarctic Science* 22: 39–42.

Le Corre, M. (2008). 'Conservation biology: cats, rats and seabirds'. *Nature* 451: 134–135.

le Roux, P.C. and McGeoch, M.A. (2008). 'Changes in climate extremes, variability and signature on sub-Antarctic Marion Island'. *Climatic Change* 86: 309–329.

Lombard, A.T., Reyers, B., Schonegevel. L.Y., Cooper, J., Smith-Adao, L.B., Nel, D.C., Froneman, P.W., Ansorge, I.J., Bester, M.N., Tosch, C.A., Strauss, T., Akkers, T., Gon, O., Leslie. R.W. and Chown, S.L. (2007). 'Conserving pattern and process in the Southern Ocean: designing a Marine Protected Area for the Prince Edward Islands'. *Antarctic Science* 19: 39–54.

McClelland, G.T.W., Altwegg, R., van Aarde, R.J., Ferreira, S., Burger, A.E. and Chown, S.L. (2018). 'Climate change leads to increasing population density and impacts of a key island invader'. *Ecological Applications* 28: 212–224.

Manne, L.L., Brooks, T.M. and Pimm, S.L. (1999). 'Relative risk of extinction of passerine birds on continents and islands'. *Nature* 399: 258–261.

Martin, A.R. and Richardson, M.G. (2017). Rodent eradication scaled up: clearing rats and mice from South Georgia. *Oryx*: doi:10.1017/S003060531700028X.

Mooney, H.A. and Cleland, E.E. (2001). 'The evolutionary impact of invasive species'. *Proceedings of the National Academy of Sciences USA* 98: 5446–5451.

Nel, D. and Omardien, A. (eds). (2008). *Towards the Development of a Marine Protected Area at the Prince Edward Islands*. WWF South Africa Report Series 2008/Marine/001. South Africa: WWF Sanlam Living Waters Partnership.

Parkes, J. (2008). *A Feasibility Study for the Eradication of House Mice from Gough Island*. RSPB Research Report No.34. Sandy, UK: Royal Society for the Protection of Birds.

Parkes, J. (2016). 'Eradication of House Mice *Mus musculus* from Marion Island: A Review of Feasibility, Constraints and Risks'. In: R.M. Wanless (ed). *BirdLife South Africa Occasional Report Series* No. 1. Johannesburg, South Africa: BirdLife South Africa.

Parkes, J.P. (2019). 'Timing aerial baiting for rodent eradications on cool temperate islands: mice on Marion Island'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 36–39. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Phiri, E.E., McGeoch, M.A. and Chown, S.L. (2009). 'Spatial variation in structural damage to a keystone plant species in the sub-Antarctic: interactions between *Azorella selago* and invasive house mice'. *Antarctic Science* 21: 189–196.

Rand, R.W. (1954). 'Notes on the birds of Marion Island'. *Ibis* 96: 173–206.

Rayner, M.J., Hauber, M.E., Imber, M.J., Stamp, R.K. and Clout, M.N. (2007). 'Spatial heterogeneity of mesopredator release within an oceanic island system'. *Proceedings of the National Academy of Sciences USA* 104: 20862–20865.

Ryan, P.G. and Bester, M.N. (2008). 'Pelagic Predators'. In: S.L. Chown and P.W. Froneman (eds). *The Prince Edward Islands: Land–Sea Interactions in a Changing Ecosystem,* pp. 121–164. Stellenbosch, South Africa: Sun Press.

Ryan, P.G., Whittington, P.A. and Crawford, R.J.M. (2009). 'A tale of two islands: contrasting fortunes for sub-Antarctic skuas at the Prince Edward Islands'. *African Journal of Marine Science* 31: 431–437.

Ryan, P.G., Dilley, B.J. and Jones, M.G.W. (2012). 'The distribution and abundance of white-chinned petrels (*Procellaria aequinoctialis*) breeding at the sub-Antarctic Prince Edward Islands'. *Polar Biology* 35: 1851–1859.

Schoombie, S., Crawford, R.J.M., Makhado, A.B., Dyer, B.M. and Ryan, P.G. (2016). 'Recent population trends of sooty and light-mantled albatrosses breeding on Marion Island'. *African Journal of Marine Science* 38:119–127.

Springer, K. (2016). 'Methodology and challenges of a complex multi-species eradication in the sub-Antarctic and immediate effects of invasive species removal'. *New Zealand Journal of Ecology* 40: 273–278.

Taylor, M.R., Peacock, F. and Wanless, R.M. (2015). *The 2015 ESKOM Red Data Book of Birds of South Africa, Lesotho and Swaziland*. Dunkeld West, South Africa: BirdLife South Africa.

Trathan, P.N. and Reid, K. (2009). 'Exploitation of the marine ecosystem in the sub-Antarctic: historical impacts and current consequences'. *Papers and Proceedings of the Royal Society of Tasmania* 143: 9–14.

van Aarde, R.J. (1980). 'The diet and feeding behaviour of feral cats, *Felis catus* at Marion Island'. *South African Journal of Wildlife Research* 10: 123–128.

Wanless, R.M., Angel, A., Cuthbert, R.J., Hilton, G.M. and Ryan, P.G. (2007). 'Can predation by invasive mice drive seabird extinctions?' *Biology Letters* 3: 241–244.

Wanless, R.M., Fisher, P., Cooper, J., Parkes. J. and Ryan, P.G. (2008). 'Bait acceptance by house mice: an island field trial'. *Wildlife Research* 35: 806–811.

Wanless, R.M., Ryan, P.G., Altwegg, R., Angel, A., Cooper, J., Cuthbert, R. and Hilton, G.M. (2009). 'From both sides: dire demographic consequences of carnivorous mice and longlining for the critically endangered Tristan albatrosses on Gough Island'. *Biological Conservation* 142: 1710–1718.

Wanless, R.M., Cooper, J., Slabber, M.J., and Ryan, P.G. (2010). 'Risk assessment of birds foraging terrestrially at Marion and Gough Islands to primary and secondary poisoning'. *Wildlife Research* 37: 524–530.

Watkins, B.P. and Cooper, J. (1986). 'Introduction, present status and control of alien species at the Prince Edward Islands, sub-Antarctic'. *South African Journal of Antarctic Research* 16: 86–94.

Zavaleta, E.S., Hobbs, R.J. and Mooney, H.A. (2001). 'Viewing invasive species removal in a whole-ecosystem context'. *Trends in Ecology and Evolution* 16: 454–459.

Improving the efficiency of aerial rodent eradications by means of the numerical estimation of rodenticide density

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Abstract Invasive rodents are present on approximately 80% of the world's islands and constitute one of the most serious threats to island biodiversity and ecosystem functioning. The eradication of rodents is central to island conservation efforts and the aerial broadcast of rodenticide bait is the preferred dispersal method. To improve the efficiency of rodent eradication campaigns, the generation of accurate and real-time bait density maps is required. Creating maps to estimate the spatial dispersion of bait on the ground has been carried out using traditional GIS methodologies, which are based on limiting assumptions and are time intensive. To improve accuracy and expedite the evaluation of aerial operations, we developed an algorithm for the numerical estimation of rodenticide density (NERD). The NERD algorithm performs calculations with increased accuracy, displaying results almost in real-time. NERD describes the relationship between bait density, the mass flow rate of rodenticide through the bait bucket, and helicopter speed and produces maps of bait density on the ground. NERD also facilitates the planning of helicopter flight paths and allows for the instant identification of areas with low or high bait density. During the recent and successful rodent eradication campaign on Banco Chinchorro in Mexico, carried out during 2015, NERD results were used to enable dynamic decision-making in the field and to ensure the efficient use of resources.

Keywords: aerial dispersal, bait density, invasive species, rodenticide

INTRODUCTION

Island ecosystems are vulnerable to the threat posed by invasive species due to the combination of high levels of endemism and isolation, coupled with smaller population sizes (Loope & Mueller-Dombois, 1989; D'Antonio & Dudley, 1995; Reaser, et al., 2007). Invasive rodent species such as *Rattus rattus* are particularly harmful to island ecosystems. Worldwide, invasive rodents are found on more than 80% of the world's islands and their high potential for dispersal indicates that this number is on the rise (Russell, et al., 2008; Harris, et al., 2012). The presence of invasive rodents on islands can lead to rapid population decreases of both flora and fauna and the extirpation of endemic species (Towns, et al., 2006; Medina, et al., 2011) as invasive rodent species begin to dominate communities (Angel, et al., 2009; Towns, et al., 2013). Island biodiversity is not only affected by the presence of invasive rodents; in cases where rodent invasion is severe, key island ecosystem functions and services are often lost (e.g., Towns, et al., 2006). Island ecosystems are unable to recover while rodents are present; as such, the first step to restore ecological functioning and island biodiversity is the eradication of invasive rodent species via the dispersal of rodenticide (Towns & Broome, 2003; Harris, et al., 2012).

The aerial-based dispersal methods of rodenticide bait via helicopter are preferable to ground-based methods in many circumstances (Towns & Broome, 2003; Broome, et al., 2014). Aerial bait dispersal strategies are designed to cover large areas rapidly, reduce the complications associated with complex topography, and target potential refuge sites (Towns & Broome, 2003; Howald, et al., 2007). The evaluation of the effectiveness of aerial rodenticide dispersal is informed by bait density maps that show the spatial variation of bait on the ground (Broome, et al., 2014). Traditionally, bait density maps have been created with *in situ* measurements or from GPS helicopter trajectories although there are challenges associated with both methods. To obtain *in situ* measurements, quadrat bait density sampling is carried out on the ground and requires a substantial investment of both time and human resources. The effectiveness of this method depends on the topography, accessibility, and climate of the island at the time of sampling, in addition to existing time constraints and available manpower. In contrast, the spatial estimation of bait density from recorded GPS helicopter trajectories is time intensive and can be imprecise as it is based on several untested assumptions, the principal one being that the bait density remains constant within the treated polygon.

We have developed a method for the numerical estimation of rodenticide density (NERD) that improves upon the aforementioned methods. NERD creates bait density maps using GPS helicopter trajectories but is not constrained by the assumptions of traditional GIS analysis. NERD does not assume that bait density is constant within the treated polygon nor is it time intensive. Results from NERD are both automatic and instantaneous, allowing for modifications to helicopter flight plans during an ongoing eradication. During helicopter refuelling, GPS data from the helicopter are downloaded into NERD and bait density maps are returned in minutes.

The NERD algorithm combines two models. The first model estimates the mass flow rate as a function of the bait bucket aperture diameter and the second model describes the bait density profile perpendicular to the flight path of the helicopter. By combining the two models, bait density on the ground is estimated as a function of the aperture diameter of the bait bucket and the speed of the helicopter.

In this paper, we present the first field implementation of NERD on the island of Banco Chinchorro, Mexico, a small false atoll in which rodents were most likely introduced during the 19th century (Samaniego, et al., 2017).

METHODS

Study site

Banco Chinchorro is comprised of four flat keys that create a false atoll measuring 0.5–539 ha, located in the Caribbean Sea approximately 35 km off the coast of Quintana Roo, Mexico, and is classified as both a Biosphere Reserve and Ramsar site (CONANP, 2000; 2006; Samaniego, et al., 2017). Banco Chinchorro presents a wet tropical climate and is primarily covered with mangrove vegetation, composed of *Rhizophora mangle*,

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Laguncularia racemosa, *Avicennia germinans*, and *Conocarpus erectus*, and has tropical trees such as *Thrinax radiata*, *Bursera simaruba*, and *Tournefortia gnaphalodes* (Samaniego, et al., 2017). The island provides habitat for a number of crab species, the American crocodile (*Crocodylus acutus*) and the seabird *Fregata magnificens* (Samaniego, et al., 2017). Prior to eradication efforts, the invasive rodent (*Rattus rattus)* occurred at densities from 6.5–47.9 rats/ha on Cayo Centro to 25.3–102.5 rats/ha on Cayo Norte Major (Samaniego, et al., 2017). The extensive mangrove presence on Banco Chinchorro and the presence of the *C. acutus* makes ground-based evaluation methods of bait density both hazardous and ineffectual.

Relationship between density, mass flow rate, and helicopter speed

The combination of the two models comprising NERD is presented. Here, we show that the function $\sigma(x,y)$ used to represent superficial bait density (kg/m²) complies with the following equation

$$\int_{-\frac{w}{2}}^{+\frac{w}{2}} \sigma(x)dx = \frac{\dot{m}}{s}$$

where $\dot{m}$ is the bait flow (kg/s), s is the speed of the helicopter (m/s) and w is the swath width (m).

We set the origin of a Cartesian coordinate system on the middle point of the bottom side of a rectangle with base w and height δy. This way, the bottom side is found at $y = 0$, the top side at $y = \delta y$, the left side at $x = -\frac{w}{2}$, and the right side at $x = +\frac{w}{2}$. The rectangle represents one dispersion cell.

After the helicopter completes a pass, in each point (x,y) of the dispersion cell, a superficial bait density is obtained $\sigma(x,y)$. In instances where two or more dispersions cells overlap, we simply add the density from each cell to get the total density on the overlap. The definition of the superficial bait density of mass m indicates that $\sigma(x,y) = \frac{dm}{dA}$. Rewriting the superficial density substituting dA by $dydx$ and integrating along the dispersion cell, it follows that

$$\delta m = \int_{-\frac{w}{2}}^{+\frac{w}{2}} \int_{0}^{\delta y} \sigma(x,y)dydx. \qquad (1)$$

Assuming superficial density is uniform with respect to the helicopter's flight path, represented in Fig. 1, equation (1) becomes

$$\frac{\delta m}{\delta y} = \int_{-\frac{w}{2}}^{+\frac{w}{2}} \sigma(x)dx. \qquad (2)$$

The left-hand side of the equation represents the linear bait density, which is related with the mass flow of bait from the bucket and the speed of the helicopter. A helicopter equipped with a dispersion bucket with a constant mass flow rate,

$$\dot{m} = \frac{\delta m}{\delta t} \qquad (3)$$

flies from the point (0,0) to the point $(0,\delta y)$ with a speed of

$$s = \frac{\delta y}{\delta t}. \qquad (4)$$

Combining equations (3) and (4), the linear bait density

$$\frac{\delta m}{\delta y} = \frac{\dot{m}}{s} \qquad (5)$$

is obtained.

Finally, setting equations (2) and (5) equal to each other, we obtain

$$\int_{-\frac{w}{2}}^{+\frac{w}{2}} \sigma(x)dx = \frac{\dot{m}}{s}. \qquad (6)$$

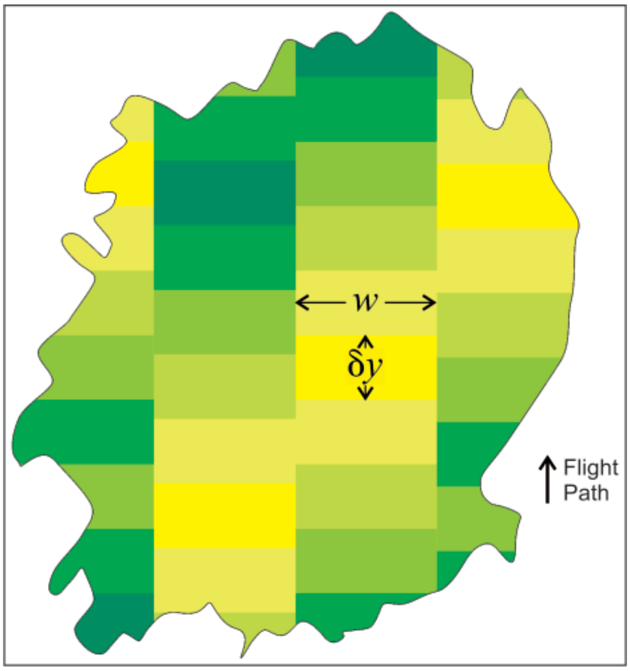

Fig. 1 Hypothetical island with bait swaths. Each vertical band represents one bait swath. Each shaded rectangle represents one dispersion cell. Shade intensity corresponds to bait density, with darker shades indicating higher densities.

Equation (6) relates the bait density on the ground with the mass flow rate and the speed of the helicopter. In order to get an explicit form of σ, a model is fitted to cross-density profiles, such as the ones shown in Fig. 2.

Simplified relationship between density, mass flow rate, and helicopter speed

The required bait density for the successful eradication of an invasive species on an island is determined by evaluating the ecosystems present and the biology of the target species. Once this density has been determined, NERD can be used to estimate the aperture of the bait bucket needed for the eradication operation in question and to plan helicopter flight paths. During the planning phase of an eradication campaign, prior to arriving on the island, a simplified relationship between density, mass flow rate, and helicopter speed is used where bait density is assumed to be constant along and across the flight path of the helicopter.

Assuming density is independent of x, i.e. σ does not change perpendicular to the flight path, equation (6) can be easily solved to obtain

$$\sigma = \frac{\dot{m}}{s \cdot w}. \qquad (7)$$

To write equation (7) as a function of the aperture diameter of the bait bucket, we express the mass flow rate of bait as a function of the aperture diameter, $\dot{m}(d)$. To do this, the bait in the bucket was weighed and the time required to empty the bucket was measured and repeated using several aperture diameters (Fig. 3).

The resulting three-dimensional model,

$$\sigma(d,s) = \frac{\dot{m}(d)}{s \cdot w} \qquad (8)$$

is shown in Fig. 4.

An implementation of this model can be found at <*http://github.com/IslasGECI/nerd*>.

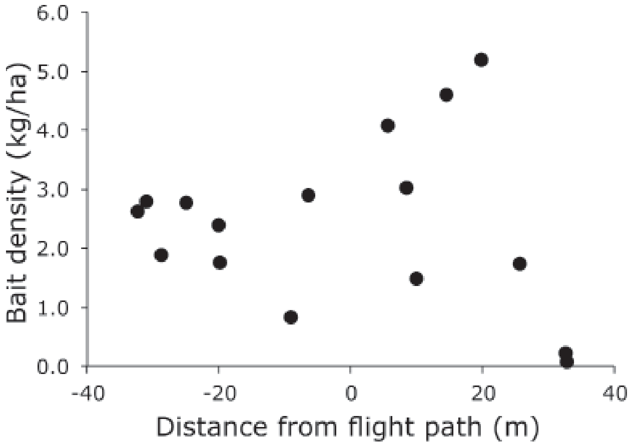

Fig. 2 Bait density profile perpendicular to the flight path of the helicopter during a test flight in Oxnard, CA in 2013. Each black dot shows the bait density measured within a quadrat.

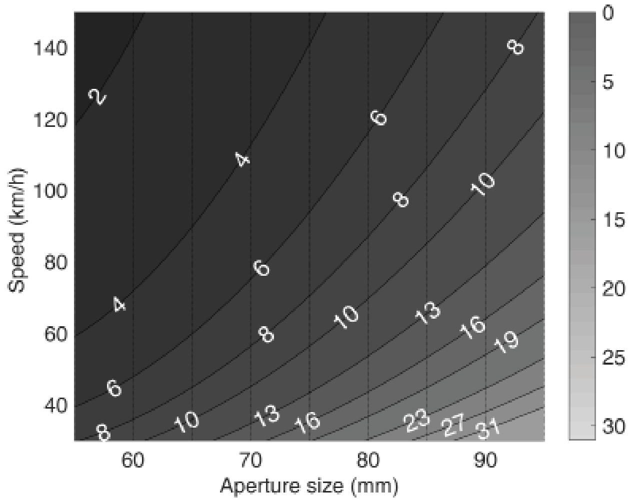

Fig. 4 Surface bait density σ (kg/ha) as a function of aperture diameter d (mm) and speed s (km/hr). The horizontal axis shows the aperture diameter of the bait bucket and the vertical axis shows the speed of the helicopter. The resulting bait density on the ground is shown in white superimposed numbers and in the second vertical grayscale axis.

RESULTS AND DISCUSSION

NERD was used to plan and carry out the 2015 eradication campaign on Cayo Centro of Banco Chinchorro. Given the desired helicopter speed, NERD was used to determine the aperture of the bait bucket and the flight paths of the helicopter required to achieve the desired bait density within the target polygon. The results of the 2015 rodent eradication campaign on Banco Chinchorro are detailed by Samaniego et al. (2017). During the course of the eradication campaign, NERD was operated by two people and generated an updated bait density map multiple times each day providing instantaneous visualisations of the current state of bait application over the island, such as the map shown in Fig. 5. These visualizations provided feedback in real time, allowing for helicopter course corrections and promoting the efficient use of rodenticide bait.

Fig. 5 shows the final bait density map estimated with NERD for the eradication campaign. From this map, it is apparent that all terrestrial areas of Banco Chinchorro were estimated to be covered with at least 60 kg/ha of rodenticide, which was the target bait density for this campaign. The colormap of Fig. 5 indicates bait density on the ground (kg/ha), with warmer colours corresponding to lower bait densities. The large red polygons that appear on the map represent inland lagoons, which were not covered with rodenticide bait excepting a few swaths that correspond to the presence of sandbars within the lagoons. Around these lagoons, manual bait placement was carried out by a team of field operatives. The maps generated by NERD were also used by this team to ensure even bait coverage and avoid excess bait application. Overall, few areas in Fig. 5 show bait densities near 100 kg/ha, indicating that helicopter flight paths were rarely redundant. Furthermore, any small isolated areas of low bait density were always surrounded by areas with target bait densities of at least 60 kg/ha.

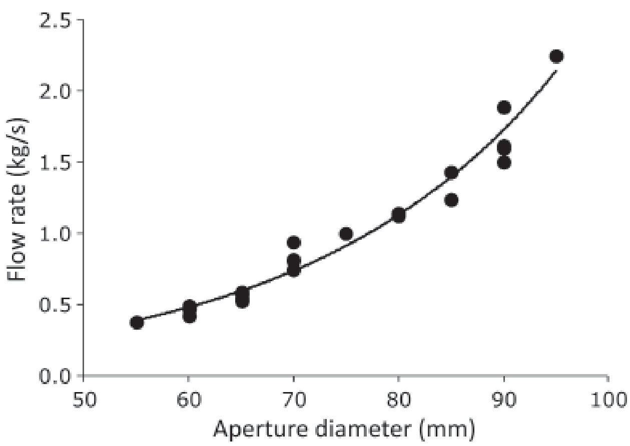

Fig. 3 Mass flow rate (kg/s) as a function of aperture diameter d (mm). Each dot represents a calibration event and the black curve is the quadratic model fitted to the data.

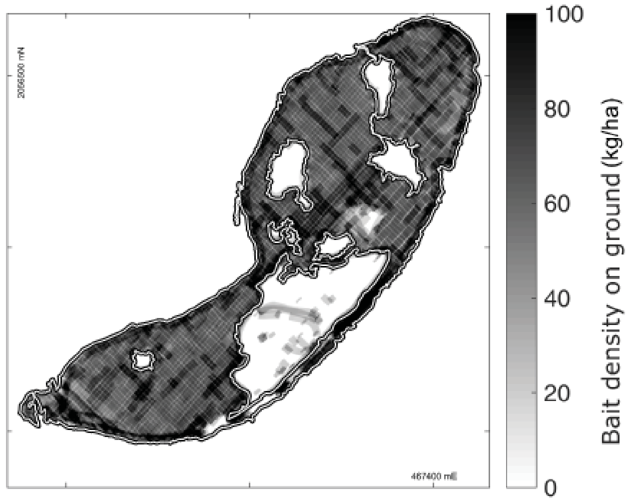

Fig. 5 Estimated bait density (kg/ha) resulting from the aerial operation of the rodent eradication campaign in Banco Chinchorro, Mexico, during 2015.The shade bar on the right indicates predicted bait density on the ground (kg/ha), with lighter shades indicating lower densities. The large white polygons show the location of inland lagoons.

The information provided by NERD was indispensable to the eradication campaign on Banco Chinchorro and allowed for immediate decisions to be made regarding not only the aerial dispersal of rodenticide bait, but also for the manual placement of bait on the ground. Until now, efforts to generate bait density maps have been inefficient and results were often not available until after the end of an eradication campaign. NERD provides information in real time, enabling dynamic decision making in the field and ensuring the efficient use of resources.

REFERENCES

Angel, A., Wanless, R.M. and Cooper, J. (2009). 'Review of impacts of the introduced house mouse on islands in the Southern Ocean: Are mice equivalent to rats?' *Biological Invasions* 11: 1743–1754.

Broome, K.G., Cox, A., Golding, C., Cromarty, P., Bell, P. and McClelland, P. (2014). *Rat Eradication Using Aerial Baiting: Current Agreed Best Practice Used in New Zealand (Version 3.0)*. Wellington, New Zealand: Department of Conservation.

CONANP. (2000). *Programa de Manejo Reserva de la Biosfera Banco Chinchorro*. Mexico City, Mexico: INE-SEMARNAT.

CONANP. (2006). *Programa de Conservación y Manejo Parque Nacional Arrecife Alacranes*. Mexico City, Mexico: Comisión Nacional de Áreas Naturales Protegidas.

D'Antonio, C.M. and Dudley, T.L. (1995) 'Biological Invasions as Agents of Change on Islands Compared with Mainlands'. In: P.M. Vitousek, L. Loope and H. Adsersen (eds.) *Islands: Biological Diversity and Ecosystem Function*, pp. 103–121. Berlin, Germany: Springer Verlag.

Harris, D.B., Gregory, S.D., Bull, L.S., and Courchamp, F. (2012). 'Island prioritization for invasive rodent eradications with an emphasis on reinvasion risk'. *Biological Invasions* 14(6): 1251–1263.

Howald, G., Donlan, C.J., Galvan, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007) 'Invasive rodent eradication on islands'. *Conservation Biology* 21(5): 1258–1268.

Loope, L. and Mueller-Dombois D. (1989). 'Characteristics of Invaded Islands, with Special Reference to Hawaii'. In: J.A. Drake, H.A. Mooney, F. di Castri, R.H. Groves, F.J. Kruger, M. Rejmanek and M. Williamson (eds.). *Scope 37 – Biological Invasions: A Global Perspective*, pp. 257–280. Chichester: John Wiley & Sons.

Medina, F.M., Bonnaud, E., Vidal, E., Tershy, B.R., Zavaleta, E.S., Donlan, C.J., Keitt, B.S., Le Corre, M., Horwarth, S.V. and Nogales, M. (2011). 'A global review of the impacts of invasive cats on island endangered vertebrates'. *Global Change Biology* 17(11): 3503–3510.

Reaser, J. K., Meyerson, L. A., Cronk, Q., De Poorter, M., Eldrege, L. G., Green, E. and Vaiutu, L. (2007). 'Ecological and socioeconomic impacts of invasive alien species in island ecosystems'. *Environmental Conservation* 34(2): 98–111.

Russell, J.C., Towns, D.R. and Clout, M.N. (2008). *Review of Rat Invasion Biology: Implications for Island Biosecurity.* Science for Conservation 286. Wellington, New Zealand: Department of Conservation.

Samaniego-Herrera, A., Aguirre-Muñoz, A., Bedolla-Guzmán, Y., Cárdenas-Tapia, A., Félix-Lizárraga, M., Méndez-Sánchez, F., Reina-Ponce, O., Rojas-Mayoral, E. and Torres-García, F. (2017). 'Eradicating invasive rodents from wet and dry tropical islands in Mexico'. *Oryx* 1–12: DOI:10.1017/S0030605316001150.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?' *Biological Invasions* 8(4): 863–891.

Towns, D.R. and Broome, K.G. (2003). 'From small Maria to massive Campbell: Forty years of rat eradications from New Zealand islands'. *New Zealand Journal of Zoology* 30(4): 377–398.

Towns, D.R., West, C.J. and Broome, K.G. (2013). 'Purposes, outcomes and challenges of eradicating invasive mammals from New Zealand islands: An historical perspective'. *Wildlife Research*, 40: 94–107.

Preventing extinctions: planning and undertaking invasive rodent eradication from Pinzon Island, Galapagos

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Abstract Invasive black rats (*Rattus rattus*) were successfully eradicated during 2012 from Pinzon Island in the Galapagos archipelago using the rodenticide brodifacoum. Potential exposure to brodifacoum in Pinzon tortoises (*Chelonoidis ephippium*), Pinzon lava lizards (*Microlophus duncanensis*) and Galapagos hawks (*Buteo galapagoensis*) was mitigated by captive holding of subpopulations. This was successful for all species during and shortly after baiting, however mortality of Galapagos hawks occurred post-release, likely due to the persistence of residual brodifacoum in lava lizards. Since 2013, Pinzon tortoise hatchlings are surviving in-situ for the first time in at least 120 years and the eradication of black rats is expected to have significant benefits for at least 15 other island-endemic species.

Keywords: brodifacoum, endemic species, eradication, *Rattus rattus*, restoration

INTRODUCTION

Islands are centres of endemism and endangerment, with about one-fifth of the world's threatened amphibians, one-quarter of the threatened mammals and more than one-third of the threatened birds being endemic to islands (Fonseca, et al., 2006). Invasive non-native species are major extinction drivers, with predators like rodents being particularly damaging (Bellard, et al., 2016; Doherty, et al., 2016). Four rodent species (*Rattus rattus, R. norvegicus, R. exulans, Mus musculus*) have been introduced to islands holding 88% of all insular critically endangered or endangered terrestrial vertebrates (TIB Partners, 2014). Invasive rodents cause population declines and extinctions of insular flora and fauna and interrupt ecosystem processes with negative cascading effects (Fukami, et al., 2006; Towns, et al., 2006; Jones, et al., 2008; Kurle, et al., 2008). To recover endangered populations and restore ecosystem processes, invasive rodents on islands are increasingly targeted for eradication, with at least 650 eradication attempts of introduced *Rattus* spp. populations to date (Russell & Holmes, 2015). Eradication of invasive mammals from islands results in positive responses by native species with few exceptions (Jones, et al., 2016).

Pinzon Island (1,815 ha), in the Galapagos archipelago, is uninhabited and is entirely within the Galapagos National Park. Pinzon endemics include three reptiles (Pinzon Island tortoise (*Chelonoidis ephippium*), Pinzon lava lizard (*Microlophus duncanensis*), Pinzon leaf-toed gecko (*Phyllodactylus duncanensis*)), six land snails (*Bulimulus duncanus, B. eschariferus ventrosus, B. pinzonensis, B. pinzonopsis, B. prepinguis, Bulimulus* sp. undescribed), and six insects in the orders Homoptera and Hemiptera. Thirteen species considered threatened by the IUCN are present, such as marine iguanas (*Amblyrhynchus cristatus*), Galápagos hawk (*Buteo galapagoensis*), land snails and the cactus *Opuntia galapageia*, along with several species of unassessed conservation status (IUCN, 2015).

The island was most heavily used by whalers harvesting tortoises in the early to mid-1800s and it is during this period that black rats (*R. rattus*) were most likely introduced, with specimens first collected in 1891 (Patton, et al., 1975). Black rats are the only invasive mammals that successfully populated the island. On visiting Pinzon Island in 1903, Rolland Beck noted "We… captured altogether nearly thirty live tortoises…. We were much chagrined, however, at finding no very small specimens, but soon came to the conclusion that the large rats, of recent introduction, and now common everywhere on the island, eat the young as soon as they are hatched" (Beck, 1903 p. 174). Heavy predation by black rats on eggs and hatchlings saw a halt of recruitment into the tortoise population for over a century, leaving fewer than 65 old tortoises that had survived human harvesting efforts (MacFarland, et al., 1974; Jensen, et al., 2015). In response, a 'head-starting' programme was initiated nearly 50 years ago. This entailed collecting eggs or recently hatched individuals from nests on-island, transporting them to the Galapagos National Park's centre on Santa Cruz Island where hatchlings were reared ex-situ until 4–5 years old, at which time they were repatriated back to Pinzon Island (Jensen, et al., 2015). Elsewhere in the Galapagos Archipelago, invasive black rats have been implicated in the extinction of native rodents, declines and extirpations of sea- and land-bird populations and other fauna (Cruz & Cruz, 1987; Steadman, et al., 1991; Dowler, et al., 2000). By consuming seeds and seedlings they impede vegetation regeneration and alter forest dynamics, affecting entire ecosystems (Clark, 1981). Impacts on invertebrates have not been quantified in the Galapagos Archipelago but likely occur based on reports from elsewhere (e.g. Towns, et al., 2006).

Conservationists attempted to eradicate black rats from Pinzon Island in 1988 utilising rodenticide bait dumps (coumatetralyl powder combined with rice in paper bags) and hand broadcast of baits containing brodifacoum and coumatetralyl (Cayot, et al., 1996; Harper & Carrion, 2011). The project was unsuccessful, although rodents were not detected for nine months after the operation (Cayot, et

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al., 1996). This rodent suppression resulted in recruitment of Pinzon tortoises, anecdotal reports of increases in the abundance of juvenile marine iguanas, populations of Pinzon lava lizards and Galapagos doves (*Zenaida galapagoensis*), and decreases in populations of short-eared owl (*Asio flammeus galapagoensis*) and Galapagos hawks (Muñoz, 1990; Cayot, et al., 1994; Cayot, et al., 1996). A cessation of predation of Pinzon tortoise hatchlings by black rats was recorded, however an 80% predation rate by native Galapagos hawks occurred for two years after the eradication attempt (Morillo Manrique, 1992). Ambitious for its time, this failed eradication attempt set back rodent eradications in the archipelago for the next three decades, with the exception of attempts on just a few small (<20 ha) islands (Harper & Carrion, 2011).

Large-scale feral pig (*Sus scrofa*), goat (*Capra hircus*) and donkey (*Equus asinus*) eradications were implemented in the Galapagos Archipelago throughout the late 1990s and 2000s (Cruz, et al., 2005; Carrion, et al., 2007; Carrion, et al., 2011) renewing interest in large-scale rodent eradications. In 2007, an international workshop laid out a plan for developing capacity and confidence to eventually eradicate rodents from inhabited Floreana Island (17,253 ha) with complexity and scale being increased at each step (CDF & GNPS, 2007). Later in 2007, North Seymour Island (184 ha) was hand baited with wax blocks containing brodifacoum, successfully eradicating black rats (Harper, et al., 2011). In 2011, the first aerial broadcast of brodifacoum baits in South America eradicated rodents from Rabida and 11 other islands totalling 705 ha (Campbell, et al., 2013). Pinzon (1,815 ha) and Plaza Sur (12 ha) islands were originally considered within the group of islands to be targeted along with Rabida but their operations were delayed to allow trials to be conducted for increasing certainty of non-target risks to tortoises and for pilot mitigation strategies for Galapagos hawk (Campbell, et al., 2013). As part of the Rabida project, 20 Galapagos hawks were kept in captivity and released once the risk of mortality from rodenticide poisoning was considered past (Campbell, et al., 2013).

Here we describe the successful eradication of invasive black rats from Pinzon Island and the measures taken to mitigate negative impacts of rodenticide bait application on non-target wildlife.

METHODS

Site description

Pinzon Island, located in the centre of the Galapagos Archipelago, has a maximum elevation of 458 m and approximately 18 km of rocky coastline with steep cliffs on the southern and north-western coasts. Large lava blocks dominate the slopes of Pinzon. There are two craters at the centre of the island. The vegetation is xerophytic and there are no permanent bodies of water. Two small islets, each of approximately 0.4 ha in size, lie close inshore. Pinzon has no terrestrial visitor sites and is more than 10 km from any other island with invasive rodents, making unassisted reinvasion highly unlikely.

Baseline genetic sampling of rodents from Pinzon

In 2011, black rats were trapped, euthanised and samples taken (n=89) for future genetic analyses in case rodents were detected after the eradication attempt. If this occurred, as island populations of black rats can be differentiated in the Galapagos (Willows-Munro, et al., 2016), genetic samples from the pre- and post-eradication attempt could be compared to help determine whether reintroduction or eradication failure occurred (Abdelkrim, et al., 2007).

Bait application

As with previous rodent eradications in the archipelago, bait application was timed for the last three months of the dry season (October–December), when rat breeding ceases and their numbers are at a minimum, after a typical six-month dry-spell (Clark, 1980). Bait type used was ‘Brodifacoum 25D Conservation’ (Bell Laboratories, Madison WI). Baits were 2.5 g compacted crushed grain pellets of 13 mm diameter, containing 25 µg (25 ppm) of brodifacoum per kg of bait, blue dye and pyranine biomarker, a non-toxic, odourless and tasteless dye that fluoresces green under UV light. Bait was applied in two aerial applications 23 days apart at an average rate of 6.72 kg/ha for the first application (15–17 November, 2012) and 4.85 kg/ha for the second application (8–9 December, 2012; Fig. 1). Pre-eradication trials in 2010 had determined that target application rates of 6 kg/ha followed by 3 kg/ha ensured bait was available in all habitats for at least four days. It had been planned to have bait applications 7–10

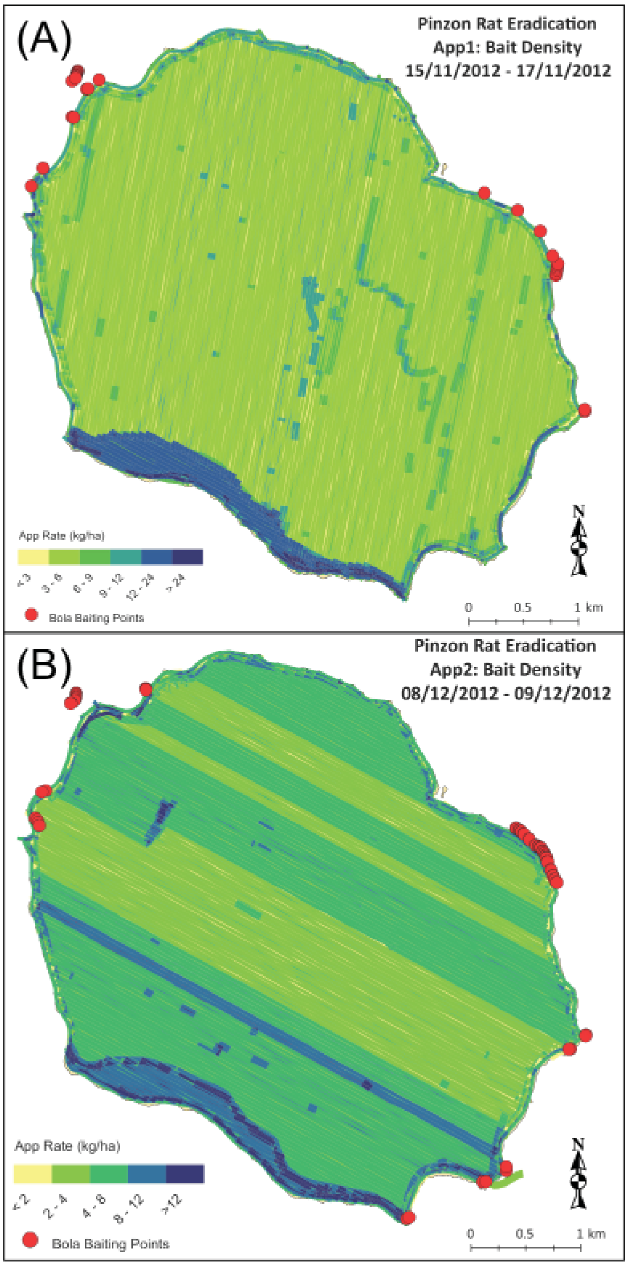

Fig.1 Bait density (kg/ha) maps of Pinzon Island from (A) first, and (B) second bait applications. Circles indicate where baits in paper bags were applied (bola baiting points).

days apart, however a pregnant rat was reported after the first bait application, prompting a decision to extend the duration between applications to maximise the probability that all rats would be exposed to bait (Keitt, et al., 2015).

An experienced pilot flew the helicopter (Eurocopter AS350-B2, France) guided by GPS, pre-programmed flight lines and light-bar (Tracmap flight unit, New Zealand). The helicopter was fitted with a custom agricultural style bait spreader bucket (CSI Helicopters, New Zealand) that was used to spread bait systematically over Pinzon Island (Fig. 1). Pre-programmed flight-lines 40 m apart provided a 100% overlap for inland areas, as previous bucket calibration indicated the bucket had an 80 m effective baiting swath width. Interior flight-lines ran coast-to-coast, with lines starting and ceasing 40 m inside of the coast to minimise the amount of bait entering the marine environment. Interior flight-lines were flown approximately north–south on the first application and east–west on the second. Two flight-lines were flown around the coast. The 'outer' coastal swath was flown along the coastline with a deflector attached to the bucket, providing 40 m unidirectional sowing towards the inland, to minimise bait entering the ocean. The inner coastal swath was carried out with the standard bucket, 60 m inland from the coast, thereby achieving a 50% overlap with the outer coastal swath. Sections of cliff over 50° slope on the southern side of the island were treated as a separate block to achieve twice the bait application rate of the interior, which is considered best practice (Broome, et al., 2014). GPS tracks were inspected periodically throughout each application. Any gaps identified in bait coverage were then baited in subsequent flights the same day.

Hand-baiting was conducted around the on-island camp, temporary hawk aviaries and one islet. The second islet was baited by hand from the helicopter using paper bags with 10 baits in each to achieve target application rates. Any areas along the coast that may not have received bait due to extreme steepness, overhangs, and deep cut gullies were also hand baited with bait in paper bags. Bait availability plots (25 m × 1 m; n=10) were used to monitor bait persistence after each aerial application at points from the coast to the highlands on the northern side of the island. Plots contained the number of pellets that corresponded to the bait application rate for each application. Each bait pellet was marked with a pin flag, which was removed as pellets were consumed. Plots were checked daily between the first and second applications and for 13 days following the second application.

Two boats acted as a floating base during helicopter baiting operations. One boat, fitted with a helicopter landing platform, also acted as the helicopter refuelling station. The second boat was fitted with a wooden platform from which bait was loaded into the bait spreader bucket as the helicopter hovered to one side.

Non-target species

Brodifacoum is the most commonly used toxicant for rodent eradications on islands and has the highest success rate (DIISE, 2016). Although an effective rodenticide, brodifacoum is highly toxic to mammals and birds, is known to persist in tissue containing vitamin K epoxide reductase (Eason, et al., 2002) and therefore presents risks to non-target wildlife through primary and secondary pathways of exposure (Broome, et al., 2015). Reptiles are considered to be less susceptible to brodifacoum (Weir, et al., 2015) but may also present a secondary exposure pathway to their predators.

An *a priori* non-target risk assessment which included Pinzon wildlife was conducted in 2010 (Campbell, 2010). A revised assessment (Fisher & Campbell, 2012) incorporated a suite of new information from the 2011 rodent eradications, and captive feeding trials used to assess risk of brodifacoum exposure in giant tortoises, lava lizards, geckos and snakes (Fisher, 2011a; Fisher, 2011b). Lava lizard samples were taken from Rabida Island before and after bait application to assess the incidence and persistence of residual brodifacoum in lava lizards but all these samples perished when a freezer was unplugged. Population-level impacts of brodifacoum applications were assessed for lava lizards and land birds using a before-after control-impact study design on Rabida, Bartolome, Bainbridge #3 and Beagle Sur islands, with Pinzon acting as a control. Based heavily upon the 2012 non-target risk assessment the Galapagos National Park Directorate and other partners determined that mitigation actions should be conducted for Pinzon tortoise, Galapagos hawk, Pinzon lava lizard, lava gull (*Larus fuliginosus*) and endemic land snails. Mitigation plans were developed for each taxon (Cunninghame, 2012; Cunninghame, et al., 2012; Oberg & Campbell, 2012; Parent & Campbell, 2012) except tortoises. Mitigation plans for lava gulls and land snails were not implemented. Lava gulls were not present on Pinzon Island during operations, and in searches undertaken before bait application all snails found were estivating so would not be exposed to bait.

Fifteen adult Pinzon tortoises were brought into captivity two years prior to baiting operations, housed on Santa Cruz Island and returned in good health two years after the rodent eradication was complete. Forty Pinzon lava lizards were taken into captivity prior to baiting and were maintained in enclosures on Pinzon Island. Termite larvae were provided as food every other day. Ten days after the second application the potential for bait consumption by lava lizards, as determined by bait degradation plots, was determined to be minimal and all surviving individuals were released near their capture sites.

Sixty hawks were taken into captivity on Pinzon Island, most prior to baiting operations, held in purpose-built aviaries and maintained on diets of goat meat, day-old chicks and (prior to baiting) rats. All hawks were ringed and genetic samples taken for future study. Four additional hawks were captured, ringed, and treated with injectable (intramuscular) vitamin K_1, however due to limited aviary space they were released. Three hawks were identified, but never captured. Captive hawks were released 12–14 days after the second aerial application of bait. Telemetry transmitters were fitted to 32 hawks before release.

Confirmation of eradication

Efficacy of rat detection methods was demonstrated prior to the eradication. Corrugated plastic chew cards with peanut butter (Oberg, et al., 2014), visual sightings, and signs of activity (prints, faeces, gnawed seed pods) readily indicated rodent presence across Pinzon Island. In January 2015 (25 months after the second bait application), these same methods were used to confirm black rat eradication with 1,140 chew cards deployed for at least 54 days, spaced at 25 m intervals along a trail network covering the island.

RESULTS

Baiting operations successfully applied bait across the island at the desired rates, as determined by helicopter GPS, baiting rate and effective swath width being overlaid on island maps (Fig. 1). Monitoring conducted more than two years after bait applications did not detect invasive rodents on Pinzon Island. None of the 1,140 chew cards deployed had rodent sign, while nearly 100% of chew cards placed pre-eradication did. Seed pods of *Acacia* spp. were intact and showed no sign of rodent damage across the island.

Based on this evidence we conclude that black rats were eradicated from Pinzon Island.

Bait availability plots indicated that after the first application (6.3 kg/ha) the average remaining density across all plots was above 2 kg/ha until day three (2.07 ± 1.75 kg/ha) and did not drop below 1 kg/ha until day 12 (0.9 ± 1.04 kg/ha). One plot had no bait available on day four; tortoises were observed consuming the baits. When the second application occurred, the average bait density remaining was approximately 0.5 ± 1.14 kg/ha. Bait availability plots for the second application (4.2 ± 1.19 kg/ha), indicated average availability remained above 2 kg/ha until day seven (2.7 ± 1.64 kg/ha), and less than 1 kg/ha remaining at day 12 (0.99 ± 1.67 kg/ha). Individual plots went to zero within two days due to Pinzon tortoises consuming bait.

Consumption of bait by Pinzon lava lizards and Pinzon tortoises was observed at higher rates than anticipated and evidenced by faeces containing blue dye, however no mortality in these species was observed in the wild. Additionally, mitigation efforts were successful at maintaining a separate population of Pinzon lava lizards and Pinzon tortoises as insurance in the case of any unexpected mortality in wild populations. Two lava lizards escaped captivity and five captive lizards died (survival rate of 87%).

All captive Galapagos hawks survived captivity and were released in healthy condition. Between 12 and 170 days after release, mortality of 22 tracked Galapagos hawks was recorded (Rueda, et al., 2016). Necropsy of four of these birds showed signs of anticoagulant toxicosis, with 379 ppb brodifacoum measured in one hawk liver (Rueda, et al., 2016). Monitoring of live-caught Pinzon lava lizards also showed residual brodifacoum in liver, for at least 850 days after bait application (Rueda, et al., 2016). The fate of 28 released hawks remains unknown, but they likely died. The remaining Pinzon Island Galapagos hawk population (n=10) was recaptured, placed into captivity in June 2013 and treated with Vitamin K_1, while toxicological monitoring of Pinzon lava lizards continued (Rueda, et al., 2016). These captive Galapagos hawks from Pinzon Island, representing 15% of the original population, were released when risk was considered acceptable in July and August 2016 with telemetry and GPS transmitters. Within three months of release, Galapagos hawks from Pinzon Island had nests with eggs. As of April 2018, nine nesting attempts have resulted in five healthy chicks, two nest failures, one unknown outcome and one pending (P. Castaño, unpublished data 2018). These and related events will be reported in greater detail elsewhere. Galapagos hawks continue to be monitored on Pinzon Island, as does toxicological monitoring in Pinzon lava lizards.

The eradication of black rats and actions taken to mitigate non-target impacts on Pinzon Island cost an estimated $1,501,000 (2013 US dollars). Cost breakdown estimates include planning ($101,000), implementation ($909,000), non-target species management ($101,000) and indirect costs ($390,000) (Holmes, et al., 2015).

DISCUSSION

Recovery of native and endemic species due to the successful eradication of invasive black rats from Pinzon is already evident, with ongoing monitoring expected to reveal further biodiversity gains. Pinzon tortoise hatchlings are now surviving in the wild for the first time in over 120 years (Tapia Aguilera, et al., 2015). With natural recruitment now occurring the Pinzon tortoise head-starting may soon no longer be required (Jensen, et al., 2015). Land-bird surveys in early 2018 found two species (cactus finch *Geospiza scandens*, Galapagos rail *Laterallus spilonota*) never before recorded from the island (Fessl, et al., unpublished data 2018). Endemic land snails also appear to be on the increase, indeed a new species of land snail was discovered two years post-eradication in permanent snail monitoring plots on the island and is currently being described (C. Parent, unpublished data 2015). With a major threat now removed, threatened land snails and other species may now be eligible for down-listing from the IUCN Red List. Similarly, on Rabida Island, two years after invasive Norway rats (*Rattus norvegicus*) were eradicated, two island endemic land snails that were considered extinct for over 100 years were rediscovered (Campbell, et al., 2013; C. Parent, unpublished data 2012). Also, on Rabida Island, a leaf-toed gecko was found post-eradication in late 2012 (Campbell, et al., 2013). The only known geckos from Rabida were recorded from subfossils estimated at more than 5,700 years old, which were classified to genus only (Steadman, et al., 1991). Although the specimen was identified at the time as the archipelago endemic *Phyllodactylus galapagensis* (W. Tapia Aguilera pers. comm. 2013), a recently proposed taxonomic split divides *P. galapagensis* into four species by major islands, including Pinzon (Torres-Carvajal, et al., 2014). Future analyses including samples of geckos from Rabida Island may also see a unique species identified for that island.

Eradication of black rats from Pinzon Island was arguably a cost-effective conservation action at US$827 / ha, resulting in the removal of a significant threat for at least 15 Pinzon Island endemic species, several archipelago endemic species and 12 IUCN threatened species. The negative impact of this conservation action on Pinzon's population of Galapagos hawks is expected to be short-term, with breeding already underway on the island. However, without additional mitigation actions this population may have been lost due to secondary poisoning, potentially requiring a translocation to re-establish Galapagos hawks on Pinzon Island. Longer-term impacts will only be discovered in time.

The persistence of brodifacoum residues in Pinzon lava lizards for at least 850 days was unexpected (Rueda, et al., 2016) and, as it was unknown at the time, was not considered within *a priori* risk assessments. Ingestion of lizards carrying residual brodifacoum for prolonged periods was likely a significant contributor to unpredicted and unexpectedly high mortality of released Galapagos hawks (Rueda, et al., 2016). The use of a prescribed duration for captive holding was, in hindsight, an error. Future mitigation efforts should use biological criteria (e.g. bait availability, sentinel animals) relevant to the pathways being managed to determine when captive held or translocated non-target species be released after brodifacoum bait has been used for rodent eradication.

Pinzon is currently the largest island in the Galapagos to be freed of invasive rodents, and the fourth-largest island globally to be eradicated of black rats (behind Macquarie, Rangitoto and Australia Islands; DIISE, 2016). Continuing on, as suggested in the original roadmap (CDF & GNPS, 2007), the next island in the Galapagos archipelago being targeted for rodent eradication is Floreana Island, nearly an order of magnitude larger than Pinzon, with 160 human inhabitants, pets, livestock, surface water and a suite of wildlife species that are expected to require mitigation actions (Island Conservation, 2013). Floreana Island represents significant challenges but also major opportunities for incorporating social well-being targets in invasive species eradication projects, as well as biodiversity targets to benefit 55 IUCN threatened species and creating the conditions for the reintroduction of 13 species extirpated by invasive species.

Removing non-native invasive rodents from islands is a proven approach to protecting endemic biodiversity (Jones, et al., 2016), and anticoagulant rodenticides are currently the most reliably effective method to achieve this. Until alternative rodent-specific methods become available (Campbell, et al., 2015), practitioners will have to become increasingly skilled at mitigating risks to non-target species related to rodent eradications to ensure the conservation benefits of this powerful tool are maximised.

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REFERENCES

Abdelkrim, J., Pascal, M. and Samadi, S. (2007). 'Establishing causes of eradication failure based on genetics: Case study of ship rat eradication in Ste. Anne Archipelago'. *Conservation Biology* 21: 719–730.

Beck, R. (1903). 'In the Home of the Giant Tortoise'. *Annual Report of the New York Zoological Society* 7:160–174.

Bellard, C., Cassey, P. and Blackburn, T.M. (2016). 'Alien species as a driver of recent extinctions'. *Biology Letters* 12: 20150623.

Broome, K., Cox, A., Golding, C., Cromarty, P., Bell, P. and McClelland, P. (2014). *Rat Eradication Using Aerial Baiting: Current Agreed Best Practice Used in New Zealand (V3.0).* New Zealand Department of Conservation internal document DOCDM-839096. Wellington, New Zealand: Department of Conservation.

Broome, K.G., Fairweather, A.A.C. and Fisher, P. (2015). *Brodifacoum Pesticide Information Review. Version 2015/1*. Docdm-25436. Hamilton, New Zealand: Department of Conservation.

Campbell, K. (2010). *Non-Target Risk Assessment for Rodenticide Applications in the Galapagos Islands*. Unpublished report. Puerto Ayora, Galapagos Islands: Island Conservation and Charles Darwin Foundation.

Campbell, K.J., Carrion, V. and Sevilla, C. (2013). 'Increasing the Scale of Successful Invasive Rodent Eradications in the Galápagos Islands'. In: L.J. Cayot, D. Cruz and R. Knab (eds.) *Galapagos Report 2011–2012,* pp. 194–198. Puerto Ayora, Galapagos, Ecuador: GNPS, GCREG, CDF and GC.

Campbell, K.J., Beek, J., Eason, C.T., Glen, A.S., Godwin, J., Gould, F., Holmes, N.D., Howald, G.R., Madden, F.M., Ponder, J.B., Threadgill, D.W., Wegmann, A. and Baxter, G.S. (2015). 'The next generation of rodent eradications: Innovative technologies and tools to improve species specificity and increase their feasibility on islands'. *Biological Conservation* 185: 47–58.

Carrion, V., Donlan, C.J., Campbell, K., Lavoie, C. and Cruz, F. (2007). 'Feral donkey (*Equus asinus*) eradications in the Galápagos'. *Biodiversity and Conservation* 16: 437–445.

Carrion, V., Donlan, C.J., Campbell, K.J., Lavoie, C. and Cruz, F. (2011). 'Archipelago-wide island restoration in the Galápagos Islands: Reducing costs of invasive mammal eradication programs and reinvasion risk'. *PLOS ONE* 6: e18835.

Cayot, L.J., Rassmann, K. and Trillmich, F. (1994). 'Are marine iguanas endangered on islands with introduced predators'. *Noticias de Galápagos* 53: 13–15.

Cayot, L.J., Muñoz, E. and Murillo, G. (1996). 'Conservation of the tortoises of Pinzón Island'. *Charles Darwin Research Station Annual Report* 1988–1989: 29–35.

CDF and GNPS (2007). *Project Pinzon: Ecosystem Restoration of the Galapagos Islands through Introduced Rodent Removal. April 2–12, 2007.* Puerto Ayora, Galápagos: Fundación Charles Darwin / Servicio Parque Nacional Galápagos.

Clark, D.A. (1981). 'Foraging patterns of black rats across a desert-montane forest gradient in the Galapagos Islands'. *Biotropica* 13: 182–194.

Clark, D.B. (1980). 'Population ecology of *Rattus rattus* across a desert-montane forest gradient in the Galapagos Islands'. *Ecology* 61: 1422–1433.

Cruz, F., Donlan, C.J., Campbell, K. and Carrion, V. (2005). 'Conservation action in the Galápagos: Feral pig (*Sus scrofa*) eradication from Santiago Island'. *Biological Conservation* 121: 473–478.

Cruz, J.B. and Cruz, F. (1987). 'Conservation of the dark-rumped petrel *Pterodroma phaeopygia* in the Galapagos Islands, Ecuador'. *Biological Conservation* 42: 303–311.

Cunninghame, F. (2012). *Lava Gull (*Leucophaeus fuliginosus*) Mitigation During an Aerial Broadcast of Brodifacoum Cereal Baits on Plaza Sur Island, Galapagos*. Unpublished report. Puerto Ayora, Galapagos Islands: Charles Darwin Foundation.

Cunninghame, F., Ponder, J. and Campbell, K. (2012). *Galápagos Hawk (*Buteo galapagoensis*) Mitigation During an Aerial Broadcast of Brodifacoum Cereal Baits on Pinzón Island, Galápagos.* Unpublished report. St Paul, Minnesota: The Raptor Center.

DIISE. (2016). 'The Database of Island Invasive Species Eradications, Developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand'. <http://diise.islandconservation.org>. Accessed 16 February 2016.

Doherty, T.S., Glen, A.S., Nimmo, D.G., Ritchie, E.G. and Dickman, C.R. (2016). 'Invasive predators and global biodiversity loss'. *Proceedings of the National Academy of Sciences*: 201602480.

Dowler, R.C., Carroll, D.S. and Edwards, C.W. (2000). 'Rediscovery of rodents (Genus *Nesoryzomys*) considered extinct in the Galapagos Islands'. *Oryx* 34: 109–117.

Eason, C.T., Murphy, E.C., Wright, G.R.G. and Spurr, E.B. (2002). 'Assessment of risks of brodifacoum to non-target birds and mammals in New Zealand'. *Ecotoxicology* 11: 35–48.

Fisher, P. (2011a). *Non-Target Risk to Galápagos Reptiles from Application of Brodifacoum Bait for Rodent Eradication: Captive Feeding Trials.* Unpublished report. Lincoln, New Zealand: Landcare Research.

Fisher, P. (2011b). *Non-Target Risk to Galápagos Tortoises from Application of Brodifacoum Bait for Rodent Eradication: Captive Feeding Trials*. Unpublished report. Lincoln, New Zealand: Landcare Research.

Fisher, P. and Campbell, K. (2012). *Non-Target Risk Assessment for Rodenticide Application on Pinzón and Plaza Sur, Galápagos Islands*. Unpublished report. Lincoln, New Zealand: Landcare Research.

Fonseca, G.A.B.d., Mittermeier, R.A. and Mittermeier, C.G. (2006). *Conservation of Island Biodiversity: Importance, Challenges, and Opportunities,* Washington, DC: Center for Applied Biodiversity Science at Conservation International.

Fukami, T., Wardle, D.A., Bellingham, P.J., Mulder, C.P.H., Towns, D.R., Yeates, G.W., Bonner, K.I., Durrett, M.S., Grant-Hoffman, M.N. and Williamson, W.M. (2006). 'Above- and below-ground impacts of introduced predators in seabird-dominated island ecosystems'. *Ecology Letters* 9: 1299–1307.

Harper, G.A. and Carrion, V. (2011). 'Introduced rodents in the galápagos: colonisation, removal and the future'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 63–66. Occasional paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Harper, G.A., Zabala, J. and Carrion, V. (2011). 'Monitoring of a population of Galápagos land iguanas (*Conolophus subcristatus*) during a rat eradication using brodifacoum'. In*:* C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 309–312. Occasional paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Holmes, N.D., Campbell, K.J., Keitt, B.S., Griffiths, R., Beek, J., Donlan, C.J. and Broome, K.G. (2015). 'Reporting costs for invasive vertebrate eradications'. *Biological Invasions* 17: 2913–2925.

Island Conservation (2013). *Floreana Island Ecological Restoration: Rodent and Cat Eradication Feasibility Analysis V.6.1. (English and Spanish).* Unpublished report. Santa Cruz, California: Island Conservation.

IUCN (2015). 'The IUCN Red List of Threatened Species. Version 2015–4'. <http://www.iucnredlist.org>. Accessed 19 November 2015.

Jensen, E.L., Tapia, W., Caccone, A. and Russello, M.A. (2015). 'Genetics of a head-start program to guide conservation of an endangered Galápagos tortoise (*Chelonoidis ephippium*)'. *Conservation Genetics* 16: 823–832.

Jones, H.P., Tershy, B.R., Zavaleta, E.S., Croll, D.A., Keitt, B.S., Finklestein, M.E. and Howald, G.R. (2008). 'Severity of the effects of invasive rats on seabirds: A global review'. *Conservation Biology* 22: 16–26.

Jones, H.P., Holmes, N.D., Butchart, S.H.M., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P., Cuthbert, R.J., Ebbert, S., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R. and Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences* 113: 4033–4038.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

Kurle, C.M., Croll, D.A. and Tershy, B.R. (2008). 'Introduced rats indirectly change marine rocky intertidal communities from algae- to invertebrate-dominated'. *Proceedings of the National Academy of Sciences* 105: 3800–3804.

MacFarland, C.G., Villa, J. and Toro, B. (1974). 'The Galápagos giant tortoises (*Geochelone elephantopus*) part I: Status of the surviving populations'. *Biological Conservation* 6: 118–133.

Morillo Manrique, G.E. (1992). 'Estado Poblacional y Ecología Reproductiva de la Tortuga Gigante de la Isla Pinzón (*Geochelone elephantopus ephippium*), Galápagos'. Undergraduate degree in biological sciences thesis. Quito, Ecuador: Pontificia Universidad Católica del Ecuador.

Muñoz, E. (1990). 'Seguimiento de la Campaña de Erradicación de Ratas en la Isla Pinzón'. *Carta Informativa (Estacion Cientifica Charles Darwin, Servicio Parque Nacional Galápagos)* 30: 2.

Oberg, E. and Campbell, K. (2012). *Pinzon Lava Lizard Mitigation Plan: Protocols for Capture and Captive Holding.* Unpublished report. Puerto Ayora, Galapagos Islands: Island Conservation.

Oberg, E., Alifano, A., Pott, M. and Wegmann, A. (2014). 'Detecting Rodents in the Presence of Land Crabs: Indicator Blocks Outperform Standard Rodent Detection Devices at Palmyra Atoll'. In*:* R.M. Timm and J.M. O'Brien (eds.) *Proceedings of the 26th Vertebrate Pest Control Conference*, pp. 370–374. Davis, Califorina: University of California, Davis.

Parent, C. and Campbell, K. (2012). *Land Snail Mitigation Plan for Pinzón Island During the Application of Brodifacoum Bait for Rodent Eradication*. Unpublished report. Berkley, California: University of California, Berkley and Island Conservation.

Patton, J.L., Yang, S.Y. and Myers, P. (1975). 'Genetic and morphologic divergence among introduced rat populations (*Rattus rattus*) of the Galapagos Archipelago, Ecuador'. *Systematic Biology* 24: 296–310.

Rueda, D., Campbell, K.J., Fisher, P., Cunninghame, F. and Ponder, J.B. (2016). 'Biologically significant residual persistence of brodifacoum in reptiles following invasive rodent eradication, Galapagos Islands, Ecuador'. *Conservation Evidence* 13: 38.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Steadman, D., Stafford, T., Donahue, D. and Jull, A. (1991). 'Chronology of Holocene vertebrate extinction in the Galápagos Islands'. *Quaternary Research* 36: 126–133.

Tapia Aguilera, W., Málaga, J. and Gibbs, J.P. (2015). 'Giant tortoises hatch on Galapagos Island'. *Nature* 517: 271.

TIB Partners. (2014). 'The Threatened Island Biodiversity Database (TIB): Developed by Island Conservation, University of California Santa Cruz Coastal Conservation Action Lab, Birdlife International and IUCN Invasive Species Specialist Group. Version 2014.1'. <www.tib.islandconservation.org>. Accessed 23 December 2016.

Torres-Carvajal, O., Torres-Carvajal, O., Barnes, C.W., Pozo-Andrade, M.J. and Tapia, W. (2014). 'Older than the islands: Origin and diversification of Galápagos leaf-toed geckos (Phyllodactylidae: *Phyllodactylus*) by multiple colonizations'. *Journal of Biogeography* 41: 1883–1894.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?' *Biological Invasions* 8: 863–891.

Weir, S.M., Yu, S., Talent, L.G., Maul, J.D., Anderson, T.A. and Salice, C.J. (2015). 'Improving reptile ecological risk assessment: Oral and dermal toxicity of pesticides to a common lizard species (*Sceloporus occidentalis*)'. *Environmental Toxicology and Chemistry* 34: 1778–1786.

Willows-Munro, S., Dowler, R.C., Jarcho, M.R., Phillips, R.B., Snell, H.L., Wilbert, T.R. and Edwards, C.W. (2016). 'Cryptic diversity in black rats *Rattus rattus* of the Galápagos Islands, Ecuador'. *Ecology and Evolution* 6: 3721–3733.

First results from a pilot programme for the eradication of beavers for environmental restoration in Tierra Del Fuego

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Abstract A pilot project for the eradication of beavers (*Castor canadensis*) in Tierra del Fuego started as part of a bi-national agreement, signed between Argentina and Chile, to restore the affected environments. The project covers nine pilot areas of different landscapes and land tenures in the Argentinian part of Isla Grande de Tierra del Fuego. We report on the results from operations in the first of the pilot areas. From October 2016 to January 2017, ten trappers (named restorers for advocacy purposes) used body-grip traps, snares and an air rifle, in a first phase, which included 2,237 trapping nights and 1,168 trap-sets. Shooting efforts were not monitored. Traps were set for 1,401 trapping-nights and caught 175 beavers at a success rate of 12.5% (captures per trap night). Snares were set for 936 snare-nights and caught 22 beavers at a success rate of 2.3%. Seven beavers were shot. Most beavers (65%) were removed during the first week of trapping in the different watercourse sections. Stopping trapping for a week or more did not increase efficiency. From March to May 2017 restorers removed 24 survivors and/or reinvaders, including 10 from two previously untrapped colonies. Capture efficiency for this removal period was low for body-gripping traps but not for snares. The sex ratio of catches was 47% females to 53% males. The age structure of catches was 15% kits, 29% yearlings, 51% adults, with 4% not aged. An estimated total of 41 colonies was trapped, giving an average of 5.6 animals per colony. After nominal eradication was declared by restorers, 154 camera trapping nights were deployed to assess eradication success. Nine cameras (of 26 cameras used) detected beavers. Therefore, eradication was not achieved using the methods and efforts in the first part of the pilot study. This highlights the need for more effort or the application of different techniques or trapping strategies. For example, daily checking of traps may cause the animals to be cautious so, the next step in the programme will involve exploring alternative trapping methods to reduce disturbance.

Keywords: Argentina, *Castor canadensis*, eradication programme, management, pilot study, trapping

INTRODUCTION

North American beavers (*Castor canadensis*) are semi-aquatic and territorial rodents. They live in family groups generally composed of two breeding adults, two yearlings and two kits; the yearlings are forced to leave the natal colony by the age of two (Lizarralde & Escobar, 1997; McTaggart & Nelson, 2003). The family group controls a group of adjacent dams, defending its territory from other beavers. Each family group can build one or more lodges (although they may also den in the river banks) and share a single food cache.

In 1946, 20 beavers were introduced from Canada to Tierra del Fuego, South America (Pietrek & Fasola, 2014), with the aim of developing a fur industry. Beavers found extensive suitable habitats, high availability of food, lack of predators and unoccupied territory (Lizarralde, 2004). These features allowed beavers to spread quickly throughout Tierra del Fuego (Skewes, et al., 2006; Anderson, et al., 2009). Several impacts on the environment of Tierra del Fuego were reported and it was suggested that beavers caused the largest landscape-level alteration to the region since the Holocene (Anderson, et al., 2009). The most obvious impacts are the reduction of the riparian vegetation due to their activities, which includes the building of at least 70,000 dams in Argentinian Tierra del Fuego (Eljall, et al., 2016), affecting at least 31,000 ha of forests, grasslands and peat bogs (Henn, et al., 2016), as well as the fen areas (Westbrook, et al., 2017). The beech forests of Tierra del Fuego are not adapted to the impact of beavers, so their impacts are long lasting (Anderson, et al., 2009). Their dams also limit the dispersal of native fish and the water in their dams changes the benthic communities, modifying the macroinvertebrate assemblages by engineering changes to the fluvial and riparian environment (Anderson, et al., 2006). Beavers also modify the dynamics of the streams by altering sedimentation (Vazquez, 2002; Martin, et al., 2015). Last, but no less important, beavers impact the economy by flooding roads and culverts, and affecting ranching activity, reducing pastures by flooding as well as affecting fences.

Attempts to control beavers by commercial hunting during the 1990s and 2000s failed. Beavers were detected in continental South America in the 1990s (Skewes, et al., 2006; Wallem, et al., 2007; Schiavini, et al., 2008; Anderson, et al., 2009), although recent dendrochronological evidence takes their arrival date to 1968 (Graells, et al., 2015). The presence of beavers in the continent raised alarm about the possibility of their dispersal through the greater American continent. In view of these issues, Argentina and Chile started, in 2005, to discuss a change in strategy.

Eradication was deemed as feasible (Parkes, et al., 2008), and adopted as a strategy by Argentina and Chile in 2008, after signing a bi-national agreement for the restoration of the southern ecosystems affected by the beaver (Malmierca, et al., 2011). At present, both countries are performing pilot projects, funded by the Global Environment Facility (GEF) and national counterparts. The pilot project in Argentina is under the umbrella of the major project "Strengthening the Governance for the Protection of Biodiversity through Formulation and Implementation of the National Strategy for Invasive Exotic Species" GEF Project ID 4768. The project runs from 2015 to 2019, covering nine pilot areas of Tierra del Fuego.

The objectives of the project (Schiavini, et al., 2016) are essentially to answer questions raised during the feasibility study: building capacity, learning about technical and organisational challenges of the process, showing the environmental benefits of beaver removal, and deciding the next steps between the two countries.

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Several research priorities and questions in relation to the eradication of beavers are expected to be answered by the pilot project:

- How much effort is needed to eradicate beavers and to declare eradication on a small scale?
- What factors affect effectivity of trapping? The tools used? The sequence of deployment? Learning by beavers to avoid traps?
- What is the effort demanded for active surveillance to avoid reinvasion?
- How to develop passive surveillance from society?
- Is the bureaucracy able to accommodate the dynamics of eradication projects?
- Are any beavers found, after nominal eradication is declared, likely to be survivors or reinvaders?
- Does the environment recover in a short time frame after beaver removal?

The nine Argentinian pilot areas cover an area of 1,017 km^2, with a range of 14–238 km^2 (Fig. 1). In this paper, we report the results of operations achieved in the first pilot area, Esmeralda-Lasifashaj, and discuss the challenges revealed for the larger major project.

MATERIALS AND METHODS

The Esmeralda-Lasifashaj area (54 km^2) belongs to the ecological region of the forest range (Collado, 2007). The landscape represents a U-shaped valley with the valley bottom covered with *Sphagnum* peat bogs and poorly drained mires (Figs 2 and 3). Slopes are covered with southern beech forests (*Nothofagus* spp.) with the vegetation line reaching about 700 m altitude. The main valley is surrounded by eight lateral valleys. The area is open to reinvasion as it has no geographical boundaries that limit beaver dispersal, mainly from the west and east. However, it was proposed as a pilot area for several reasons: it is located only 20 km from Ushuaia city, is

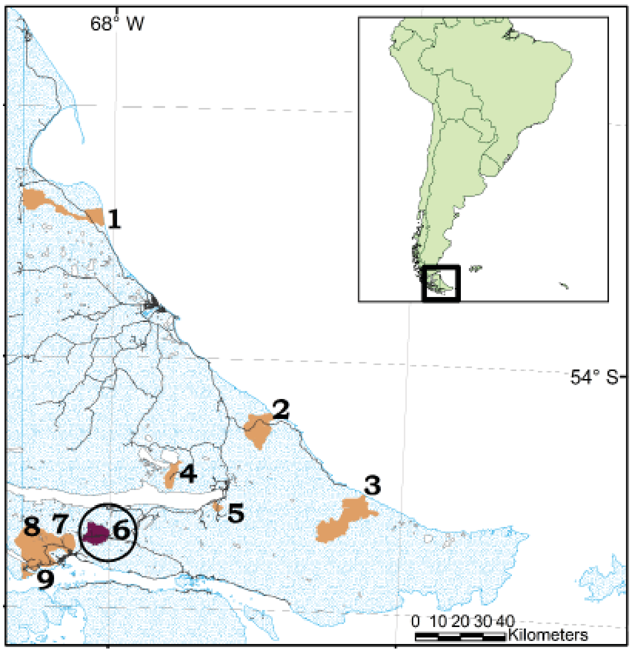

Fig. 1 The Argentinian sector of Isla Grande de Tierra del Fuego. Numbers refer to each pilot area. 1: Arroyo Gamma. 2: Arroyo Asturiana. 3: Rio Malengüena. 4: Río Mimica. 5: Arroyo Indio. 6: Esmeralda-Lasifashaj. 7: Arroyo Grande, 8: Rio Pipo, 9: south of Tierra del Fuego National Park. The black circle shows the location of the pilot area Esmeralda-Lasifashaj.

Fig. 2 An aerial view of a series of beaver dams in the bottom of the main valley of Esmeralda-Lasifashaj pilot area.

Fig. 3 An aerial view of a series of beaver dams in the Esmeralda-Lasifashaj pilot area, in an area of poor drainage at the contact between peat bogs and forest. Note the riparian forest impacted by cutting.

used by the public for recreation and tourism, and the area holds a permanent cross-country ski trail, which is affected by beavers. For these reasons, the area was selected as a way of showing the environmental, social and economic benefits of removing beavers.

The dams and lodges built by beavers are so conspicuous that they can be identified in satellite images. During the planning process, beaver dams and lodges were mapped using Google Earth and integrated with the dams identified by Eljall, et al. (2016). Then, 363 locations of beaver activity were loaded into the GPS units used during the operation (Garmin eTrex 20x), to be used as a general guide for moving through the terrain to the areas impacted by beavers.

The skills of the personnel involved in hunting should include not only good trapping skills, but also the ability to spend several days in the field in the harsh weather of Tierra del Fuego and deliver good trapping data, essential for assessing trapping efforts and eradication success. Good, traditional trappers work with a focus on yield, while personnel needed for eradication need to "look for the last animal". With this change in focus, 10 people were selected and trained from a group of 39 people interviewed. The training was performed by our own personnel, staff from the National Parks Administration and from the volunteer fire brigade. Training included the use of trapping tools, data recording and first aid in the field. The final selection included a combination of people with previous trapping skills and people with good outdoor abilities and a willingness to learn. Hunters are publicly called "restorers" as a way of helping to advocate for the final objective of the project, i.e. building the correct conditions for environmental restoration by means of beaver eradication.

The trapping equipment and tools were purchased with advice from the Animal and Plant Health Inspection Service of the USA, who also provided a handbook for best-practice management. Two main tools are being tested, body-gripping traps and non-powered cable devices (snares), complemented with a PCP air rifle. The group was commanded by a chief of operations and assisted by a logistics officer.

The spatial and temporal progression of trapping differs from traditional trapping operations, where hunters deploy their tools progressively through the landscape, usually in a regular or grid mode. Given that the trapping target is located along watercourses or sectors of poor drainage such as edges of peatlands, trapping effort follows these landscape features. For planning purposes, the pilot area was divided into sectors that brought together groups of sections of channel or activity detected during planning. Watercourse sections were trapped inside sectors until "nominal" eradication was achieved, when trappers moved to another watercourse section. After nominal eradication of a sector, operations progressed to another sector.

At the watercourse section scale, trapping was made according to decisions made by each restorer. A "trap-set" is a trap (either a body-gripping trap or a snare) set at a particular location and for a number of consecutive trapping nights. Traps are usually set along watercourses and near dams with beaver activity denoted by the girdling of trees, fresh beaver trails, freshly gnawed branches in front of the dams, castor mounds, and /or accumulation of submerged tree branches with leaves. Traps are also set either in trails or slides made by beavers or in purpose-made openings at the front of the dam. The limits of beaver colonies are not always evident. However, during fall and winter, family groups gather at one lodge, so colonies are more easily distinguishable. During spring and summer, young animals disperse from their natal colonies, so the movement of animals leads to colony boundaries being confused. Also, traps can be set in the same place for more than one night. After a number of trapping nights, hunters noticed a reduction in their trapping efficiency, and at some point, they decided that a "nominal" eradication was achieved in this watercourse section and moved to another section. As a result, data recording is quite different from some other hunting and trapping operations, where hunters either traverse a landscape searching for their prey, or traps are set up more permanently at sites or along transects or grids.

The records of trapping and yields attempted to reflect the operation in great detail. An account of each trap set and its subsequent outcome (set, capture, activation without capture, not activated, removed) was recorded every day, taking into account the use of both the body-gripping traps and snares, with each one requiring daily checks for humanitarian reasons. Each trap had a unique number for identification. For data recording, an application was built into Cybertracker software (Steventon, 2017), allowing us to build a database with a record of each trap (set, revision and retirement, with or without capture), as well as ancillary data (e.g. location of placement, use of attractant). The application is available upon request, or at <http://cybertrackerwiki.org/index.php?title=Community_applications>. For data recording, we used an outdoor rugged tablet (Boolean A71, Boreal Technologies Inc). The database can be transferred to Spreadsheets or to any GIS system, as Cybertracker software can export shapefiles. Restorers also carried a GPS unit for tracking their activity.

Operations ran from October 24, 2016 to January 31, 2017 in the first phase. From March 2 to May 15, 2017 (Fig. 4), the area was checked again to remove survivors/invaders. Restorers worked mostly daily, during blocks of five days or four trapping nights, commuting each day from Ushuaia to the pilot area that is traversed by a National Route highway. When restorers worked on the lateral valleys, they camped for between three and five days. A Robinson R44 helicopter was used to search for dams in specific areas (Johnston & Windels, 2015) and to transport personnel and equipment to lateral valleys. Two colonies were left untrapped until the survivors-reinvaders removal phase, as they were used by tourist operators during the summer. Tour operators agreed as this would be the last time they would be using these colonies for their tours.

Trapped animals were aged in the field, based on external measurements, as kits, yearlings or adults, and were sexed by detection of the baculum. Samples were stored for accurate age determination, the breeding status of females and for future assessment of the accuracy of genetic tools to distinguish survivors from new invaders in areas free of beavers.

For verification of eradication, an independent team visited a sample of the watercourse sections, as restorers declared the "nominal" eradication, between December 12, 2016, and May 24, 2017. Twenty-six camera traps were set in front of artificial castor mounds with beaver lure at a 1–2m distance from the camera and no more than 1m from the water body. Each camera was placed at a height of between 20 and 40 cm from the ground to capture full images of beavers, and operated, on average, six days, with a range of 3–10 days. Cameras were located both in the main valley and in all the lateral valleys.

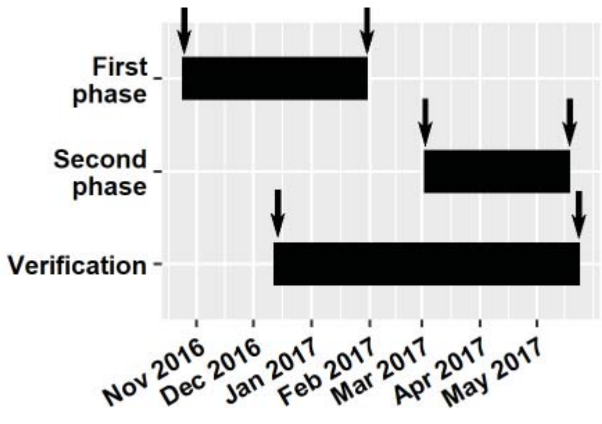

Fig. 4 Gantt chart including the first and second eradication step (the arrows mark the first and last capture) and the period of camera-trap vigilance (the arrows mark the first and the last detection).

As operations took place during spring and summer, territorial limits were difficult to assess. The total number of colonies was estimated based on the spatial distribution of catches following Johnston & Windels (2015). The Esmeralda-Lasifashaj area was divided into 18 sectors for data analysis. All statistical analysis was performed in Infostat (Rienzo, et al., 2016).

A monitoring plan measuring the environmental benefits of removal of the beavers is being developed by independent groups. The monitoring includes assessment of the of trees that will not be subject to beaver cutting after beaver removal, water quality, macroinvertebrate diversity, metabolism of the watercourse and fish diversity.

RESULTS

Mop-up phase

From October 2016 to January 2017, restorers walked 2,930 km over the area (Fig. 5). For logistic purposes, a helicopter was flown for nine hours. Trapping nights were derived from trapping records by summing up trap revisions and retirements. An additional 5% was added to the effort for the offset for traps that were not checked daily (based on an analysis of a subset of data).

Body-gripping traps were deployed in 715 trap sets, yielding 1,401 trapping nights. Snare traps were deployed in 453 trap sets, yielding 936 trapping nights. This represents a total of 2,337 trapping nights with 1,168 sets. Each trap operated on average 1.97 nights with a range of one to four nights. Rifle effort was not monitored, as it was employed in an opportunistic fashion. A total of 197 beavers were removed by trapping; 175 with body-gripping traps and 22 with snares, together with seven individuals that were shot (Fig. 6). The trapping efficiency was 12.5% for body-gripping traps and 2.3% for snares, giving an average efficiency of 9% for trapping.

The capture efficiency for each day of the working blocks was assessed. For example: during the first day of the working block the main activity was setting traps; during the second day of the working block there were 443 reviews or removals and 46 catches, which gives an efficiency of 10.4%; on the sixth or seventh day, very little field work was performed. This analysis was then limited to reviews and retirement of traps from Tuesday to Friday. Using a test of more than two proportions (Zar, 2010), the null hypothesis of the difference of proportions revealed no differences in catch efficiency over the different days of the week (χ^2 statistic, p=0.152, df=3). Therefore, restorers did not reduce trapping efficiency through cumulative disturbance by working consecutive days in a watercourse segment, since the efficiency was similar between the days of the working block. Another explanation might be that even though beavers are more "relaxed" or “naïve” to trapping early in the week (i.e. Tuesdays), restorers gradually perform better in a particular area during the week, compensating for the increasing caution of beavers with improved trapping sets.

The effect of disturbance from hunting over the weeks was also assessed, checking if leaving a section of the watercourse without trapping for a week after trapping for one or two weeks increases the trapping efficiency by reducing the awareness of traps by the beavers. The scarce data available for this analysis revealed no positive effect by leaving a watercourse section without traps. The first week of trapping in the watercourse´s section yielded 65% of the beavers, giving an average capture efficiency higher than the efficiency of the rest of the trapping days (10.3% and 9% respectively; p <0.0001, difference of proportions of Infostat). The capture efficiency did not differ between the main valleys and the lateral valleys, comparing the 10 channel sections of the main valley with the six channel sections of the lateral valleys (p=0.88).

A total of 151 traps (289 trapping nights) were set with attractant (beaver hormone, food lure): 142 traps (263 trapping nights) set with attractant, six traps (13 trap nights) with attractant added after the first review and three traps (13 trap nights) with attractant added after the second review. These 151 traps produced 13 catches (289 trap nights), giving an efficiency of 4.5%. If only beaver lure was considered, there were 10 catches in 89 traps (163 trapping nights), giving an efficiency of 6.1%.

The sex ratio of catches did not differ from 1:1 (p=0.26, 45% females vs 52% males, 3% unsexed). Also, the

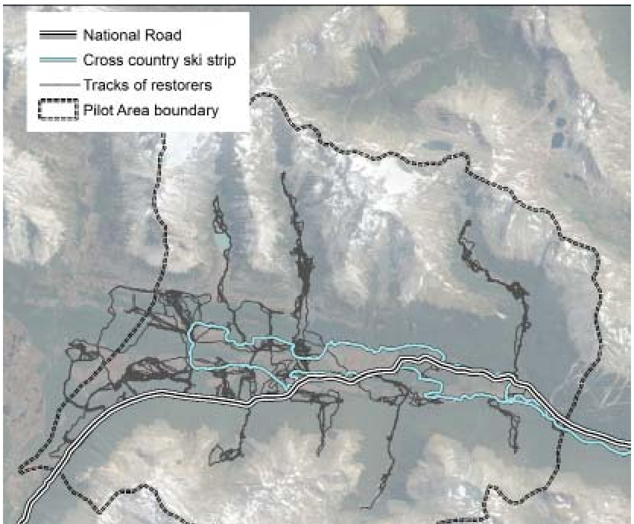

Fig. 5 Tracks recorded by restorers in the pilot area Esmeralda-Lasifashaj. Some tracks were not recorded due to failure of the GPS units.

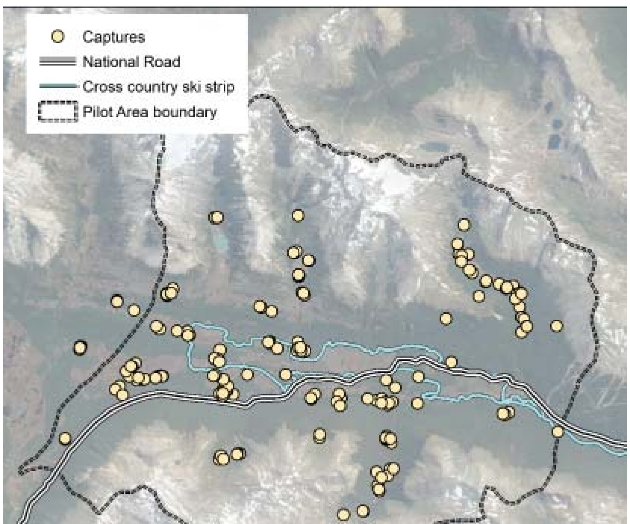

Fig. 6 Catches recorded for the pilot area Esmeralda-Lasifashaj

proportion of females did not differ between body-gripping and snares (difference of proportions = 0.003; p≈1). The age assessment made by restorers revealed an age structure of 15% kits, 29% yearlings and 51% adults (4% not aged), with similar proportions of age classes between body-gripping and snare traps (p=0.21). During the first days of trapping along each watercourse section, 83% of the sections yielded females while 50% of them yielded males (marginally significant difference, p=0.043).

Survivors/reinvaders removal phase

During this phase, restorers walked 380 km deploying 735 trap nights (529 body-gripping and 206 snares). This represented 23% and 31% of the previous walking and trapping effort respectively. Twenty-four animals were removed (22 with body-gripping traps and two with snares, Fig. 6). From them, 10 animals came from the two colonies left untrapped during the mop-up phase, and therefore 14 animals should be considered survivors-reinvaders. The main valley provided most of the captures (83%), although most of the trapping effort was focused there (83%).

Capture efficiency was 3.97% for body-gripping traps and 0.97% for snares. Trapping efficiency, compared with the first phase, was lower for body-gripping traps (p<0.0001), but not for snares (p=0.20).

One of the two colonies originally left untrapped yielded six males (one adult, four juveniles and one kit), one female and one animal of unidentified sex. The second colony yielded two males (one juvenile and one kit) and two females (one adult and one kit).

The survivors/reinvaders captured consisted of 10 males (six adults, three juveniles and one kit), six females (four adults and two juveniles) and two animals of unidentified sex. Five sites provided only males in this phase (including a site with only three males). Attractant was used in only seven of the sets, therefore the outcome was not analysed due to the low sample size.

Population assessment

Analysis of the spatial distribution of catches concluded that 41 colonies were trapped (plus a few recolonised sites). The average number of beavers per colony was 5.6, although this may exclude offspring, presumably dead inside dens (see Discussion). The survivors/reinvaders came from what we identified as 11 different colonies. As beavers were dispersing during the time of operations, it is difficult to compare the age/sex of the beavers caught during the mop up with those captured during the survivor/reinvader phase.

Non-target catches

Trap specificity was 90%. Non-target catches were recorded only during the first phase. One culpeo fox (*Lycalopex culpaeus*), and one upland goose (*Chloephaga picta*) were released alive. Native species killed included two spectacled ducks (*Speculanas specularis*), three unidentified ducks and two upland geese (*Chloephaga picta*). Exotic species captured included 10 muskrats (*Ondathra zibethicus*) and one mink (*Neovison vison*) which were killed and one grey fox (*Lycalopex griseus*) which was released alive.

Eradication verification phase

The 26 cameras yielded a total of 154 camera trapping nights. Nine cameras detected beavers after a period between zero to five days (average two days), and 17 cameras did not detect animals after a period of between three and 10 days (average six days). In addition, two persons walked 155 km to check for signs of presence/absence at the same time that the cameras were set. The last beaver detection was confirmed on 24 May, 2017, nine days after the last capture. Later in the year, from August to October, surveys for survivors/reinvaders were planned to continue.

DISCUSSION

This is the first eradication attempt for beavers from one area in a short time frame. The finding of survivors/reinvaders has two explanations, not mutually exclusive. First, operations may not have reached the last individuals. Second, the lack of physical barriers may ease the movements of dispersing beavers from neighbouring colonies. There had been two previous attempts at beaver removal (Schiavini, et al., 2016). The first attempt took place in the Tierra del Fuego National Park, where a sustained control plan aimed to reduce the size of the beaver colonies was followed by their complete removal from 2,000 ha in 2011. The second attempt took place in the provincial protected area of Reserva Provincial Corazón de la Isla in 2014, where beavers were removed from 4,900 ha in two months, although the project was discontinued for financial reasons and this area has been included as one of the pilot areas to be treated in the near future.

The estimated efficiency of body-gripping traps (12%) was lower than the 22% reported by Lizarralde, et al. (1996) for Tierra del Fuego. However, it must be noted that the first estimate derives from tests for trapping aimed at performance-oriented catches per number of captures. In contrast, the complete removal of animals from one area explains the lower trapping efficiency reported here. Results from the next pilot areas will allow us to have a broader view of the calculation.

The original trapping set and reviewing approach required daily checking of traps. The presence of people walking every day over the dams and dens, and in the vicinity of colonies, can make beavers more “cautious”, affecting the likelihood of removing the last animals. The potential of beavers “learning” from disturbance and becoming wary (*sensu* Morrison, et al., 2007) is a problem for efficient eradication operations. Initial data analysis did not reveal the cumulative effect of the presence of the restorers in the capture efficiency. Neither did it find beneficial effects of not setting traps for a number of days. Because part of this pilot area was subject to different intensities of trapping over the years, animals from there may already have been cautious to human disturbance. However, capture efficiency did not differ between areas with more historical trapping effort (the main valleys) and areas less accessible to trapping (the lateral valleys), suggesting a lack of “memory” from previous trapping disturbance in the area.

The next trials will give us a chance to answer the questions raised above, and explore alternative trapping effort schemes – for example, the exclusive use of body-gripping traps. This lethal tool would allow us to leave traps unattended for several days, reducing the likelihood of disturbance. However, the size and weight of body-gripping traps limit the number of traps a person can transport and manage during a day, and the trade-off is that trapping effort would be overestimated by this approach. Nevertheless, the benefits of eradication would overcome the uncertainty associated with estimating the eradication effort.

The unexpectedly small number of kits present in the catch may be because they were too young to leave the dens. The trapping effort coincided with much of the breeding season. Also, the lodges were not destroyed as part of the management process because we wanted to avoid the escape of animals from their colonies. Consequently, the most likely scenario is that kits remained in the den and starved after the mother was captured. This poses a potential constraint on the timing of future eradication attempts if animal welfare issues are considered. Although the sex ratio of the capture was even overall, females outnumbered males by 1.66:1 ($p = 0.043$) early in the trapping of each watercourse section, when 83% of females were caught. These numbers support the idea of greater mobility of females outside the lodges due to their maternal duties.

Trapping efficiency was lower during the survivor/reinvader removal phase than during the first phase. This is to be expected due to fewer remaining animals, and/or because they may have "learnt" to be more cautious. However, it is expected that reinvaders would not be as cautious as survivors. More data are needed to explore this issue. During the mop-up phase we could not identify family colonies accurately from the spatial distribution of catches, and consequently we could not discriminate survivors from reinvaders based on their sex and/or age. It is expected that genetic analyses would assist in identifying survivors from reinvaders.

Of the 28 individuals captured during the survivor/reinvader phase, 18 came from colonies previously trapped; 10 males, six females and two of undetermined sex. In five sites only males were captured, and three males were captured at one site. Most of the females were captured at the same site next to males. The sex ratio of captures for this phase did not differ significantly from 1:1 ($p=0.3$), although male catch seemed to be larger. This could be a reflection of greater male dispersion from neighbouring areas, following source–sink dynamics.

Analysis of the spatial distribution of catches indicated that 41 colonies were trapped, plus a few recolonised sites. These values are in agreement with previously known colony densities for the area. Lizarralde (1993) reported 4.72 colony sites/km, defining a colony site as "a pond, or series of ponds used by a colony of beavers throughout the year or years", different than the usual definition of a colony, that refers to a family group living in a series of ponds and sharing a common food cache. Lizarralde & Escobar (pers. comm. 2000) reported, for 1998 and 1999, densities of 0.91 and 0.45 active colonies/km for the Olivia River and of 0.67 and 0.52 colonies/km for the Lasifashaj River, respectively. Schiavini, et al. (2016), reported densities of 0.42 and 0.37 colonies/km for the Olivia and Lasifashaj rivers in March 2010.

The estimated number of beavers per colony (5.6 individuals/colony) may underrepresent kits for the reasons explained above. On the other hand, since trapping occurred during a period of high juvenile mobility, the total catch is likely to overestimate the number of individuals per colony, since it would include animals from colonies neighbouring the pilot area.

Eradication was not achieved during operations in this first pilot area since beavers were detected by trap-cameras during the verification phase and the removal and revision work continued after the month of May. The main reasons are likely to be that the area is open to reinvasion and that trapping took place during a time of high juvenile dispersal. In view of these preliminary results, a large-scale eradication programme in the Isla Grande de Tierra del Fuego (48,000 km^2), must consider the spatial progression of the operations, adjusted to the possibility of reinvasion of the area under management and to the biological cycle of beaver dispersal. Large-scale operations should be carried out either in larger areas, covering areas with physical barriers for reinvasion, and/or restorers should cover the landscape in a more structured way. It is expected that the experience gained in the rest of the trial will allow us to adjust the strategy.

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REFERENCES

Anderson, C.B., Griffith, C.R., Rosemond, A.D., Rozzi, R. and Dollenz, O. (2006). 'The effects of invasive North American beavers on riparian plant communities in Cape Horn, Chile: Do exotic beavers engineer differently in sub-Antarctic ecosystems?'. *Biological Conservation* 128: 467–474.

Anderson, C.B., Pastur, G., Lencinas, M.V., Wallem, P.K., Moorman, M.C. and Rosemond, A.D. (2009). 'Do introduced North American beavers *Castor canadensis* engineer differently in southern South America? An overview with implications for restoration'. *Mammal Review* 39: 33–52.

Collado, L. (2007). 'La vegetación de Tierra del Fuego: de la estepa a la selva'. In: J. Godoy Manríquez (ed.) *Enciclopedia Patagonia Total*, pp. 772–775. Buenos Aires: Barcelbaires Ediciones.

Eljall, A., Dieguez H., Menvielle M.F. and Hodara, K. (2016*). 'Distribución y abundancia del castor en Tierra del Fuego, Argentina: Percepción remota de la invasión de un ingeniero de los ecosistemas'* (Distribution and abundance of the beaver in Tierra del Fuego, Argentina: Remote perception of the invasion of an ecosystem engineer). Paper delivered at the "VI Reunión Binacional de Ecología". September 18–22 2016. Iguazú, Argentina.

Graells, G., Corcoran, D. and Aravena, J.C. (2015). 'Invasion of North American beaver (*Castor canadensis*) in the province of Magallanes, Southern Chile: Comparison between dating sites through interviews with the local community and dendrochronology'. *Revista Chilena de Historia Natural* 88(3): 1–9.

Henn, J.J., Anderson, C.B., and Pastur, G.M. (2016). 'Landscape-level impact and habitat factors associated with invasive beaver distribution in Tierra del Fuego'. *Biological Invasions* 18: 1679–1688.

Johnston, C.A. and Windels, S.K. (2015). 'Using beaver works to estimate colony activity in boreal landscapes'. *The Journal of Wildlife Management* 79: 1072–1080.

Lizarralde, M.S. (1993). 'Current status of the introduced beaver (*Castor canadensis*) population in Tierra del Fuego, Argentina'. *Ambio* 22: 351–358.

Lizarralde, M.S., Deferrari, G., Escobar, J., Álvarez, S. and Camilión, C. (1996). 'El trampeo de animales mamíferos: Un ensayo para el trampeo de castor y rata almizclera con trampas Conibear 330-2 y 110-2' (Trapping of mammals: An essay for the trapping of beaver and muskrat with Conibear traps 330-2 and 110-2). *Publicación Técnica de la Dirección de Recursos Naturales de la Provincia de Tierra del Fuego* 4: 1–12.

Lizarralde, M.S. and Escobar, J. (1997). '*Avances Sobre la Ecología de los Mamíferos Silvestres Introducidos en la Provincia de Tierra del Fuego*' (Advances in ecology of Introduced Wild Mammals in the Province of Tierra del Fuego). La Fauna Silvestre en Tierra del Fuego, No. 27.

Lizarralde, M.S. (2004). 'Invader species in Argentina: a review about the beaver (*Castor canadensis*) population situation on Tierra del Fuego ecosystem'. *Interciencia* 29: 352–358.

Malmierca, L., Menvielle, M.F., Ramadori, D., Saavedra, B., Saunders, A., Soto Volkart N. and Schiavini A. (2011). 'Eradication of beaver (*Castor canadensis*), an ecosystem engineer and threat to Southern Patagonia'. In: C.R. Veitch, M.N. Clout and D.R Towns (eds.) *Island invasives: eradication and management,* pp. 87–90. Occasional paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Martin, S.L., Jasinski, B.L., Kendall, A.D., Dahl, T.A. and Hyndman, D.W. (2015). 'Quantifying beaver dam dynamics and sediment retention using aerial imagery, habitat characteristics, and economic drivers'. *Landscape Ecology* 30: 1129–1144.

McTaggart, S.T. and Nelson, T.A. (2003). 'Composition and demographics of beaver (*Castor canadensis*) colonies in central Illinois'. *The American Midland Naturalist* 150: 139–150.

Morrison, S.A., Macdonald, N., Walker, K., Lozier, L. and Shaw, M.R. (2007). 'Facing the dilemma at eradication's end: uncertainty of absence and the Lazarus effect'. *Frontiers in Ecology and the Environment* 5: 271–276.

Parkes, J.P., Paulson, J., Donlan, C.J. and Campbell, K. (2008). *Control of North American Beavers in Tierra del Fuego: Feasibility of Eradication and Alternative Management Options*. Landcare Research, LC0708/084, Lincoln, NZ.

Pietrek, A.G. and Fasola, L. (2014). 'Origin and history of the beaver introduction in South America'. *Mastozoología Neotropical* 21: 355–359.

Rienzo, J.A.D., Casanoves, F., Balzarini, M.G., Gonzalez, L., Tablada, M. and Robledo, C. W. (2016). InfoStat versión 2016. <http://www.infostat.com.ar>.

Schiavini, A., Cerezani, R., Silva, M.R., Loekemeyer, N., Malmierca, L., Escobar, J., Deferrari G. and Lizarralde, M. (2008). 'Gestión y técnicas de control de castores en Argentina. Lecciones aprendidas' (Management and control techniques of beavers in Argentina. Lessons learned). In: B. Saavedra and C.A. Silva (eds.) *Proceedings of Taller Internacional para el Control de Castores en la Patagonia. Digital edition*, pp. 73–92. Chile: Wildlife Conservation Society. < http://www.karukinkanatural.cl>.

Schiavini, A., Carranza, M.L., Deferrari, G., Escobar, J., Malmierca, L. and Pietrek, A.G. (2016). 'Erradicación de especies invasoras: Ciencia, actitud y entendimiento. El castor en Tierra del Fuego' (Eradication of invasive species: Science, attitude and understanding). *Mastozoología Neotropical* 23: 279–288.

Skewes, O., Gonzalez, F., Olave, R., Ávila, A., Vargas, V., Paulsen, P. and König, H.E. (2006). 'Abundance and distribution of American beaver, *Castor canadensis* (Kuhl 1820), in Tierra del Fuego and Navarino islands, Chile'. *European Journal of Wildlife Research* 52: 292–296.

Steventon, J. (2017). Cybertracker v. 3.443. Cybertracker Software. <http://www.cybertracker.org>.

Wallem, P.K., Jones, C.G., Marquet, P.A. and Jaksic, F.M. (2007). 'Identificación de los mecanismos subyacentes a la invasión de *Castor canadensis* (Rodentia) en el archipiélago de Tierra del Fuego, Chile' (Identification of the mechanisms underlying the invasion of *Castor canadensis* (Rodentia) in the archipelago of Tierra del Fuego, Chile). *Revista Chilena de Historia Natural* 80: 309–325.

Westbrook, C.J., Cooper, D.J. and Anderson, C.B. (2017). 'Alteration of hydrogeomorphic processes by invasive beavers in southern South America'. *Science of the Total Environment* 574: 183–190.

Zar, J. H. (2010). *Biostatistical Analysis*. New Jersey, USA: Pearson Education.

Towards a genetic approach to invasive rodent eradications: assessing reproductive competitiveness between wild and laboratory mice

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Abstract House mice are significant invasive pests, particularly on islands without native mammalian predators. As part of a multi-institutional project aimed at suppressing invasive mouse populations on islands, we aim to create heavily male-biased sex ratios with the goal of causing the populations to crash. Effective implementation of this approach will depend on engineered F1 wild-lab males being effective secondary invaders that can mate successfully. As a first step in assessing this possibility, we are characterising genetic and behavioural differences between *Mus musculus* strains in terms of mating and fecundity using wild house mice derived from an invasive population on the Farallon Islands (MmF), a laboratory strain C57BL/6/129 (t^{w2}), and F1 wild-lab offspring. Mice with the 't allele' (t^{w2}) have a naturally occurring gene drive system. To assess fertility in F1 wild-lab crosses, t^{w2} males were paired with wild-derived females from the Farallon Islands (MmF). Results of these matings indicate litter sizes are comparable but that weaned pup and adult wild-lab mice are heavier in mass. Next, we initiated tests of male competitiveness using larger (3 m^2) enclosures with enrichment. We introduced both an MmF and a t^{w2}-bearing male to two MmF females to assess mating outcomes. Preliminary results of these experiments show none of the offspring carried the t-allele. However, performing the same experiment with F1 wild-lab males instead of a full lab background resulted in 70% of offspring carrying the t^{w2} allele. This indicates that F1 wild-lab males may be able to successfully compete and secondarily invade. It will be important in subsequent experiments to determine what characteristics contribute to secondary invasion success. More generally, a better understanding of characteristics contributing to overall success in increasingly complex and naturalistic environments will be critical in determining the potential of a gene drive-based eradication approach for invasive mice on islands.

Keywords: competition, gene drive, invasive rodents, reproductive fitness, secondary invasion

INTRODUCTION

Invasive rodents are a key biodiversity threat for the majority of the world's islands and eradication campaigns are often employed to prevent loss of island endemics (Howald, et al., 2007; Campbell, et al., 2015). These eradications employ rodenticides and have been successful in eliminating invasive rodents from over 400 islands (DIISE, 2017). Rodenticides, however, have a higher failure rate with mice (*Mus musculus*), as opposed to rats (*Rattus* spp.) (MacKay, et al., 2007) and their use on inhabited islands presents severe logistical challenges. Additionally, rodenticides are not species-specific and present animal welfare concerns (Campbell, et al., 2015). These challenges have created a compelling need for alternative approaches to rodent eradication.

One potentially promising approach to eliminating invasive mice from islands would be to bias offspring sex ratios by genetically engineering mice that produce only one sex of offspring. Pairing this approach with a genetic drive mechanism to spread this trait in an invasive mouse population would be critical. Key first steps are to understand the processes of reproductive competitiveness and the capability of an introduced mouse to introgress into established island populations, a process we are terming 'secondary invasion'. The phenomenon of secondary invasions and multiple introductions has been documented in invasive brown anole (*Anolis sagrei*) populations with evidence that secondary invasions may be frequent and can add genetic variation to existing invasive populations (Kolbe, et al., 2004). This secondary invader phenomenon in house mice, however, is less well understood and genetic evidence suggests variation in how this occurs across islands. Some studies suggest that secondary invaders may be frequent (Berry, et al., 1991; Bonhomme & Searle, 2012) while others suggest instead only single primary invasions (Hardouin, et al., 2010; Gabriel, et al., 2015). For rodent eradications these secondary invaders would be carrying the gene drive and spread of this construct through the population would be necessary for this approach to be effective.

The development of the CRISPR/cas9 genome editing technology has recently revolutionised genetic engineering capabilities (Barrangou & Doudna, 2016). This has increased interest in genetic pest management approaches first conceptualised by Burt (2003) and built upon by other authors more recently (Sinkins & Gould, 2006; Esvelt, et al., 2014). Many of these approaches centre on gene drives, systems in which a genetic construct producing a desired phenotype (e.g., sex ratio manipulation, sterility) is preferentially inherited by offspring. These are considered 'selfish' genetic elements because the majority of offspring will inherit the genetic construct and it therefore could spread quickly through a population (Lyttle, 1991). In mice, a naturally occurring gene drive is found on chromosome 17 and is termed the t-allele (Silver & Buck, 1993). The t-allele bearing sperm impact the motility of non-t bearing sperm and this leads to an inheritance rate of greater than 90% for the t-allele (Bauer, et al., 2005; Baker, 2008). Homozygosity of the t-allele (t/t) is typically lethal, but this is not true of the variant form termed the t^{w2} allele, although homozygosity does cause sterility (Levene & Dunn, 1961).

A gene drive-based approach to eradication could use either a naturally occurring drive or a synthetic drive based on CRISPR/Cas9 and functional drives with this technique have now been demonstrated in mosquitoes, flies, and yeast (Harris, et al., 2012; DiCarlo, et al., 2015; Gantz & Bier, 2015); see also early contributions by Craig, et al. (1960) and Hamilton (1967). Theoretically, by biasing offspring sex ratios heavily towards males, reproduction could be impaired and populations reduced. One way this could be done would be to use the *Sry* gene. The *Sry* gene is the key male determining factor in mammals and is sufficient to start the cascade of events leading to male development (Hacker, et al., 1995). Placing the *Sry* into an autosome induces development that is phenotypically male in mice that are genotypically XX (Koopman, et al., 1991). Inserting *Sry* into a naturally-occurring gene drive such as the t-allele or a synthetic drive based on

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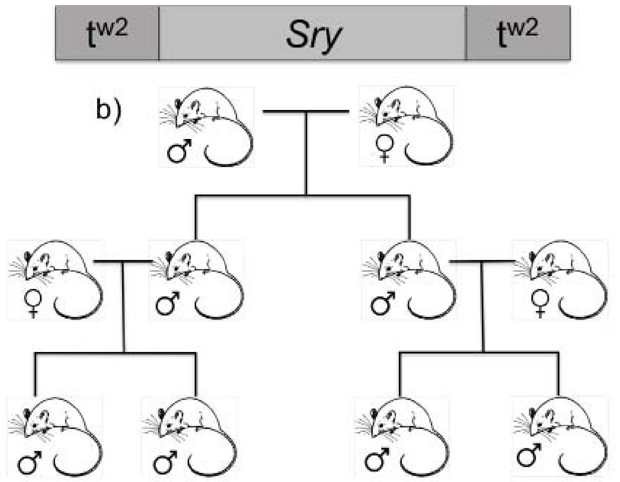

Fig.1 Depiction of the *Sry* gene inserted into the t^{w2} gene drive accompanied by a depiction of how the population would bias to be all male.

CRISPR/Cas9 should create the potential for reduction of an invasive mouse population by reducing and ultimately potentially eliminating production of fertile females (Fig. 1; Backus & Gross, 2016; Piaggio, et al., 2017; Prowse, et al., 2017). A synthetic gene drive using CRISPR/Cas9 could theoretically be employed in a similar way to ensure all offspring inherit a feminising gene.

Regardless of the genetic mechanism employed, the reproductive competitiveness and relative fitness of gene drive carriers are likely to be important in determining the success of any genetic approach to reducing invasive mouse populations. Assessing reproductive competitiveness is the focus of this study. Since mice introduced with a gene drive mechanism would essentially be secondary invaders into an established invasive mouse population, it is important to better understand processes affecting introgression into established demes. Mice are social animals and dominant males will often hold and defend a territory (i.e. deme) that provides reproductive access to reproductive females while subordinate males do not (Bonhomme & Searle, 2012). How incoming mice are able to successfully integrate into island demes is not clear. If a gene drive approach was used, then the incoming males would need to compete with the resident island males for females. Competition and aggression tend to occur between male mice when there are limited territories (Gray & Hurst, 1998). Mouse populations living non-commensally on islands can instead exhibit an 'island syndrome' where they show important differences with commensal populations. These can include increases in body mass and, importantly in the context of this study, lower levels of aggression (Adler & Levins, 1994; Gray & Hurst, 1998; Cuthbert, et al., 2016). In the 1980s, a study was conducted by capturing house mice on the Orkney island of Eday (commensal) and releasing them onto the Isle of May, which was uninhabited by humans but had an established population of non-commensal wild house mice (Berry, et al., 1991). This study followed the spread of genetic markers unique to Eday and found that these alleles moved quickly through the Isle of May population (Berry, et al., 1991; Jones, et al., 1995). Differences in aggression may relate to whether the mice are living commensally or not, with evidence indicating that commensalism and perhaps increased density favours more aggressive individuals (Berry, et al., 1991; Gray & Hurst, 1998). Overall, the limited studies to date have strongly suggested that island mice may not be as competitive as their mainland/commensal counterparts (Mackintosh, 1981; Berry, et al., 1991; Gray & Hurst, 1998).

Secondary invader success may also depend on female mate choice (Jones, et al., 1995). In terms of female mate choice, there is evidence that females prefer the scent of foreign males and are more likely to mate with unrelated males (Roberts & Gosling, 2003; Frynta, et al., 2010). Importantly, however, there is also evidence of female choice favouring non-t haplotype carrier males or males carrying a different t-haplotype variant (Lenington et al., 1994; Manser, et al., 2015; Sutter & Lindholm, 2016). The relative fitness of gene drive carriers will be a critical determinant of effectiveness for this approach. Fitness costs have been documented with other forms of the t-allele (Carroll, et al., 2004; Lindholm, et al., 2013), but have not been examined for the t^{w2} variant to our knowledge. Information about the t-allele presence on islands and modelling of population dynamics would help us further understand the transmission of the Sry/t^{w2}gene drive in island mouse populations (Backus & Gross, 2016).

Central questions

A critical aspect of exploring gene drive eradication techniques for island rodents is that the gene drive originates in a mouse strain with a standard laboratory background that is amenable to manipulation. Laboratory mice, however, have been inbred and housed in non-hierarchical social conditions for generations (Morse, 2007; Fawcett, 2012) and they have also undergone both deliberate and inadvertent selection under these captive conditions (Fawcett, 2012). It is encouraging to note, however, that wild-type behaviour can be restored quickly by backcrossing with wild-derived mice to create wild-lab crosses (Chalfin, et al., 2014). The central goals of this study are to one i) confirm that a gene drive mechanism can be bred into a wild background and ii) assess whether key reproductive measures such as litter size, pup weight, and adult weight are impacted in F1 and F2 wild-lab mice. We also present preliminary findings regarding the success of laboratory and F1 wild-lab males in competitive mating situations.

MATERIALS AND METHODS

Strains of mice

These studies employed several different strains of mice. A primary laboratory strain is C57BL/6J referred to as (B6) mice. B6 mice are the most common strain of lab mice and are easily manipulated genetically (Silver, 1995). Compared to other laboratory strains B6 mice are considered more defensive and aggressive in response to perceived threats (Blanchard, et al., 2009). A second strain was donated from the Threadgill lab at Texas A&M University. These mice are of a mixed C57BL/6J and a 129S1/SvlmJ (B6;129) background (hereafter referred to as "lab" strain) and carry the t^{w2} variant of the t-allele. The t^{w2} variant stems from a wild background but was brought into laboratory stocks in 1946 (Dunn & Morgan, 1953). These mice are not transgenic (no *Sry* inserted) and so heterozygotes produced are either male or female. The t^{w2} allele is inherited by 95% of offspring in matings with a t^{w2}/+ sire (Kanavy & Serr, 2017). To maintain t^{w2} mice, B6 females are mated to males heterozygous for the t^{w2} allele (t/+). The wild-derived mice (MmF) we use are derived from wild progenitors captured on Southeast Farallon Island, which is part of the Farallon National Wildlife Refuge, located about 30 miles off the coast of California near San Francisco (Farallon, 2013). Invasive mice are the only terrestrial mammals on the island currently (Schoenherr, et al., 1999; Farallon, 2013). These mice show annual cyclic population variation with peak densities in late summer and early fall. MmF mice do not carry the t allele (Threadgill, pers. comm. 2013).

Some of the highest mouse densities ever recorded in non-commensal habitats are seen on Southeast Farallon Island at over 1300/ha (490/acre) (Farallon, 2013; Newser, 2013). Their diet consists primarily of invertebrates (Jones, et al., 2006). The Farallons mice pose direct threats to an endemic invertebrate and indirect threats to native seabirds. The USFWS plans for a future mouse eradication with rodenticide (Farallon, 2013). We established a colony of wild-derived Farallons mice (MmF) at NCSU in 2013 and they are now 8th generation derived from the wild. These Farallon mice serve as the 'island mouse' model being used to form demes for testing the ability of secondary invaders to establish and mate successfully.

All experiments were conducted under an approved Institutional Animal Care and Use Committee protocol at North Carolina State University between 2015–2017. Mice were maintained in a temperature-controlled greenhouse with natural lighting and conditions suitable for reproduction year round. Animals were fed *ad libitum* with 5058 LabDiet® and daily health and welfare checks were performed. To test if mating between wild-derived MmF females and laboratory males occurred pairs of lab males with wild-derived MmF females were created and housed in 29 cm wide × 40 cm long × 19 cm high standard laboratory cages. Each cage contained aspen bedding, natural cotton, a 15 cm PVC tube and black oil sunflower seeds for enrichment. Mice were housed in this manner with weekly cage changes. To minimise disturbance, mice were transferred over to a clean cage using a 15 cm PVC pipe whenever possible. Pups were weaned at the mouse standard of 21 days +/- 3 days (Silver, 1995) and the litter size, sex and weight of the pups in grams were recorded. In addition, an ear punch or tail snip was taken for genotyping. Pups were then weighed as adults and their weight in grams was collected for nulliparous individuals between the ages of 70–140 days.

Tests of male competition were conducted in semi-natural enclosures. The size of these 'arenas' is 3 m^2, closely approximating the size of those used by Slade, et al. (2014). To allow for formation of hierarchies and nesting, we added enrichment and complexity in the form of sand, bricks, plastic blocks ('Legos') supporting multilevel clear Plexiglass structures, galvanized wire mesh (1.25 × 1.25 cm mesh size), cardboard boxes and cardboard egg cartons, and PVC pipes for environmental complexity. For trials, all mice were placed into the arena at the same time. Males were either weight matched to within 1 g (~5% of body weight) or age matched within 8–10 weeks. All mice used in the arenas were nulliparous and sexually mature. Coloured ear tags as well as Clairol 'Just For Men' Black Hair dye® was used to identify males. Trials included combinations of MmF and t^{w2} males as well as MmF and F1 wild-lab males. At the start of each trial, both males and two non-related MmF females were placed into the arena and filmed for one hour. During this hour, we counted the number of bouts, chases and attempts to copulate, or time in proximity with females, as a means of assessing dominance. Animal welfare checks and monitoring for pups were performed daily. Any pups born in the enclosures were weaned at the standard of 21 days and a tissue sample was collected for genotyping.

To confirm the presence of the t^{w2} haplotype, we used a modified protocol where we amplified a portion the Hba-ps4 (alpha-globin pseudogene-4) locus (Schimenti & Hammer, 1990). The procedure uses a 'dirty' DNA extraction developed by one of our collaborators at Texas A&M University (Kanavy, pers. comm. 2016). Tissue is collected and either a 2–3 mm tail snip or a 2 mm ear punch is used. The 'dirty' DNA extraction buffer contains (50 µl 5 M NaOH, 4 µl 0.5 M EDTA, and 10 ml sterile water). 100µl of extraction buffer is then added to the tissue sample and incubated at 95°C for 20 minutes. After vortexing and cooling 5 µl of 1 M HEPES is added. The sample is then centrifuged at 6,000 g for five minutes and 40 µl of DNA is extracted from the top. DNA electrophoresis of PCR products shows a distinct band at 198 bp for wildtype mice (+/+) while t^{w2} homozygotes (t/t) display a band at 214 bp and heterozygotes (t/+) show the presence of both bands.

Statistical analyses were conducted using JMP® Pro 12.2.0 (SAS) where 1-way ANOVAS were used for adult weights and litter sizes. A mixed model ANOVA with the fixed effect of litter size was used to separate litter size from pup weight to compare pup weights. Next, post-hoc analyses including orthogonal contrasts and Tukey's HSD tests were used to identify group differences. Litter sizes and weights are presented as mean ± SEM.

RESULTS

Adult weights were taken for males and females. Sample sizes for males were as follows: B6 (33), t^{w2} (24), MmF (53), F1 (21), and F2 (22). For females sample sizes were: B6 (19), t^{w2} (25), MmF (44) and F1, (23). The average day of age that adult male weights were measured at was the following: B6=80.43± 21.95; t^{w2} =90.43±27.65; MmF 92.63±34.90; F1 93.03±19.46; and F2 89.48±28.27. Similarly, for females the average day of age that the adult weight was taken was: B6 91.24±28.99; t^{w2} 88.66±24.09; MmF 89.20±36.14; and F1 82.25±38.15. Adult weights varied by strain and sex, $F_{8,257}=28.35$, $p<0.0001$. In addition, t^{w2}carrying males (t^{w2}, F1, F2) were larger than MmF males, $F=58.00$, $p<0.0001$. Similarly, t^{w2} carrying females (t^{w2} and F1) were larger than MmF females, $F=7.75$, $p=0.0058$ (Fig. 2). Due to space restrictions for husbandry, not enough F2 adult females had been reared to allow calculation of a meaningful average for this group.

While litter size varied across strains $F_{5,141}=4.59$, $p<0.0007$, MmF, F1 and F2 wild-lab mice had litter sizes that were comparable (Fig. 3). Sample sizes for litter size were as follows: B6 (27); t^{w2} (20); MmF (45); MmF/B6 (19); F1 (21); and F2 (20). There were no differences detected in the sex ratios for pups born, nor in the time of gestation (data not shown).

Weaning weight was measured with a mixed model ANOVA with litter size being a fixed effect. The samples are as follows: B6 (18); t^{w2} (14); MmF (44); MmF/B6 (20);

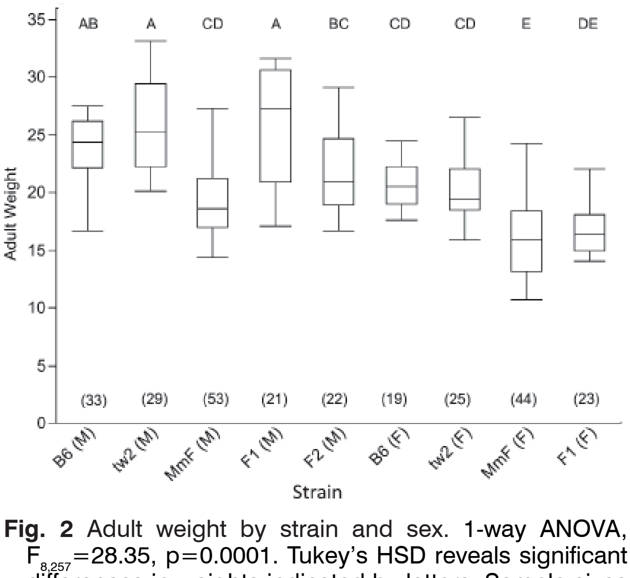

Fig. 2 Adult weight by strain and sex. 1-way ANOVA, $F_{8,257}=28.35$, $p=0.0001$. Tukey's HSD reveals significant differences in weights indicated by letters. Sample sizes are indicated in parentheses.

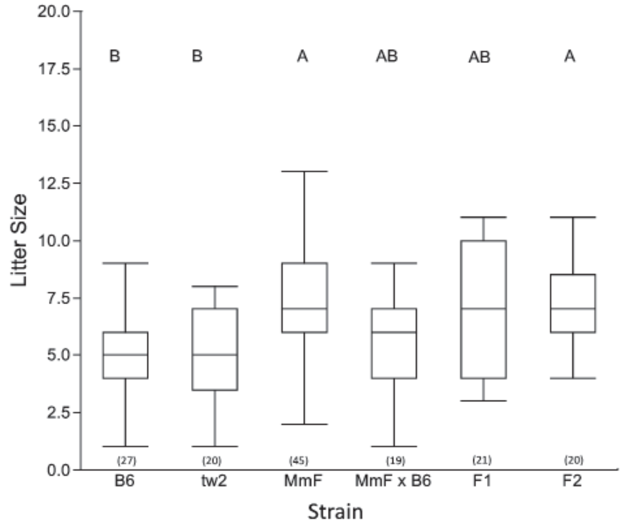

Fig. 3 Litter size by strain 1-way ANOVA, $F_{5,141}=4.59$, $p=0.0007$ indicates significant differences in litter size across strains. Tukey's HSD reveals significant differences in weights indicated by letters. Sample sizes are indicated in parentheses.

F1 (13); and F2 (20). Pup weaning weight was significantly different across strains ($F_{133,383}=13.922$, $p=0.0001$) and the highest weaning weights were found in F1 wild-lab F2 and F1s respectively (Fig. 4). Highest mean weights at weaning were 10.46± 0.40 g (F1) and 9.82± 0.33 g (F2).

In the arenas, preliminary trials of male competition between t^{w2} males (laboratory strain) and MmF males revealed no t^{w2} transmission based on genotyping (three trials with 35 pups total). The t^{w2} male initially appeared behaviourally dominant. He pursued females and chased the MmF male away, but on subsequent days was subordinate and tended to stay on top of the feeder out of view of the MmF male. Preliminary trials with MmF males and F1 wild-lab males (eight trials, 47 pups) revealed strongly contrasting results and a 70% transmission rate of the t^{w2} allele. Here, five of the eight litters did carry the t^{w2} with 31 of 33 pups from these litters confirmed. The F1 wild-lab males appeared to be behaviourally dominant throughout the trial in the same five trials where t^{w2} pups were produced.

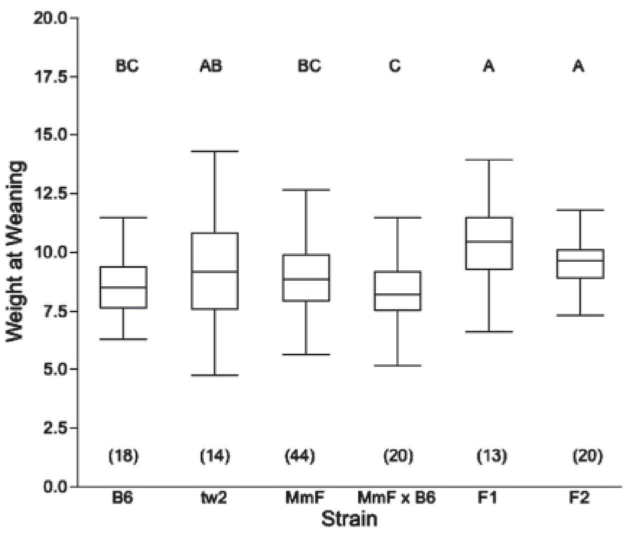

Fig. 4 Weaning weight with fixed effect of litter size Mixed Model, strain $F_{5,119}=4.98$, $p=0.0004$., litter size $F_{1,117}=12.46$, $p=0.0006$. Tukey's HSD reveal significant differences in pup weights across strains, which is indicated by letters. Sample sizes are indicated in parentheses.

Dominance was again based on initiation of chasing or fighting with the MmF male and by time spent pursuing or mating with females. When subordination did occur, the subordinate males appeared to place themselves so as not to be visible to the dominant individual. Behavioural results are ongoing and were beyond the scope of this manuscript.

DISCUSSION

Relative fitness of gene drive carriers is likely to be critical in determining the success of this approach (Burt, 2003; Manser, et al., 2015; Backus & Gross, 2016). Carriers of gene drive constructs would need to be successful in reproduction and reproductive competition if a genetic approach to invasive rodent eradication is to be effective. This work establishes some key initial conditions for this success. First, lab mice and wild mice can breed and produce viable litters. Second, while litters of the common lab background t^{w2} mice were smaller than those of wild-derived mice under the more naturalistic conditions used in this study, the F1 wild-lab litters were of comparable size to those having two wild-derived parents. Preliminary results also suggest F1 wild-lab males may have strong potential for reproductive success, a likely prerequisite for initial introgression of gene drive constructs into an island population.

This work established that wild-derived Farallon females will mate with laboratory males in standard cages and at similar frequencies to those seen in matings with wild-derived males (M. Serr, unpublished data). This was an initial but critical step in assessing reproductive output across strains and in F1 wild-lab mice. Furthermore, results indicate that both F1 wild-lab and F2 wild-lab backcrossed mice have litter sizes that are not different statistically than those of Farallon mice. This is important in terms of fitness and exploring the effectiveness of using the Sry/t^{w2} haplotype technique. It is also important to note that the reverse holds true, as wild-derived MmF males will mate with B6 and t^{w2} females in standard laboratory cages although sample sizes are not adequate for statistically meaningful comparisons. Results for pup weights indicate F1 and F2 wild-lab pups have the greatest weight at weaning and that this trend continues for adult males. Body size affects male competitiveness in mice (Cunningham, et al., 2013; Ruff, et al., 2017) with evidence suggesting that in semi-natural enclosures male mice of intermediate weight have the highest fitness (Ruff, et al., 2017). Matching mice based on body size for our experiments helps rule out this confounding factor, but for a potential gene drive release it could be beneficial for the drive-bearing mice released to weigh more than their wild counterparts.

Preliminary results from experiments in our larger arenas examining competition suggested a surprising pattern. Arena trials between MmF and t^{w2} males suggest the wild-derived MmF males are dominant to pure laboratory strain males, preventing transmission of the t^{w2} allele. Interestingly, however, weight-matched F1 wild-lab males carrying the t^{w2} allele appear more competitive and behaviourally dominant to MmF males. Consistent with this observation, we find a 70% transmission rate of the t^{w2} allele in arena trials analysed thus far. In addition, of the three trials where the F1 wild-lab male was not dominant MmF litter sizes were small with two of the three litters only having two pups each. This suggests that F1 wild-lab males are strong competitors and that females will mate with F1 wild-lab males even when both male types are present. It will be important to conduct further arena trials to assess this competitiveness with greater sample sizes and also assess the competitiveness of F2 wild-lab males. Other reproductive comparisons we are conducting include measuring testes weights. Testes weight is correlated to total sperm count in mice (Le Roy, et al., 2001). Testes weight can also predict dominance and mating success, as mice with higher testicular weight are more likely to initiate mating with females and attack behaviour towards conspecific males (McKinney & Desjardin, 1973). Finally, nesting behaviour and the temperature of nests will be important to examine across wild-derived, laboratory and F1 wild-lab mice as anecdotal observations suggest poor nest construction by laboratory mice. This could be important too because in cooler environments studies have indicated that nest building behaviour, thermoregulation, and fitness are correlated (Bult & Lynch, 1997).

Our results suggest that F1 wild-lab males could be efficient secondary invaders. This would be generally consistent with other studies from island populations (Jones, et al., 1995; Bonhomme & Searle, 2012). However, the situation may be different for females. Introduction of mice from a commensal population on the Isle of Eday to the Isle of May did not lead to the spread of mitochondrial DNA markers, which are maternally inherited. These results were in contrast to those for a Y-chromosome marker and suggested females were unable to secondarily establish while males did (Jones, et al., 1995). Studies from other islands have corroborated these results in suggesting no integrations of new maternal haplotypes from later-arriving females (Searle, et al., 2009; Gabriel, et al., 2010; Jones & Searle, 2015). This apparent male-female asymmetry in secondary establishment ability, however, has not been experimentally tested. One approach to addressing this apparent asymmetry is having records of detailed behaviour in more naturalistic arena settings. We have designed and implemented a Radio Frequency Identification (RFID) system for tracking mouse movements. RFID tracking allows collection of detailed behavioural records and works well with wild house mice (Weissbrod, et al., 2013; Auclair, et al., 2014). Behavioural measures include time spent at nest boxes, running wheels and food. With this information we can assess the number of visits, the timing of visits, the number of interactions and time in social contact with one another (König, et al., 2015; Lopes, et al., 2016).

A second approach is to test the ability of different strains to establish dominance in a standard test termed resident-intruder paradigm. A previous study used this approach to compare competitive behaviour in house mice from the Isle of Eday and the mainland, finding the island mice were significantly less aggressive (Gray & Hurst, 1998). Expanding trials to increasingly complex naturalistic experimental arenas should give insight into the relative abilities of male and female mainland mice to secondarily invade and therefore genetically introgress into an island population.

Other factors that could influence the potential success of an eradication effort include mate-choice and tolerance of island conditions. Mate-choice factors known for mice include odorant cues such as urinary proteins and ultrasonic vocalisations (Hurst & Beynon, 2004; Blanchard, et al., 2009; Musolf, et al., 2010). Island conditions and climate, in particular, could be important influences on the success of introduced mice (Berry, 1992). The island syndrome for rodents predicts increased body mass and decreased aggression (Adler & Levins, 1994; Gray & Hurst, 1998; Cuthbert, et al., 2016). In addition, the island syndrome in rodents is often associated with high population densities, increased reproductive output, and increased survival rates on islands (Adler & Levins, 1994). Mice are able to adapt to new conditions and islands (Anderson, 1978; Bronson & Pryor, 1983) and this adaptation could be critical for fitness, although any construct would presumably be introgressed into an island genetic background relatively quickly as it spread. The population genetic structure of the mice already present on an island would be critical for a synthetic gene drive, but other factors including the rate of inbreeding, ratio of reproductive males to females, and age structure of the mouse population(s) might also prove important. These are also likely to impact spread of either a synthetic or natural drive like the t-haplotype considered here. In regions with seasonality and temperature variations, mouse populations often undergo a 'boom and bust' cycle, as seen in the Farallon Islands, where the populations can erupt only to die off with changes in temperature. The timing of release of secondary invaders will likely be important in these situations (Singleton, et al., 2005; Farallon, 2013; Backus & Gross, 2016). Both natural and sexual selection could influence the number of drive carrier mice that would be required for eradication success. A study by Backus & Gross (2016) modelling the Sry/t^{w2} gene drive found that the relative fitness of the mice carrying the gene drive determined whether multiple releases would be required. Similarly, Prowse, et al. (2017) modelled synthetic gene drives and found that a sex reversing drive would require multiple releases to achieve eradication success.

The concept of reducing invasive mouse populations through release of genetically-modified mice is still in the early stages of development. Many key issues will need to be addressed to determine whether this is a feasible approach. We have shown that an island-derived wild strain will mate with t^{w2}-carrying laboratory males and produce comparable litter sizes to those of wild–wild matings. Promisingly, we also see that pup-weaning weights are larger for F1 and F2 wild-lab mice and that F1 wild-lab males may be stronger competitors in semi-natural enclosures. A key future step will be to scale up trials in arena size and environmental complexity. Larger enclosures could be used with greater numbers of mice to test whether a gene drive can spread under controlled and biosecure, but naturalistic conditions. Finally, beyond the technical issues discussed above, social license for any environmental releases would be crucial (NASEM, 2016). As gene drives are a new technology still in development, input from the relevant publics and regulatory authorities will be very important moving forward and this input is also likely to lead to additional interesting and important questions that developers will need to address.

REFERENCES

Adler, G.H. and Levins, R. (1994). 'The island syndrome in rodent populations'. *The Quarterly Review of Biology* 69(4): 473–490.

Anderson, P. K. (1978). 'The serendipitous mouse'. *Natural History* 87: 38–43.

Auclair, Y., König, B., Ferrari, M., Perony, N. and Lindholm, A.K. (2014). 'Nest attendance of lactating females in a wild house mouse population: Benefits associated with communal nesting'. *Animal Behaviour* 92: 143–149.

Backus, G.A. and Gross, K. (2016). 'Genetic engineering to eradicate invasive mice on islands: Modeling the efficiency and ecological impacts'. *Ecosphere* 7(12): e01589.

Baker, A.E. (2008). 'Mendelian inheritance of t haplotypes in house mouse (*Mus musculus domesticus*) field populations'. *Genetics research* 90: 331–339.

Barrangou, R. and Doudna, J.A. (2016). 'Applications of CRISPR technologies in research and beyond'. *Nature Biotechnology* 34(9): 933–941.

Bauer, H., Willert, J., Koschorz, B. and Herman, B. (2005). 'The t complex–encoded GTPase-activating Protein Tagap1 acts as a transmission ratio distorter in mice'. *Nature Genetics* 37(9): 969–973.

Berry, R.J., Triggs, G.S., King, P., Nash, H.R. and Noble, L.R. (1991). 'Hybridization and gene flow in house mice introduced into an existing population on an island'. *Journal of Zoology* 225(4): 615–632.

Berry, R.J. (1992). 'The significance of island biotas'. *Biological Journal of the Linnean Society* 46: 3–12.

Blanchard, C.D., Crawley, J.N., Arakawa, H. and Blanchard, R. (2009). 'Social Behaviors in Wild and Laboratory Mice with a Special Emphasis on the C57BL/6J inbred strain'. In: W.E. Crusio, F, Sluyter, R.T. Gerlai, and S, Pietropaolo (eds.) *Behavioral Genetics of the Mouse*: *Volume I,* pp. 205–217.Cambridge, MA: Cambridge University Press.

Bonhomme, F. and Searle, J. (2012). 'House Mouse Phylogeography'. In: M. Macholan (ed.) *Evolution of the House Mouse*, pp. 278–296. Cambridge, MA: Cambridge University Press.

Bronson, F.H. and Pryor, S. (1983) 'Ambient temperature and reproductive success in rodents living at different latitudes'. *Biology of Reproduction* 29: 72–80.

Bult, A. and Lynch, C.B. (1997). 'Nesting and fitness: Lifetime reproductive success in house mice bidirectionally selected for thermoregulatory nest-building behavior'. *Behavior Genetics* 27(3): 231–240.

Burt, A. (2003). 'Site-specific selfish genes as tools for the control and genetic engineering of natural populations'. *Proceedings of the Royal Society of London: Biological Sciences*, 270(1518): 10.1098/rspb.2002.2319.

Campbell, J.K., Beek, J., Eason, C.T., Glen, A.S., Godwin, J., Gould, F., Holmes, N.D., Howald, G.R. and Madden, F.M. (2015). 'The next generation of rodent eradications: Innovative technologies and tools to improve species specificity and increase their feasibility on islands'. *Biological Conservation* 185: 47–58.

Carroll, L.S., Meagher, S., Morrison, L., Penn, D.J. and Potts, W.K. (2004). 'Fitness effects of a selfish gene (the *Mus* T complex) are revealed in an ecological context'. *Evolution* 58: 1318–1328.

Chalfin, L., Dayan, M., Levy, D.R., Austad, S., Miller, R., Iraqi, F., Dulac, C. and Kimchi, T. (2014). 'Mapping ecologically relevant social behaviours by gene knockout in wild mice'. *Nature Communications* 4569: 10.1038/ncomms5569.

Craig, G., Hickey, W. and Van de Hey, R. (1960). 'An inherited male-producing factor in *Aedes aegypti*'. *Science* 132: 1887–1889.

Cunningham, C.B., Wanless, R.M., Angel, A., Burle, M.H., Hilton, G.M., Louw, H Visser, P., Wilson, J.W. and Ryan, P.G. (2013). 'Competitive ability in male house mice (*Mus musculus*): Genetic influences'. *Behavior genetics* 43(20): 151–60.

Cuthbert, R.J., Wanless, R.M., Angel, A., Burle, M.H., Hilton, G.M., Louw, H Visser,P., Wilson, J.W, and Ryan, P.G (2016). 'Drivers of predatory behavior and extreme size in house mice *Mus musculus* on Gough Island'. *Journal of Mammalogy* 97(2): 533–544.

DiCarlo, J.E., Chavez, A., Dietz, S.L., Esvelt, K.M. and Church, G.M. (2015). 'Safeguarding CRISPR-Cas9 gene drives in yeast'. *Nature Biotechnology 33*(12): 1250–1255.

DIISE (2017). 'Database of Island Invasive Species Eradications, developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand'. <http://diise.islandconservation.org>. Accessed 13 June 2017.

Dunn, L.C., and Morgan, W.C. (1953). 'Alleles at a mutable locus found in wild house mice (*Mus musculus*) *Genetics* 39: 391–402.

Esvelt, K.M., Smidler, A.L., Catteruccia, F. and Church, G.M. (2014). 'Concerning RNA-guided gene drives for the alteration of wild populations'. *Elife* 3: e03401.

Farallon (2013). *Farallon National Wildlife Refuge South Farallon Islands Invasive House Mouse Eradication Project: Revised Draft Environmental Impact Statement.* San Francisco Bay National Wildlife Refuge Complex, Fremont, California: United States Fish and Wildlife Service. <https://www.fws.gov/uploadedFiles/Farallon_House_Mouse_Eradication_Revised_DEIS_compressed.pdf>. Accessed 27 June 2017.

Fawcett, A. (2012). *Animal Research Review Pane Guideline 22: Guidelines for the Housing of Mice in Scientific Institutions.* Australia: Animal Welfare Unit, NSW Department of Primary Industries. <http://www.animalethics.org.au/__data/assets/pdf_file/0004/249898/Guideline 22-mousehousing.pdf/>. Accessed 27 June 2017.

Frynta, D., Volfová, R., Fraňková-Nováková, M. and Stejskal, V. (2010). 'Oestrous females investigate the unfamiliar male more than the familiar male in both commensal and non-commensal populations of house mice'. *Behavioural Processes* 83(1): 54–60.

Gabriel, S.I., Jóhannesdóttir, F., Jones, E.P. and Searle, J.B. (2010). 'Colonization, mouse-style'. *BioMed Central Biology* 8(131): 2–4.

Gabriel, S.I., Mathias, M.L. and Searle, J.B. (2015). 'Of mice and the "Age of Discovery": The complex history of colonization of the Azorean archipelago by the house mouse (*Mus musculus*) as revealed by mitochondrial DNA variation'. *Journal of Evolutionary Biology* 28(1): 130–145.

Gantz, V.M. and Bier, E. (2015). 'The mutagenic chain reaction: A method for converting heterozygous to homozygous mutations'. *Science* 348(6233): 442–444.

Gray, S. and Hurst, J. (1998). 'Competitive behaviour in an island population of house mice, *Mus domesticus*'. *Animal Behaviour* 56(5): 1291–1299.

Hacker, A., Capel, B., Goodfellow, P. and Lovell-Badge, R. (1995). 'Expression of *Sry*, the mouse sex determining gene'. *Development* 121(6): 1603–1614.

Hamilton, W.D. (1967). 'Extraordinary sex ratios'. *Science* 156(3774): 477–488.

Hardouin, E.A., Chapuis, J.L., Stevens, M.I., Bettine van Vuuren, J., Quillfeldt, P. Scavetta, R.J., Teschke, M. and Tautz, D. (2010). 'House mouse colonization patterns on the sub-Antarctic Kerguelen Archipelago suggest singular primary invasions and resilience against reinvasion'. *BioMed Central Evolutionary Biology* 10(1): 325.

Harris, A.F., McKemey, A.R., Nimmo, D., Curtis, Z., Black, I., Morgan, S.A., Oviedo, M.N., Lacroix, R., Naish, N., Morrison, N.I., Collado, A., Stevenson, J., Scaife, S. Dafa'alla, T., Fu, G., Phillips, C., Miles, A., Raduan, N., Kelly, N., Beech, C., Donnelly, C.A., Petrie, W.D. and Alphey, L. (2012). 'Successful suppression of a field mosquito population by sustained release of engineered male mosquitoes'. *Nature Biotechnology* 30: 828–830.

Howald, G., Donlan, C., Galván, J., Russell, J., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Hurst, J.L. and Beynon, R.J. (2004). 'Scent wars: The chemobiology of competitive signaling in mice'. *BioEssays*, 26(12): 1288–1298.

Jones, C.S., Noble, L.R., Jones, J.S., Tegelström, H., Triggs, G.S. and Berry, R.J. (1995). 'Differential male genetic success determines gene flow in an experimentally manipulated mouse population'. *Proceedings Biological Sciences*260(1359): 251–256.

Jones, M.A., Golightly, M.A., Howald, G.R.A, Saminego, A. (2006). 'Annual Variation in the Diet of House Mice (*Mus musculus*) on Southeastern Farallon Island'. Unpublished Report. Arcata, California: Department of Wildlife, Humboldt State University.

Jones, E.P. and Searle, J.B. (2015). 'Differing Y chromosome versus mitochondria DNA ancestry, phylogeography, and introgression in the house mouse'. *Biological Journal of the Linnaean Society*115(2): 348–361.

Kanavy, D. and Serr, M. (2017). '*Sry* gene drive for rodent control: Reply to Gemmell and Tompkins'. *Trends in Ecology and Evolution* 32(5): 315–316.

Kolbe, J.J., Richard, E.G., Schettino, L.R., Lara, A.C., Larson, A. and Losos, J.B. (2004). 'Genetic variation increases during biological invasion by a Cuban lizard'. *Nature* 431(7005): 177–181.

König, B., Lindholm, A.K., Lopes, P.C., Dobay, A., Steinert, S. and Buschmann, F.J.U. (2015). 'A system for automatic recording of social behavior in a free-living wild house mouse population'. *Animal Biotelemetry* 3(1): 39.

Koopman, P., Gubbay, J., Vivian, N., Goodfellow, P. and Lovell-Badge, R. (1991). 'Male development of chromosomally female mice transgenic for *Sry*'. *Nature* 351(6322): 117–21.

Lenington, S., Coopersmith, C., and Erhart, M. (1994). 'Female preference and variability among t-haplotypes in wild house mice'. *The American Naturalist,* 143(5): 766–784.

Le Roy, I., Tordjman, S., Migliore-Samour, D., Degrelle, H. and Roubertoux, P. (2001). 'Genetic architecture of testis and seminal vesicle weights in mice'. *Genetics* 158(1): 333–340.

Levene, H. and Dunn, L.C. (1961). 'Population dynamics of a variant t-allele in a confined population of wild house mice'. *Evolution* 15(4): 385–393.

Lindholm, A.K., Musolf, K., Weidt, A. and König, B. (2013). 'Mate choice for genetic compatibility in the house mouse'. *Ecology and Evolution* 3(5): 1231–1247.

Lopes, P.C., Block, P. and König, B. (2016). 'Infection-induced behavioural changes reduce connectivity and the potential for disease spread in wild mice contact networks'. *Scientific Reports* 6: 31790.

Lyttle, T.W. (1991). 'Segregation distorters'. *Annual Review Genetics* 25: 51–57.

MacKay, J.W.B., Russell, J.C. and Murphy, E.C. (2007). 'Eradicating House Mice from Islands: Successes, Failures, and the Way Forward'. In: G.W. Witmer, W.C. Pitt and K.A. Fagerstone (eds.) *Managing Vertebrate Invasive Species: Proceedings of an International Symposium,* pp. 293–304. Fort Collins, Colorado, USA: USDA/APHIS Wildlife Services, National Wildlife Research Center.

Mackintosh, J.H. (1981). 'Behaviour of the house mouse'. *Symposia of the Zoological Society of London* 47: 337–365.

Manser, A., König, B. and Lindholm, A.K. (2015). 'Female house mice avoid fertilization by t haplotype incompatible males in a mate choice experiment'. *Journal of Evolutionary Biology* 28(1): 54–64.

McKinney, T.D. and Desjardin, C. (1973). 'Postnatal development of the testis, fighting behavior, and fertility in house mice'. *Biological Reproduction* 9: 279–294.

Morse, H.I. (2007). 'The Mouse in Biomedical Research, Volume 1'. In: J. Fox, S. Barthold, M. Davisson, C. Newcomer, F. Quimby and A. Smith (eds.). *The Mouse in Biomedical Research: History, Wild Mice, and Genetics,* pp. 1–11. Burlington: Elsevier Inc.

Musolf, K., Hoffmann, F. and Penn, D.J. (2010). 'Ultrasonic courtship vocalizations in wild house mice, *Mus musculus musculus'*. *Animal Behaviour* 79(3): 757–764.

National Academies of Sciences, Engineering, and Medicine. (2016). *Gene Drives on the Horizon: Advancing Science, Navigating Uncertainty, and Aligning Research with Public Values*. Washington, DC: The National Academies Press.

Newser, K.S. (2013). 'World's most rodent-packed island has just 2 options'. *USA Today*. (9 August 2013). <www.usatoday.com/story/news/nation/2013/08/19/newserrodent-island/2670717/>. Accessed 8 June 2017.

Piaggio, A.J., Segelbacher, G., Seddon, P.J., Alphey, L., Bennett, E.L., Carlson, R.H., Friedman, R.M., Kanavy, D., Phelan, R., Redford, K.H., Rosales, M., Slobodian, L. and Wheeler, K. (2017). 'Is it time for synthetic biodiversity conservation?' *Trends in Ecology and Evolution* 32(2): 97–107.

Prowse, T.A.A., Cassey, P., Ross, J.V., Pfitzner, C., Wittmann, T. and Thomas, P. (2017) 'Dodging silver bullets: Good CRISPR gene-drive design is critical for eradicating exotic vertebrates'. *Proceedings of the Royal Society B: Biological Sciences* 284: 20170799.

Roberts, S.C. and Gosling, L.M. (2003). 'Genetic similarity and quality interact in mate choice decisions by female mice'. *Nature Genetics* 35(1): 103–106.

Ruff, J.S., Cornwall, D.H., Morrison, L.C., Cauceglia, J.W., Nelson, A.C., Gaukler, S.M., Meagher, S., Carroll, L.S. and Potts, W.L. (2017). 'Sexual selection constrains the body mass of male but not female mice'. *Ecology and Evolution* 7(4): 1271–1275.

Schimenti, J. and Hammer, M. (1990). 'Rapid identification of mouse t haplotypes by PCR polymorphism(PCRP)'. *Mouse Genome* 87: 108.

Schoenherr, A., Feldmeth, R. and Emerson, M. (1999). *Natural History of The Islands of California*. Los Angeles: University of California Press.

Searle, J.B., Jones, C.S., Gündüz, İ., Scascitelli, M., Jones, E.P., Herman, J.S., Rambau, R.V., Noble, L.R., Berry, R.J., Giménez, M.D. and Jóhannesdóttir, F. (2009). 'Of mice and (Viking?) men: Phylogeography of British and Irish house mice'. *Proceedings. Biological Sciences* 276(1655): 201–207.

Silver, L. and Buck, C. (1993). 'The mouse t complex distorter-3 (*Tcd*-3) locus and transmission ratio distortion'. *Genetics Research* 62(2): 133–137.

Silver, L. (1995). *Mouse Genetics: Concepts and Applications*. New York: Oxford University Press.

Singleton, G.R., Brown, P.R., Pech, R.P., Jacob, J., Mutze, G. and Krebs, C.J. (2005). 'One hundred years of eruptions of house mice in Australia – a natural biological curio'. *Biological Journal of the Linnean Society* 84(3): 617–627.

Sinkins, S.P. and Gould, F. (2006). 'Gene drive systems for insect disease vectors'. *Nature Review Genetics* 7: 427–435.

Slade, B., Parrott, M.L., Paproth, A., Magrath, M.J.L., Gillespie, G.R. and Jessop, T.S. (2014). 'Assortative mating among animals of captive and wild origin following experimental conservation releases'. *Biology Letters* 10: 20140656.

Sutter, A., and Lindholm, A.K. (2016). 'Meiotic drive changes sperm precedence patterns in house mice: potential for male alternative mating tactics?'. *BioMed Central Evolutionary Biology* 16(1): 133.

Weissbrod, A., Shapiro, A., Vasserman, G., Edry, L., Dayan, M., Yitzhaky, A., Hertzberg, L., Feinerman, O. and Kimchi, T. (2013). 'Automated long-term tracking and social behavioural phenotyping of animal colonies within a semi-natural environment.' *Nature Communications* 4: 2018.

Considerations and consequences when conducting aerial broadcast applications during rodent eradications

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Abstract Aerial broadcast application is currently one of the most common methods for conducting rodent eradications on islands, particularly islands greater than 100 ha or with complex and difficult topography where access by ground teams is difficult. Overall, aerial broadcast applications have a high success rate, but can be burdened by logistical, regulatory, and environmental challenges. This is particularly true for islands where complex shorelines, sheer terrain, and the interface with the marine environment pose additional risks and concerns. Using data collected during ten eradication projects we investigate the influence that operational realities have on broadcast applications. We tested the association between the amount of bait used and island size, topography, and the desire to reduce bait application into the marine environment and then compared planned bait application to actual bait application quantities. Based on our results, islands of decreasing size and increasing coastal complexity tended to use more bait than anticipated and experienced greater variability in localised bait densities. During operations, we recommend analysing flight data to identify treated areas with localised bait densities that fall below the target application rate. We recommend that areas with low localised bait densities may result in biologically significant gaps that should receive an additional application of bait based on project risk variables such as target home range size, non-target bait competitors, and alternative foods. We also recommend a common language for discussing aerial broadcast applications and where future work can be done to improve operational decision making.

Keywords: bait density, gaps, geographic information systems (GIS), island invasives, operational monitoring

INTRODUCTION

History of aerial broadcast applications

One of the primary principles for rodent eradication is ensuring sufficient bait is distributed to every potential rodent home range, so that every rodent is exposed to bait for long enough to cause mortality (Bomford & O'Brien, 1995; Howald, et al., 2007). The aerial application of rodenticide is one of the most common and effective ways for eradicating rodents from islands (Holmes, et al., 2015). Aerial broadcast techniques were first developed in the 1980s and methodology and principles were developed over several decades as lessons learnt were applied to projects of increasing size and complexity (Towns & Broome, 2003). The first aerial applications relied on the use of modified "monsoon" fire-fighting buckets slung beneath a helicopter and flown by eye or guided by ground personnel. These early projects were often successful in removing rodents, despite difficulty in controlling application rates and the need to use hand spreading to fill gaps (Garden, et. al., 2019). The advent of specialised mechanical spreading buckets to control bait application rates and distribution, and global positioning systems (GPS) to guide pilots along straight flight paths and record bait spread, revolutionised aerial application techniques (Garden, et. al., 2019). These changes allowed rodent bait to be delivered with far greater precision over much larger areas, resulting in the successful removal of rodents from islands larger than 10,000 ha (Campbell 11,300 ha; Macquarie 12,800 ha; and South Georgia 108,700 ha) (Broome, 2009; Russell & Broome, 2016; Martin & Richardson, 2017).

Aerial application principles

It is impossible to predict where all rodent home ranges are and, because rodents are highly tolerant of a wide range of habitat types, the whole island must be assumed to support rodents, and the entire island is ultimately treated. Bait application rates are set to ensure that bait is readily available in all potential rodent home ranges and target bait application rates are often informed by bait availability trials (Pott, et al., 2015) or rates used on similar islands that were previously successful (Broome, et al., 2014). These rates are conservatively selected to ensure enough bait for all the rodents on the islands while accounting for loss and uptake by non-target competitors, like land crabs, that reduce the amount of bait rodents are exposed to (Pott, et al., 2015).

In general, one bait application rate is targeted across an entire island because stratification increases complexity and the risk of gaps in bait coverage (i.e. areas where some rodents may not be exposed to bait), increasing the risk of eradication failure (Keitt, et al., 2015). Subsequently, projects are generally designed to use parallel flight lines with 50% overlap between lines and additional parallel flights along the coast to reduce the risk of gaps. Projects may apply additional bait on steep cliffs because they have a larger surface area (3D) than planar area (2D), resulting in un-even bait distribution from bait falling downslope (Broome, et al., 2014).

Challenges in aerial application

There are technical limitations of helicopters and mechanical bait spreaders in applying bait over an entire island. Operational realities, like wind, flight speed and turning capabilities of the helicopters, steep terrain, and unevenness of bait pellet distribution from the mechanical spreader can impact bait placement on the ground, leading to potential gaps in coverage. To ensure sufficient coverage the pilot must reapply bait over potential gap areas, resulting in locally increased bait densities where this additional application partially overlaps with previous flight lines. Additional complications arise when areas need to be excluded from aerial application, such as human habitation, inland water features or the marine environment. These operational constraints tend to increase the total amount of bait needed because additional overlapping flight lines are required to ensure no gaps in coverage exist along edge boundaries.

When trying to eradicate a rodent population, planning tends to focus on targeting the worst-case scenario,

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ensuring that there are no gaps, meaning that bait overlaps with the smallest known home range. However, it is not well understood if applying less bait could constitute biologically significant gaps where reduced bait availability within a rodent's home range decreases the likelihood of a rodent being exposed to a lethal dose. The potential risks posed by biologically significant gaps may be particularly relevant on tropical islands, which tend to have more non-target bait competitors and alternative food sources (Holmes, et al., 2015), or when targeting multiple rodent species.

These challenges have generally led to an "over-engineering" approach to project design under the perception that more bait increases the likelihood of eradication success (Cromarty, et al., 2002); however, higher bait use has trade-offs, such as increasing risk to non-target species (Parkes, et al., 2011). We sought to improve existing knowledge of what constitutes an 'optimal' bait application rate, and what is a biologically relevant gap in baiting. We examined ten projects to 1) understand factors influencing the difference in bait use between what was planned and what happened on the ground, and 2) characterise localised bait application rates amongst these ten projects to further understand what may constitute a gap. Specifically, we asked:

What are the differences in total bait used between three baiting scenarios and what physical and operational factors are associated with these differences?

How does localised bait application rate vary and how do areas estimated to be below the target application compare to rodent home range size?

METHODS

Aerial application terminology

The *target application rate* is the desired rate of bait deployment, in mass per unit area (e.g. kg/ha), to be applied across the island. The target application rate is usually based on bait availability trials and is set to maintain bait availability for a certain period. The *average application rate* is the total amount of bait distributed over an island divided by the area of the island, in bait mass per unit area, and is generally used for comparing eradication projects.

In general, bait is applied via a modified fertiliser bucket underslung from a helicopter that distributes bait either 360 degrees (*full swath*) or 180 degrees (*half swath* or directional) from the bucket. Each bucket throws bait pellets a certain distance as a function of bait product size and weight and the speed of the distribution spinner. The *swath width* is the effective distance that baits are consistently sown, which is conservatively set during calibration trials and less than the maximum distance the bucket can throw bait.

The *flow rate* is the rate, in mass per unit time (i.e. kg/sec), at which bait is distributed by the bucket. This may be controlled in a variety of ways, depending on the mechanics of a bucket, but is often controlled manually with *aperture discs* that vary in size to restrict how much bait can enter the spinner.

A bucket's *sow rate* is the rate, in mass per unit area (e.g. kg/ha), that bait is distributed from the bucket and is a function of the helicopter's *flight speed* and the bucket's *flow rate.* In general, a faster *flight speed* will decrease the *sow rate* while a larger *aperture disc* will increase the *sow rate.*

Using a GPS unit, bait is generally spread in parallel *flight lines* employing planned *overlap* between flight lines to reduce the possibility of gaps in bait coverage. When using overlap the *sow rate* must be reduced to achieve the desired *target application rate* (i.e. using a planned 50% overlap buckets would require a sow rate of 5 kg/ha if the target application rate was 10 kg/ha). In areas where multiple flight line *swaths overlap* localised *bait densities* achieved on the ground, in mass per unit area (e.g. kg/ha), may be higher than the *target application rate,* and where planned *overlap* does not occur *bait densities* may be lower – resulting in *undertreated areas.* The GPS unit assists helicopter pilots during bait application by indicating deviance from the desired flight line and displaying the current flight speed.

Supplemental bait is additional bait needed to fill unplanned *gaps*, *undertreated areas*, or areas that require additional treatment like steep cliffs or preferred habitat. *Contingency bait* is bait held in reserve to replace spoiled bait and is generally intended to be left unused at the end of an operation.

Data from aerial broadcast eradication projects

Between 2008 and 2016 aerial baiting data were collected and analysed across ten different rodent eradication projects representing a variety of different island habitats, sizes, strategies, outcomes, and regulatory environments (Table 1). We used these data for our analyses.

For each operation, an aerial baiting plan was developed to estimate the total amount of bait required to complete the operation. High resolution satellite imagery (<1 metre per pixel) was acquired and used to estimate the island area by digitising along the mean high-water mark at a scale of 1:2,500. Treatment area estimates were generated by calculating the area from hypothetical parallel flight lines over the island with 50% overlap, using an estimated effective swath width, and a single directional coastal boundary swath, at half the estimated effective swath width, along the coastline. For the nine projects with the most conservative regulatory guidelines that restricted bait entry into the marine environment, the start and end of the parallel flight lines were brought in from the coast by half of the estimated effective swath width, and an additional coastal overlap buffer was estimated that overlapped with the ends of the interior flight lines and the coastal swath.

On several operations, areas were identified for supplemental treatment (e.g. steep cliffs) or exclusion from aerial treatment (e.g. inland bodies of water, human habitation) and treatment areas were calculated based on the operational parameters. Steep cliff areas were estimated by acquiring Digital Elevation Models (DEM) with a resolution of 30 metres per pixel or better. Slope estimates were calculated based on the DEM and used to identify areas for additional treatment. Exclusion zones were treated like the coastal edge, with flight line ends starting and stopping at least half the effective swath width from the exclusion boundary and a half swath flown around the exclusion boundary to minimise gaps.

To estimate the total amount of bait required per application treatment, area estimates were multiplied by the sow rates required to achieve the target application rate on the ground.

Aerial bait tracking

During each operation, a tracking worksheet was completed that recorded detailed information about each bucket load including: helicopter departure time, helicopter arrival time, bucket type, disc size, bait placed in the bucket, bait returned in the bucket, and cumulative area treated as recorded by GPS (TracMap Ltd., Otago, New Zealand iOS 1.7.2). For each bucket load the amount

Table 1 Operational data analysed to evaluate factors influencing total bait used.

Project	Country	Year	Block	Habitat	Island type	Max. elev. (m)	Size (ha)	Coastline (km)	No. of flight lines	Supplemental treatment	Coastal overlap buffer	Target rate (kg/ha)	Δuniform.planned	Δplanned.actual	Δuniform.actual
Rabida	EC	2010	Plaza Norte	Tropical	Volcanic	5	8.4	2.1	9	None	FALSE	6.0	0.0	82.5	82.5
Pinzon	EC	2012	Plaza Sur	Tropical	Volcanic	10	14.8	2.7	25	None	FALSE	6.0	31.1	7.2	40.5
Rabida	EC	2010	Bainbridges, Sombrero Chino and Beagles	Tropical	Volcanic	35	72.2	9.2	116	Coast	FALSE	6.0	41.7	70.6	141.8
Acteon-Gambier	FP	2015	Gambier (Manui, Makaroa, Kamaka*)	Tropical	Volcanic	140	88.0	8.1	103	None	FALSE	24.0	34.1	34.1	79.8
Desecheo	US	2012	Desecheo*	Tropical	Volcanic	220	117.1	7.9	85	None	TRUE	20.0	3.2	-6.0	-3.0
Desecheo	US	2016	Desecheo	Tropical	Volcanic	220	117.1	7.9	159	Cliff	TRUE	34.0	36.7	-1.7	34.5
Rabida	EC	2010	Bartolome	Tropical	Volcanic	80	129.3	7.2	68	Coast	TRUE	6.0	47.9	2.0	50.8
Palmyra	US	2011	Palmyra	Tropical	Coral	10	234.9	64.0	664	None	TRUE	80.0	12.2	-6.5	5.0
Acteon-Gambier	FP	2015	Vahanga	Tropical	Coral	10	380.0	25.0	288	None	FALSE	24.0	21.1	6.1	28.5
Acteon-Gambier	FP	2015	Tenarunga	Tropical	Coral	10	425.0	23.5	214	None	FALSE	24.0	16.5	13.5	32.2
Acteon-Gambier	FP	2015	Temoe	Tropical	Coral	5	431.0	36.9	431	None	FALSE	24.0	28.5	13.7	46.1
Rabida	EC	2010	Rabida	Tropical	Volcanic	340	499.0	11.1	117	Coast	TRUE	6.0	19.1	6.5	26.9
Wake	US	2012	Wake+	Tropical	Coral	5	637.0	39.6	776	None	TRUE	15.2	15.6	5.6	22.1
Murchison & Faraday	CA	2013	Murchison and Faraday	Temperate	Volcanic	190	806.0	40.9	607	None	TRUE	16.0	11.8	23.4	38.0
Pinzon	EC	2012	Pinzon	Tropical	Volcanic	430	1,789.6	18.3	408	Cliff	TRUE	6.0	9.2	2.5	12.0
Antipodes	NZ	2015	Antipodes	Temperate	Volcanic	366	2,129.5	33.7	661	Cliff	TRUE	16.0	7.1	20.8	29.3
Hawadax	US	2008	Hawadax	Temperate	Volcanic	340	2,900.0	43.8	1096	Coast	TRUE	6.0	5.3	1.2	6.5

*Project failed to remove invasive rats.
+Project successfully removed one of two species of rats.

of bait used and area treated were calculated and used to estimate the sow rate achieved. The sow rate information was relayed to project management and the pilot to inform decisions about adjusting disc size or flight speed to ensure a consistent sow rate.

Flight line data were downloaded from the GPS unit and treatment polygons (spatial representations of where bait was spread) were estimated by buffering the flight lines based on the effective swath width calculated during operational bucket calibration. Using the helicopter times from the tracking worksheet and the times recorded in the flight line GPS data, the recorded sow rates were assigned to treatment polygons (now spatial representations of where bait was spread and at what rate it was applied). GIS-derived bait density estimates were calculated by dissolving overlapping treatment polygons into new non-overlapping polygons and summing the sow rates of the overlapping parts. Bait density estimates and flight line maps were reviewed to identify gaps or undertreated areas.

Factors associated with difference in planned and actual bait amounts used

To evaluate what factors were associated with the total bait applied during an aerial operation, aerial baiting data from the ten projects, comprising 17 different island blocks, were collated (Table 1). In some cases, an island block comprised of multiple treatment units (i.e. motu or small islets) that were treated collectively. There were three projects where multiple island blocks were treated as independent units. Ten exploratory factors thought to be associated with differences in aerial bait applications were collected for each application (Table 2). Only the first application for each island block was analysed as they were the most comparable because the amount of bait applied during the second application could be influenced by the amount of bait used during the first application, bait availability monitoring data, or the use of supplemental bait.

These ten factors were compared against two response variables, referred to as bait use scenarios: 1) the percent change between the bait amount in a hypothetical uniform scenario, where bait is evenly distributed across an island, and the planned amount of bait to be used (Δuniform.planned); and 2) the percent change between the planned amount of bait and the actual amount of bait used (Δplanned.actual) (Fig. 1). The variable 'Δuniform.planned' represents the change in bait required between a uniform application and what was planned to account for physical island characteristics and strategy decisions such as reducing bait into the marine environment. The variable 'Δplanned.actual' represents the difference between what was planned and what happened on the day due to operational realities, such as unexpected deviations in sow rates and flight path.

We used Spearman's rank correlation to explore relationships between variables we thought may influence planning (Δuniform.planned) and how the reality of the day affects the plan (Δplanned.actual). To minimize the chance of Type I error resulting from multiple pairwise tests, we chose to test four variables (elevation, size, coastline, and flight lines) for correlation with the two bait use scenarios and penalized the p-value by a factor of 8 ($P<0.0006$). The remaining explanatory variables were expressed as boxplots and compared with exploratory statistics.

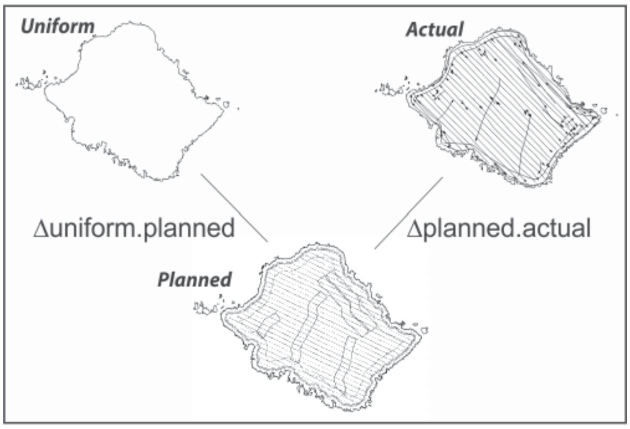

Fig. 1 Examples of bait use scenarios used and normalised percent change, delta, in bait use between scenarios. Uniform represents bait needed in an even distribution of bait across island area, planned represents bait needed based on predicted flights paths and overlap, and actual represents bait used.

Table 2 Explanatory physical and operational characteristics evaluated.

Physical characteristic	**Definition**
Country	Country operation was implemented in
Habitat	Tropical or temperate
Island type	Volcanic or coral atoll
Max. elevation	Maximum elevation in meters as a proxy for steep terrain
Size	Size of area to be treated (km^2)
Coastline	Length of coastline to be treated (km)
Operational characteristic	**Definition**
Target rate	Minimum application rate expected to be achieved on the ground, in some cases the coast and interior had different expected rates. The lowest expected rate was selected
Number of flight lines	The total number of flight lines flown
Supplemental treatment	Cliff, coast, or none to represent areas that received additional treatment above the target application rate
Coastal overlap buffer	True or false if the coastal overlap buffer strategy was employed to reduce bait into the marine environment

Variability in bait densities achieved

To evaluate the distribution of bait densities (kg/ha) we used GIS-derived bait density estimates from the 17 island blocks. For each island block, the area and estimated bait density of each polygon representing the bait density achieved on the ground from overlapping swaths was exported. Polygon areas representing areas smaller than 100 square meters (0.01 ha) were excluded as they were smaller than what is commonly considered a significant gap. For each island block, a bait density distribution was calculated to represent the total amount of island area treated at each bait density rate (e.g. 10 ha at 5 kg/ha) by summing the areas of treatment polygons at each bait density rate. To normalise bait density distributions across island blocks, values were represented as a percentage of the target application rate (e.g. 50% = half, 100% = target rate, 200% = twice target) and areas as a percentage of the total island area treated.

RESULTS

Factors associated with differences in planned and actual bait amounts used

The 10 projects analysed most often occurred in tropical regions (7 projects, 14 of 17 island blocks) and ranged in size from 8–2,900 ha, and 5–430 m in elevation. Target application rates ranged from 6 to 80 kg/ha, supplemental baiting used in seven island blocks, the coastal buffer overlap strategy used in 10 island blocks, and the number of flight lines flown spanned 9–1096.

On average, 20% more bait than the uniform scenario (Δuniform.planned) was planned for, and 16% more bait was used than planned (Δplanned.actual). The variables Δuniform.planned and Δplanned.actual showed no associations with the four factors investigated (elevation, size, coastline length and the number of flight lines) (Table 3). Median results of the 17 island blocks were 380 hectares, 214 flight lines, 80 m in elevation, and an 18 km coastline. Although no statistical correlation was evident among the island blocks and these factors, those blocks below the median showed a mean Δuniform.planned that was two to three times greater than blocks above the median, suggesting that compared to larger islands in our sample, planning on smaller islands typically identified proportionally more bait than a uniform distribution. The same trend is evident for Δplanned.actual with mean values for islands blocks below the median being one and a half times greater than above the median, showing that among our sample, smaller islands used proportionally more bait than planned for, compared to larger islands. Of the 14 tropical island blocks, five were on coral atolls, and these generally had a higher number of flight lines (M = 474.6, SD=215.1), compared to volcanic islands (M=121.1, SD=110.5).

Three island blocks conducted in the United States (Desecheo 2012, 2016, and Palmyra) had a negative Δplanned.actual, putting less bait on the ground than planned. The 10 blocks using the coastal buffer overlap strategy to reduce bait into the marine environment showed, on average, lower Δuniform.planned and Δplanned.actual compared to blocks that did not use this strategy.

Analysis of bait density estimates

On average, 5.1% (SD=3.8) of total island area received less than 50% of the target application rate and 0.8% (SD=1.6) of total island area received more than 400% of target (Fig. 2). The GIS derived bait density estimate polygons representing these areas had an average size of 0.12 ha (SD=0.2) and 0.03 ha (SD=0.04), respectively. Bait density estimates from each island block are shown in Fig. 3. Bait density estimates of less than 75% of the target application rate were visually compared against grids representing conservative minimum (0.01 ha) and average (0.1 ha) rodent habitats on tropical islands based on available literature (Fig. 4).

DISCUSSION

Factors associated with differences between planned and actual bait amounts used

From a statistical perspective, the sample size we used is considered small (n=17), and less than ideal because it was opportunistically collected (and not experimentally collated). From a conservation practitioners perspective, the opportunity to compare 17 different island blocks consistently is rare, and a positive example of collaboratively working to answer questions relevant across the island restoration field. A key result from our investigation is that projects planned to use 20% more bait than the hypothetical uniform application and used 16% more bait than planned, suggesting that simply estimating bait quantities by multiplying island area by target application rate is insufficient to judge how much bait will be needed. On average, the percent change between the planned amount of bait and actual bait used was less than the percent change between the hypothetical uniform

Table 3 Spearman's correlation and p-value of factors thought to influence bait use. Factors were considered associated with changes in bait use if Rho > 0.3 and p-value<0.006. Negative numbers represent a negative association (i.e. as one factor increases the other decreases) and positive numbers a positive association (i.e. as one factor increases so does the other).

Scenario	Factor	Rho	p-value
Δuniform. planned	Max. elevation	-0.193	0.458
	Size	-0.389	0.123
	Coastline	-0.288	0.262
	Flight lines	-0.311	0.224
Δplanned.actual	Max. elevation	-0.252	0.328
	Size	-0.212	0.414
	Coastline	-0.185	0.477
	Flight lines	-0.272	0.291

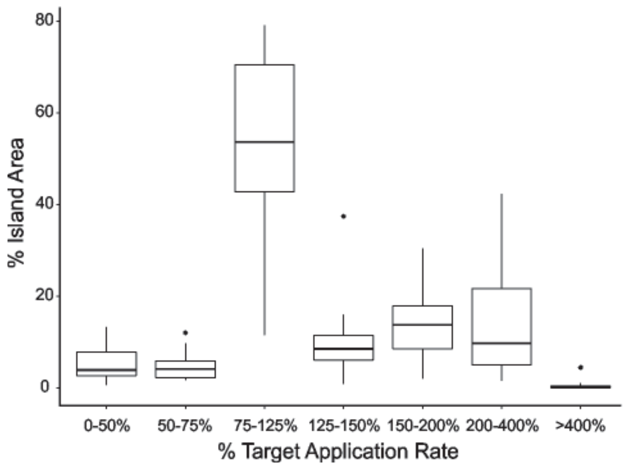

Fig. 2 Box plot of bait densities across projects represented as % of total island area treated vs % of target application rate.

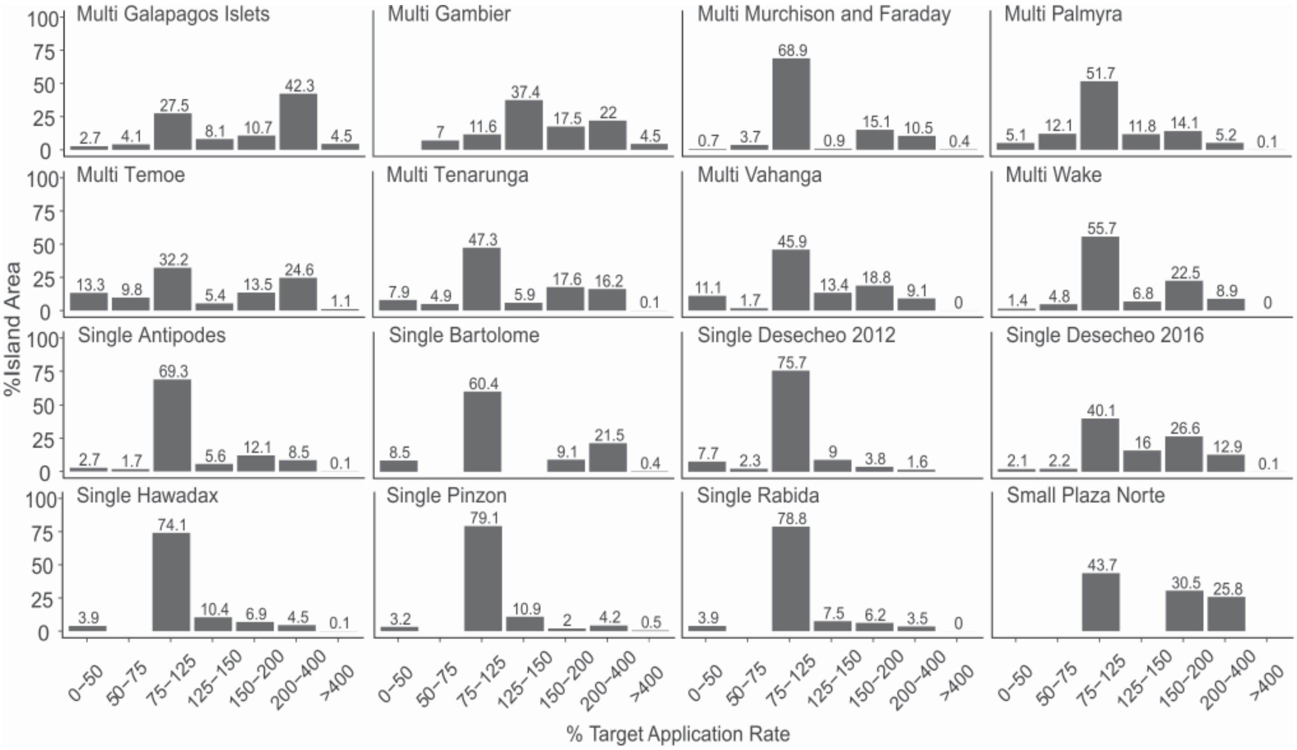

Fig. 3 Estimated bait density distributions per island block as % of total island area treated vs % of target application rate. Projects are grouped into multi (i.e. multiple treatment areas), single (i.e. single continuous treatment area > 100 ha), and small (i.e. < 100 ha).

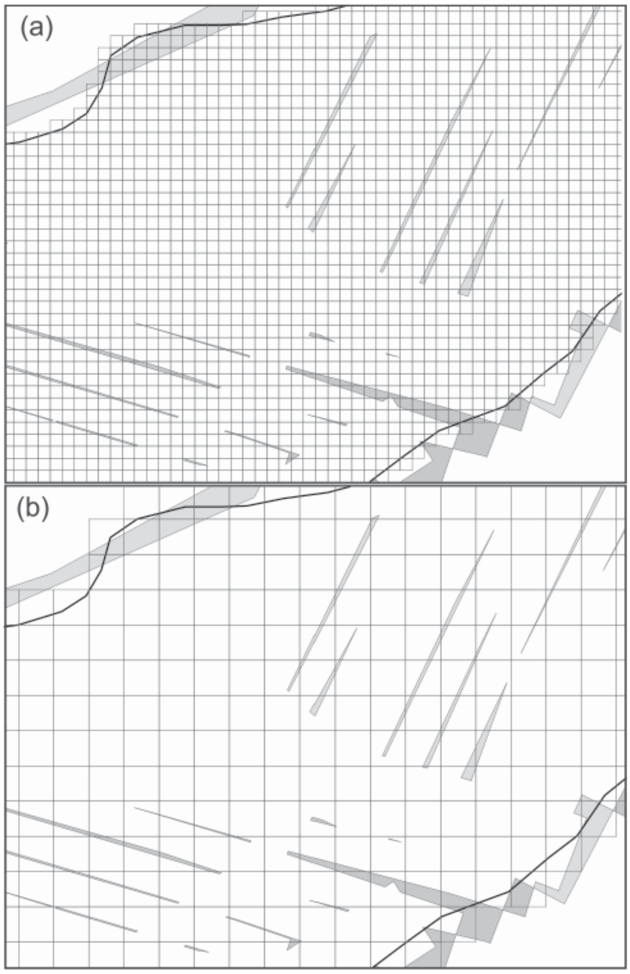

Fig. 4 GIS derived bait density estimates showing shaded areas less than 75% of the target application rate against potential (A) minimum (0.01 ha) and (B) average (0.1 ha) home range sizes from literature review.

amount of bait and actual bait used, suggesting that the aerial bait plans were more accurate at forecasting bait use but still underestimated actual bait required.

In general, smaller islands and islands with shorter coastlines, less elevation, or fewer flight lines planned to use, and actually used, a higher percentage of bait than projects on larger islands or those with more topography or flight lines. This suggests that small islands use proportionally more bait and that projects with fewer flight lines are more complex. While coastline length and maximum elevation were likely not differentiated enough from island size to detect a significant difference, the four-fold increase in the number of flights flown on tropical coral atolls, which have two coastal edges (lagoon and ocean), compared to volcanic tropical blocks suggests coastal complexity needs to be factored into planning. While the number of flight lines is also related to size, projects with fewer flight lines also have less room for error and could experience greater variability in bucket sow rates. Small islands may be able to improve bait applications, and reduce unplanned bait use, by employing strategies to increase the number of flight lines flown such as flying the parallel flight lines twice per application at half the target rate.

Perhaps the most interesting result was that projects implemented in the United States were the only projects, on average, to use less bait than planned. The United States has a complex regulatory environment, and aerial broadcasts are required to stay below permitted application rates. When implementing an eradication, projects in the United States had to balance the desire to achieve the desired target application rate with not exceeding the permitted application rate. Striking this balance resulted in projects using less bait than planned, particularly when the desired target rate was close to the permitted rate. This suggests that regulators should be involved early in the planning process so that regulatory approval can be sought to maximise project success. A single permitted application rate, such as the one designated on the bait product registration in the United States, is not necessarily appropriate for every project and, when appropriate, projects should develop site-specific operational strategies using the best available science. Regulatory bodies should review these strategies and recommended application rates on a case by case basis.

Bait application variability and consequences

It is noteworthy that, on average, 5% of the total island block area had bait density estimates less than half of the target application and 0.4% had bait density estimates greater than four times the target application rate (Fig. 2). This suggests a relatively high degree of precision in balancing the risk of failure (i.e. low localised bait densities) with unintended environmental impacts (i.e. high localised bait densities). Comparing the distributions of bait densities, larger (> 100 ha) single unit island blocks (i.e. those treated as a single contiguous unit: Antipodes, Desecheo, Hawadax, Pinzon, and Rabida) generally tended to have less bait density variability, with more than 60% of total island area near the target application rate, compared to smaller islands (< 100 ha) or island blocks consisting of multiple treatment units (Galapagos Islets, Gambier, Palmyra, Plaza Norte, Temoe, Tenarunga, Vahanga, and Wake) with less than 50% of total island area near the target application rate (Fig. 3). This is logical given that large or single unit island blocks have longer flight lines with which to "settle" into consistent sow rates and a smaller coast to size ratio resulting in fewer overlapping flights. Island blocks with multiple treatment units, particularly tropical coral atolls (Palmyra, Temoe, Tenarunga, and Vahanga), tended to have a higher percentage of total island area with localised bait densities more than twice the target application. These tropical coral atolls have more coastline for their size than other similarly sized islands, and thus the consequences of the flight line overlap necessary to minimise the chance of gaps near the coastline (i.e. higher localised bait densities) are more pronounced. This result underscores the trade-offs of ensuring complete coverage along complex coastlines.

Examinations of the two failed projects (Desecheo in 2012 and Wake) suggested low bait densities as one of the potential reasons contributing to failure (Derek Brown, pers. comm.). The bait density distribution of the failed 2012 Desecheo project shows a larger proportion of the island experienced localised bait densities less than half the target application rate during the first application (7.7%), compared to similar islands. Desecheo had a high abundance of non-target bait competitors (up to 833 crabs/ha) and bait availability plots in one habitat showed bait availability reaching zero within two to three nights (Will, et al., 2019). It seems likely that areas with localised bait densities less than half the target application rate would have experienced even less bait availability. On Wake, the bait density distribution shows a smaller proportion of the island achieved less than half the target application rate (1.4%) compared to similar islands, but bait density maps also show fewer flight lines extending up to the coastal edge and the presence of bait gaps on the beaches between the mean high-water mark and predominant vegetation. These observations may be instructive in improving the quality of future bait applications, suggesting that future applications consider applying additional bait (i.e. reapply) in areas with bait densities identified to be less than the target application rate and consider minimising the amount of untreated coastal edge on tropical coral atolls. These are areas where bait availability may be much less than expected and may not be immediately obvious when inspecting flight line maps. It is impossible to know if these improvements would have resulted in successful eradication attempts on Desecheo in 2012 and on Wake, but they would have removed questions about the quality of bait coverage as a possible contributor to eradication failure.

What is a significant biological gap?

Comparing actual bait densities achieved to the hypothetically smallest potential home range size can be instructive in informing risk tolerance for future operations. Rodent home ranges are highly variable, but amongst *R. rattus* have been recorded ranging from 0.012 to > 10 ha (Shiels, et al., 2016; Harper & Bunbury, 2015). It is in the smaller home ranges, particularly for breeding female rodents, where localised deficiencies in bait density present the highest risk of a rodent not being exposed to a lethal dose of bait (i.e. undertreated areas). We considered any areas that achieved less than 75% of the target application rate to be undertreated, which were generally the result of flight line deviation and were small (<0.1 ha) and irregularly-shaped (hundreds of meters long and <20 m wide). Despite their size and shape, these undertreated areas were still large enough to encompass most, if not all, of an assumed 0.01 ha potential minimum home range, but a minority of an assumed 0.1 ha average home range (Fig. 4). This suggests that, at the extreme, localised deficiencies in bait density could make bait less available than expected in entire potential rodent habitats where rodents have small home ranges.

Whether localised bait density deficiencies (i.e. undertreated areas) constitute biologically significant gaps is largely a consequence of toxicology, rodent biology and island ecology, and is project dependent. Ultimately, projects should anticipate that localised deficiencies in bait density are almost inevitable and determine what risk they pose to project success based on site specific conditions. In the presence of alternative foods and non-target bait competitors, or on islands targeting species with small home range sizes or multiple rodent species, areas that receive less than the target application rate could result in insufficient bait availability and constitute biologically significant gaps that pose a risk to project success. Where biologically significant gaps are a concern, project managers can either choose to increase the target application rate to increase the localised bait density of undertreated areas or set area size and application rate thresholds (i.e. 0.1 ha or larger with a bait density less than half the target application rate) to reapply bait.

Improving aerial application data analysis

Although GIS-derived bait density estimates provide a useful metric for identifying gaps or undertreated areas, they do have limitations and assumptions. A key limitation is they are not a direct measure of bait on the ground, and where possible on-ground measures of bait density, particularly with adequate sample size, can improve these data. Further, GIS-derived bait density estimates assume a) that flight speed is constant along the length of a flight line, b) bait pellet distribution across a swath is even, and c) wind has no impact on bait spread. A novel model called the Numerical Estimation of Rodenticide Dispersal (NERD) models these assumptions to generate a probability density function describing bait density and was successfully implemented on several projects in Mexico (Rojas-Mayoral, pers. comm.; Samaniego-Herrera, et al., 2017). These sorts of novel models are highly appropriate on high risk islands targeting species where smaller rodent home ranges may be anticipated (e.g. tropical islands where breeding may be expected). However, regardless of the analysis method used, managers are advised to trust in the broader rodent eradication principles and exercise caution to avoid overanalysing baiting data.

RECOMMENDATIONS

In summary, we propose the following recommendations to improve the planning and implementation of aerial broadcast applications for eradications.

Use high-resolution satellite imagery to estimate island size. Accurate estimates of operational area will improve estimates of the amount of bait needed and reduce the risk of having insufficient bait or the cost penalties of transporting and disposing of too much bait.

Create predicted flight plans to inform planning and estimate bait requirements. Multiplying island area by target rate is not an accurate estimate of bait needed. Including flight line overlap between parallel swaths and at the coastal boundary will improve accuracy of bait total estimates, reducing the chance of having too little bait. Additionally, predicted flight plans are useful in communicating the desired strategy.

Projects should plan for small islands to use more bait than anticipated and islands with complex coastlines to experience greater variability in bait densities. Coral atolls with lagoons have two coastal edges, which increases complexity, and should plan to use more bait and experience more areas of high localised bait densities. Small, complex projects should plan on ordering additional bait to treat gaps and compensate for areas of unplanned overlap.

Managers of projects on small islands should consider modifying operational strategies to reduce using additional bait. Increasing the number of flight lines by flying the island twice per application (with sowing rates adjusted to achieve the target rate), reducing the amount of bait in the bucket per load to reduce the percentage of island covered per flight, or conducting additional calibration runs to ensure consistency should be considered.

Projects should seek site-specific regulatory approval that maximises project success. A single permitted application rate is not sufficient to maximise success for all projects. Where appropriate, application rates should be tailored to site-specific conditions and be informed by the best available science. Additionally, to ensure clarity, projects should seek site-specific approval to implement predicted flight plans that describe the application rates and strategy needed to maximise project success. This is particularly relevant for projects implemented in the United States.

Use bait density estimates to identify areas treated below the target application rate. Tracking sowing rates achieved per load and assigning them to flight line data improves the understanding of bait coverage and allows managers to identify undertreated areas. Novel or high-risk projects should also consider using more fine scale bait density modelling approaches like NERD (Rojas-Mayoral, pers. comm.).

Projects should set gap size tolerances and application rate thresholds to match project risk variables. Clarify in advance of the project what constitutes a biologically relevant baiting gap based on what is known about the target species, island habitat, topography, and presence of non-target bait competitors. It is highly likely that a broadcast application will result in less than expected bait availability in the smallest potential rodent home ranges. For rodents with small home ranges, or tropical islands with high densities of non-target bait competitors, alternative food sources, or multiple rodent species a smaller gap size or higher application rate threshold may be warranted.

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REFERENCES

Bomford, M. and O'Brien, O. (1995). 'Eradication or control for vertebrate pests?' *Wildlife Society Bulletin* 23(2): 249–255.

Broome, K.G. (2009). 'Beyond Kapiti – A decade of invasive rodent eradications from New Zealand islands'. *Biodiversity* 10: 14–24.

Broome, K.G., Golding, C., Brown, K., Corson, P. and Bell, P. (2014). *Rat Eradication Using Aerial Baiting: Current Agreed Best Practice Used in New Zealand (Version 3.1).* Internal document DOC-29396. Wellington, New Zealand: New Zealand Department of Conservation.

Cromarty, P., Broome, K., Cox, A., Empson, R., Hutchinson, W. and McFadden, I. (2002). 'Eradication Planning for Invasive Alien Animal Species on Islands – the Approach Developed by the New Zealand Department of Conservation'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 85–91. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Garden, P., McClelland, P. and Broome, K. (2019). 'The history of aerial application of rodenticide in New Zealand.' In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 114–119. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Harper, G.A. and Bunbury, N. (2015). 'Invasive rats on tropical islands: Their population biology and impacts on native species'. *Global Ecology and Conservation* 3: 607–627.

Holmes, N.D., Griffiths, R., Pott, M., Alifano, A., Will, D., Wegmann, A.S. and Russell, J.C. (2015). 'Factors associated with rodent eradication failure'. *Biological Conservation* 185: 8–16.

Howald, G., Donlan, C.J., Galván, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21(5): 1258–1268.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

Martin, A.R. and Richardson, M.G. (2017). 'Rodent eradication scaled up: Clearing rats and mice from South Georgia'. *Oryx*:10.1017/S003060531700028X.

Parkes, J., Fisher, P. and Forrester, G. (2011). 'Diagnosing the cause of failure to eradicate introduced rodents on islands: Brodifacoum versus diphacinone and method of bait delivery'. *Conservation Evidence* 8: 100–106.

Pott, M., Wegmann, A.S., Griffiths, R., Samaniego-Herrera, A., Cuthbert, R.J., Brooke, M.d.L., Pitt, W.C., Berentsen, A.R., Holmes, N.D., Howald, G.R., Ramos-Rendón, K. and Russell, J.C. (2015). 'Improving the odds: Assessing bait availability before rodent eradications to aid in selecting bait application rates'. *Biological Conservation* 185: 27–35.

Russell, J.C. and Broome, K.G. (2016). 'Fifty years of rodent eradications in New Zealand: Another decade of advances'. *New Zealand Journal of Ecology* 40(2): 197–204.

Samaniego-Herrera, A., Aguirre-Muñoz, A., Bedolla-Guzmán, Y., Cárdenas-Tapia, A., Félix-Lizárraga, M., Méndez-Sánchez, F., Reina-Ponce, O., Rojas-Mayoral, E. and Torres-García, F. (2017). 'Eradicating invasive rodents from wet and dry tropical islands in Mexico'. *Oryx:* 10.1017/s0030605316001150.

Shiels, A.B., Pitt, W.C., Sugihara, R.T. and Witmer, G.W. (2014). 'Biology and impacts of Pacific island invasive species. 11. *Rattus rattus*, the black rat (Rodentia: Muridae)'. *Pacific Science* 68(2): 145–184.

Towns, D.R. and Broome, K.G. (2003). 'From small Maria to massive Campbell: Forty years of rat eradications from New Zealand islands'. *New Zealand Journal of Zoology* 30(4): 377–398.

Will, D., Swinnerton, K.J., Silander, S., Keitt, B., Griffiths, R., Howald, G., Figuerola-Hernandez, C. and Herrera Giraldo, J.L. (2019). 'Applying lessons learned from tropical rodent eradications: a second attempt to remove invasive rats from Desecheo National Wildlife Refuge, Puerto Rico'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 154–161. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

It's not all up in the air: the development and use of ground-based rat eradication techniques in the UK

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Abstract Eradication techniques using ground-based devices were developed in New Zealand in the early 1970s to target invasive rodents. Since then, different bait station designs, monitoring tools and rodenticide baits have been developed, and changes in field techniques have improved and streamlined these operations. The use of these techniques has been taken around the world to eradicate rodents from islands. Eradication technology has moved rapidly from ground-based bait station operations to aerial application of rodenticides. However, regulations, presence of and attitudes of island-communities and presence of a variety of non-target species precludes the aerial application of rodenticides on islands in many countries. As such, ground-based operations are the only option available to many agencies for the eradication of invasive rodents from islands. It is important to recognise that the use of ground-based operations should be a valid option during the assessment phase of any eradication proposal even in countries that can legally apply bait from the air; in many instances the use of ground-based techniques can be as economic and rapid. The use of ground-based operations can also facilitate opportunities for in-depth monitoring of both target and non-target species. Using examples of the techniques and developments used in five ground-based rat eradication operations from the UK demonstrates how these methods can be used safely and successfully around the world, even on islands in the order of hundreds of hectares and those with communities.

Keywords: bait station, ground-based, inhabited, island, rodenticide

INTRODUCTION

Albeit unanticipated, the eradications of rats from Rouzic Island, France in 1951 and Maria Island, New Zealand in 1960 were the first successful rat eradication operations on islands anywhere in the world (Towns & Broome, 2003; Lorvelec & Pascal, 2005; Howald, et al., 2007). These unintentional eradications spurred efforts in New Zealand to develop and perfect eradication techniques (Cromarty, et al., 2002; Thomas & Taylor, 2002; Towns & Broome, 2003). Between 1965 and 1986, New Zealand wildlife managers, ecologists and scientists used a range of experimentally designed operations to determine the best methods to consistently, successfully eradicate rats from islands (Cromarty, et al., 2002; Towns & Broome, 2003). Seabirds and other native species on islands are particularly vulnerable to invasive mammal species, particularly rats. The eradication of invasive mammals is considered the first step in island restoration and the subsequent recovery of native species and biodiversity. Since these early ground-based operations, rats (*Rattus rattus, R. norvegicus, R. exulans*) have been successfully eradicated from over 400 islands ranging in size from 1 to 12,850 ha, around the world, using the full gamut of methods and technology (Moors & Atkinson, 1984; Atkinson, 1985; Towns & Broome, 2003; Howald, et al., 2007; Jones, et al., 2008; Parks & Wildlife Service, 2008; Parks & Wildlife Service, 2014, DIISE, 2015). Of these rodent eradications, the largest ground-based rat eradication operation, was on Langara Island in British Columbia at 3,100 ha, and the largest ground-based rat eradication in the United Kingdom (UK) was on the Isle of Canna at 1,300 ha (Taylor, et al., 2000; Bell, et al., 2011; DIISE, 2015).

Techniques and technology developed in those early eradications have since moved on from ground-based hand-broadcast and bait station operations to aerially-applied rodenticide operations and these have now been used across the globe. Advances in, and alterations to, techniques and tools have streamlined ground-based operations. Lessons learnt from each eradication have improved the next operation. However, in several countries, including the United Kingdom (but excluding the United Kingdom Overseas Territories), methods to eradicate rats are restricted to ground-based methods. The presence of critical non-target species, sensitive habitats, island communities and legislative requirements have restricted methods and tools for island eradications in these countries. This paper describes the history and development of ground-based rat eradications using bait stations in the United Kingdom using five eradication operations as examples and covers lessons learnt and how local communities have been involved.

INVASIVE *RATTUS* SPECIES ON UK ISLANDS

Both black (*Rattus rattus*) and brown (*R. norvegicus*) rats are present in the UK (Nowak, 1999; Long, 2003). Black rats were presumed to have been introduced by the Romans (c. 110 AD) and the brown rat via shipping between 1720 and 1728 (Thomas, 1985; Corbet & Southern, 1977; Yaldwen, 1999; McCann, 2005; Parslow, 2007). Brown rats were first recorded in the Isles of Scilly in 1728 after several shipwrecks occurred that year (Thomas, 1985; Parslow, 2007). Although the brown rat displaced the black rat throughout most of the UK, black rats can still be found in a small number of locations, particularly port cities such as London, Edinburgh and Falmouth (Matheson, 1962; Bentley, 1959; Twigg, 1992; Long, 2003). The brown rat is still present on 56% of UK islands over 100 ha (Long, 2003).

Rats are known to have very detrimental effects on seabird populations through predation and competition for food and habitat, causing local and global extinction of birds on islands throughout the world (Moors & Atkinson, 1984; Atkinson, 1985; Courchamp, et al., 2003; Towns, et al., 2006; Jones, et al., 2008; Bell, et al., 2016). The eradication of introduced predators from islands has become one of the most important tools in avian conservation in recent times and, with an initial investment, significant long-term restoration benefits such as increased productivity and population increases of seabirds and other native species as well as the establishment of new seabird species can be achieved. The eradication of rats from seabird islands is recognised as a prerequisite for the restoration of seabird populations (Atkinson, 1985; Moors, et al.,1992).

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Seabird populations on many UK islands have been recorded in decline and in at least four cases rats have been identified as one of the contributing factors for these declines (Campbell, 1892; Brooke, 1990; Mitchell, et al., 2004; Brooke, et al., 2007; Swann, et al., 2007; Dawson, et al., 2015; Hayhow, et al., 2017). Many species such as puffin (*Fratercula arctica*) which is listed as threatened due to their declining population status (IUCN, 2017), Manx shearwater (*Puffinus puffinus*) and the European storm petrel (*Hydrobates pelagicus*) may have limited distribution due to the impacts of, and predation by, rats (Heaney, et al., 2002; Mavor, et al. 2008). Currently, the majority of the UK puffin and all European storm petrel populations nest on rat-free islands (Mavor, et al. 2008, Ratcliffe, et al., 2009). The protection and enhancement of UK seabird breeding habitat has been recognised as an important conservation priority, including under international conservation agreements (Brooke, et al., 2007; Ratcliffe, et al., 2009; Dawson, et al., 2015; Thomas, et al., 2017a).

Rat eradications have occurred on over a dozen islands around the UK with brown rats being the most common target species (Bell, et al., 2011; Thomas, et al., 2017a; Bell, et al., 2019a; Pearson, et al., 2019). Black rats have been targeted on Lundy Island and the Shiant Isles (Lock, 2006; Appleton, et al., 2006; Thomas, et al., 2017a; Main, et al., 2019). Many of the eradications have occurred on islands with permanent staff or the presence of small communities (Bell, et al., 2011; Bell, et al., 2019a; Pearson, et al., 2019). These operations demonstrate how ground-based eradication techniques can be utilised on both inhabited and uninhabited islands around the UK.

Pre-1998: the early eradication operations

Despite an early attempt to eradicate rats from Ailsa Craig in 1925, the first documented successful rat eradication did not actually occur in the UK until 1968 on Cardigan Island in Wales (RSPB, 1924; RSPB, 1925a; RSPB, 1925b; Johnstone, et al., 2005; Thomas, et al., 2017a). This makes the UK the first country to intentionally undertake a rat eradication operation anywhere in the world. Four other rat eradications occurred between 1968 and 1998; Inchgarvie (Firth of Forth), Scotland in 1990, Ailsa Craig, Scotland in 1991, Handa Island, Scotland in 1997 and Puffin Island, Wales in 1998 (Ratcliffe & Sandison, 2001; Zonfrillo, 2001; Zonfrillo, 2002; Johnstone, et al., 2005; Stoneman & Zonfrillo, 2005; Thomas, et al., 2017a). Warfarin was the primary active ingredient used in each of these eradications with difenacoum used as a secondary option in the Puffin Island rat eradication (Ratcliffe & Sandison, 2001; Zonfrillo, 2001; Zonfrillo, 2002; Stoneman & Zonfrillo, 2005). All of these early eradications used ground-based methods, but focused on applying bait in holes, burrows, under rocks and vegetation and in isolated wooden bait stations or under inverted fish bins, rather than in a systematic grid pattern (Ratcliffe & Sandison, 2001; Zonfrillo, 2001; Zonfrillo, 2002; Stoneman & Zonfrillo, 2005; Thomas, et al., 2017a).

This method of baiting made it difficult to monitor bait consumption by rats and non-target species. There were no accurate records of bait take by rats or other species from any of these operations (Ratcliffe & Sandison, 2001; Zonfrillo, 2001; Zonfrillo, 2002; Stoneman & Zonfrillo, 2005; Thomas, et al., 2017a). Monitoring was limited: in most cases it didn't occur; used chewsticks across the island immediately following the eradication (it has been noted that chewsticks can be difficult to interpret sign accurately); or was determined by the recovery of the seabird or rat populations without any quantifiable measures (Zonfrillo, 2001; Ratcliffe, et al., 2009; Thomas, et al., 2017a). In the case of Inchgarvie and Puffin Islands eradication was not confirmed until years after the operation. Unfortunately, there have been recent reports of rats on Inchgarvie and rats reinvaded Handa in 2012 (Thomas, et al., 2017a).

The later operations (post-1999)

The use of toxins and the risks these presented to non-target species and the environment led to the development of Best Practice and Standard Operating Procedures for eradication operations in New Zealand in the 1990s and these documents are revised as new techniques and tools are developed (Cromarty, et al., 2002; Broome, et al., 2011). Robust protocols for eradication operations included detailed planning, operational requirements, implementation protocols, monitoring guidelines and biosecurity requirements (Cromarty, et al., 2002; Broome, et al., 2011). These best practice and standard operating techniques developed in New Zealand were followed and adapted during the UK eradications undertaken by Wildlife Management International Ltd (WMIL).

Five major eradications directed by WMIL have occurred in the UK since 1999; Ramsey Island, Wales (brown rat) in 1999/2000, Lundy Island, England (black and brown rat) in 2002–2004, Isle of Canna, Scotland (brown rat) in 2005/2006, St Agnes and Gugh, Isles of Scilly, England (brown rat) in 2013/2014 and the Shiant Isles, Scotland (black rat) in 2015/2016. In addition to these five sites, eradication attempts have also been made on Looe Island in 2006, the Calf of Man in 2012 and Caldey Island in 2015, which have not been included here because Looe Island was reinvaded by rats three years later and the Calf of Man and Caldey Island eradications are still on-going (Thomas, et al., 2017a).

These five eradications used ground-based techniques with bait stations placed out across the islands on either 25 m × 25 m, 25 m × 50 m, 50 m × 50 m, 90m × 90 m or 100 m × 100 m grids depending on the target species and type of habitat or risk areas. The smaller grid sizes (between 25 and 50 metres spacing) were used to target black rats and the larger grid sizes (between 50 and 100 metres spacing) used to target brown rats, with the smallest spacings used in high risk areas (such as around properties, seabird colonies, wharves, farms and restaurants).

A simple yet effective bait station design has been used in each of these five eradications in the UK. Although a range of commercially available lockable stations have been used in selected locations (e.g. residential homes, farm buildings, schools, etc.) during these eradications, and for on-going biosecurity to reduce the risk to the public, particularly children and the possibility of tampering with these long-term stations, the main bait stations were made from corrugated drainage pipe. This design is cost-effective and widely available. For the 1999/2000 Ramsey Island rat eradication, 500 mm lengths were used. However, these stations were found to be too short as they allowed carrion crows (*Corvus corone*) access to the bait. The stations were made longer by adding 250 mm lengths to one end. The standard length for each bait station in all subsequent eradications was 750 mm long with an access hole cut in the centre for placement of the bait (Fig. 1). This access hole is covered with a short section of drainage pipe. During the 2002–2004 Lundy Island rat eradication, crows learnt to flick the lids off the stations to reach the bait, therefore another length of wire was put around the centre of the station to hold the lid tightly in place. This "crow clip" became standard on all bait stations on any island with either carrion crows, hooded crows (*C. cornix*) or ravens (*C. corax*) present (Fig. 1).

Technological advances in GPS and GIS-linked systems helped streamline the positioning of bait stations during the grid establishment stage of eradications, as well

Fig. 1 Example of the main bait station in position with the crow clip holding the central lid in place, as used in the five ground-based eradications in the United Kingdom that were directed by Wildlife Management International Ltd. [Credit: Elizabeth Bell, WMIL]

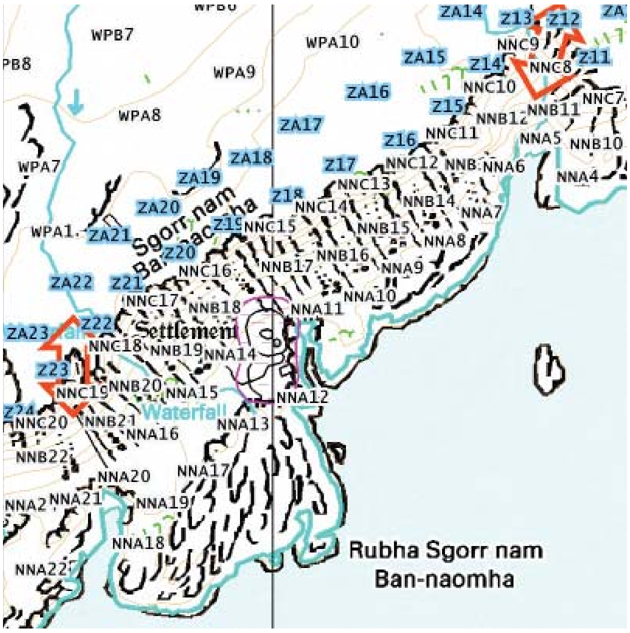

Fig. 2 An example of a detailed bait station map as used by eradication teams during the Isle of Canna operation. Where alphanumeric codes related to bait station positions (e.g. WP = West Plateau, A = line A, 9 = bait station 9; Z = Boundary line Z (two lines of stations at the top of the cliff section above the coastal slopes), 19 = bait station 19; NN = Nunnery, B = line B, 6 = bait station 6), double ended red arrows = safe access routes up or down to the coastal slope areas, pink shaded areas = important archaeological site (e.g. The Nunnery).

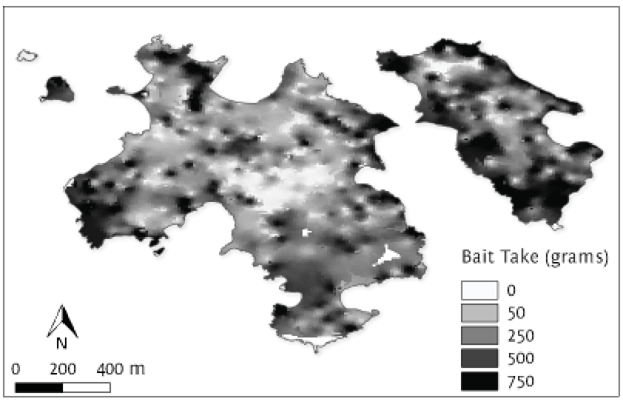

Fig. 3 Example of a heat map of bait take (g) by rats using the results from the St Agnes & Gugh brown rat (*Rattus norvegicus*) eradication.

as monitoring the level of bait take by rats and non-target species. Detailed maps can be produced for the eradication team that can give additional information such as sensitive sites like archaeological structures, location of rare plants, seabird colonies and white-tailed sea eagle (*Haliaeetus albicilla*) nesting sites and locations of access points for steep terrain where more care needs to be taken (Fig. 2). This intimate knowledge of the stations and island make it easier for the eradication team to monitor bait take by rats as the data can be linked to the specific station and activity levels can be recorded on the spatial map. These maps created by the GIS-linked database offer the team the opportunity to monitor the decline in rat numbers throughout the operation and could allow the eradication personnel to react instantly to hotspots or problem areas on the island. Specific bait take information can lead to detailed activity which shows bait take by rats throughout the operation and detailed heat maps showing complete bait take by rats at the end of an eradication (Fig. 3).

Cereal-based bait blocks (containing the anticoagulant diphacinone, bromadiolone or difenacoum) were used. Each eradication used one bait formulation as the main bait with a second option available to target ‘fussy’ (i.e. those rats that will not eat the primary bait for whatever reason) or surviving rats. This gives the option to adapt the project if rat behaviour or taste preference becomes an issue during the eradications. This has been shown to be important in certain eradications as demonstrated by the Isle of Canna brown rat eradication where the last surviving rat was targeted successfully using the alternative bait. Bait was placed loose in the bait stations for the first three weeks of the eradication operation. This allowed rats to cache bait in burrows for feeding themselves and any breeding females. Once bait take has reduced, bait is wired securely into the stations (Fig. 4) and any rat sign on these blocks is used to identify the presence of a surviving rat or monitor high risk areas, such as seabird colonies or farm buildings.

Although aerial application operations generally have a range of higher implementation costs compared to ground-based operations due to the requirement of a helicopter, sowing bucket and need for ground crew, engineer and other legal requirements for use of aircraft, the implementation time of the operation is often reduced compared to a ground-based operation. Except for the Lundy Island eradication (which took a second winter),

Fig. 4 Example of the main bait station in position showing the open bait station (with the lid off) with the bait wired in place, as used in the five ground-based eradications in the United Kingdom that were directed by Wildlife Management International Ltd. [Credit: Elizabeth Bell, WMIL]

rats were successfully targeted within 21 to 64 days (Bell, et al, 2011; Bell, et al., 2019a; Bell, et al., 2019b; Main, et al., 2019).

The use of rodenticide baits to complete eradications has enabled strong relationships to be established between bait manufacturers, eradication operators and local agencies in the UK. This has enabled open and in-depth discussions about the bait in regard to problems with formulation, taste and longevity that were identified during eradication operations. Issues such as the bait blooming (i.e. swelling and splitting after moisture on the bait) or rapidly going mouldy in the Lundy eradication were relayed to the manufacturers who altered the wax content for later operations. This meant the bait became much more robust in the damp winter conditions during the later Scottish operations, reducing the overall bait quantity required for those eradications.

European Union (EU) regulations require Bitrex™ (denatonium benzoate) or an alternative bittering agent to be added to all rodenticides to deter human consumption. Rats are not intended to be put off by Bitrex™, although research suggests that some rats can detect it even at very low concentrations, and preferentially choose bait that does not contain Bitrex™ (Veitch, 2002). Three rats actively avoided bait containing Bitrex™ on Lundy Island and, by working with the UK bait manufacturer, dispensation for a small amount of Bitrex™-free bait was obtained and was used to successfully target the rats at those sites (Bell, 2004). Despite the bait manufacturer disputing the fact that rats could detect and avoid Bitrex™ bait, they were open to experiment, assess the issue and work together with WMIL and Royal Society for the Protection of Birds (RSPB) towards a solution. Without the engagement of the bait manufacturer from the beginning of the project and open and frank discussions about the possibility of this issue with Bitrex™, the Lundy operation could have failed.

There have been recent regulatory changes to the purchase and use of rodenticides in UK. The Health and Safety Executive (HSE) require reassurance that biocide products can be used without unacceptable risk to wildlife and other non-target species and in July 2015 implemented the UK Rodenticide Stewardship Scheme. This scheme covers all rodenticide products sold to, and used by, professionals when applied outside buildings and in open areas and operates under a Code of Best Practice developed by the Campaign for Responsible Rodenticide Use (CRRU) group (CRRU, 2015). All professionals must have proof of competence at the point-of-sale for rodenticide baits (i.e. have completed certification for rodenticide control and/ or eradication by completing an approved training course) as well as comply with the best practice. These regulations generally relate to urban control operations, pest control operators and farmers, but eradication programmes must also follow these regulations. RSPB, in conjunction with CRRU, have developed an eradication-specific registered training course under the UK Stewardship Scheme.

Ground-based operations facilitate longer, wide-scale monitoring compared to aerial operations; not only using the bait itself, but also using a range of monitoring tools such as flavoured wax blocks, soap, tracking tunnels and trail cameras. Monitoring can be established at the same locations as the bait stations, as well as between the bait stations to intensify the scope of monitoring and ensure every micro-habitat is covered. Having non-toxic monitoring devices out in the open (pegged to the ground) and using a range of options gives more chance for any surviving rats to interact with at least one type of monitoring tool. This can also identify if a percentage of the rat population or rats at a specific location are avoiding the bait stations for any reason. This intensive effort can be used to detect any survivors and the operator can adapt the eradication to successfully target those last individuals. WMIL developed a range of flavoured wax blocks that have proved to be very effective in detecting the presence of surviving rats at the final stages of eradication (Fig. 5). These blocks have been freshly produced by the eradication team on-site to a standard recipe as the operation progresses. This flavoured wax recipe has been widely shared amongst the eradication industry.

This period of intensive island-wide monitoring allows the eradication operators to be much more confident that the eradication has been successful prior to leaving the island. By being able to detect and respond to surviving rats immediately, this reduces the likelihood of eradication failure (as any rat that is detected during this period can be targeted) and thus the need for a second eradication attempt (which can cost as much as the original operation). This intensive monitoring period in these five UK operations occurred for up to four months, depending on the size of the island and time required to initially target the rats during the baiting phase. Additional monitoring is completed at least quarterly for two years prior to the intensive final check phase and rat-free declaration following standard international eradication protocols for temperate operations (Broome, et al., 2011).

The use of volunteers has been an asset to these five UK eradications by giving passionate conservationists the chance to be involved in a project they feel strongly about, increasing the national (e.g. RSPB) capacity in eradication methodology, and engagement with the local communities. However, the use of volunteers can reduce the awareness of managers, decision makers and funders of the true cost and effort required to complete ground-based eradications.

The costs of these five ground-based eradications ranged from £76,000 up to £900,000, including planning, implementation, key species pre-and post-eradication monitoring, monitoring for survivors or incursions for two

Fig. 5 Examples of flavoured wax as used for monitoring in the five ground-based eradications in the United Kingdom that were directed by Wildlife Management International Ltd. [Credit: Jaclyn Pearson, RSPB]. Where the left (blue) block is aniseed flavour, centre (brown) block is chocolate flavour and right (fawn) block is peanut flavoured and each block is pegged to the ground with a piece of fencing wire and marked with a short piece of flagging tape for visibility.

years post-eradication and confirmation monitoring ('final check') prior to the declaration of rat-free status (Dr R. Luxmoore, NTS, pers. comm.; P. St Pierre, RSPB, pers. comm.; Lock, 2006).

There can be difficulty associated with accurately recording the entire costs of eradications; in many cases reported costs do not include in-kind or match funded expenses by the agencies involved (National Trust for Scotland, RSPB, etc.). In many cases, it can be difficult to accurately record these costs against the eradication operation as they relate to administration and corporate expenses.

BEST PRACTICE FOR ERADICATIONS

It has long been recognised that every island is different when it comes to planning and implementing an eradication operation. As such, although the NZ best practice gave an important starting point for the UK operations, it needed to be adapted for the local situation to become more relevant and effective, particularly in regard to local legislation and animal welfare regulations.

The RSPB, in partnership with UK-based governmental and non-governmental organisations working in island restoration, with input from international experts in this field produced The UK Rodent Eradication Best Practice Toolkit which is hosted on the Great Britain Non-Native Species Secretariat website (Thomas, et al., 2017b).

This toolkit was developed as an advisory resource to provide systematic planning and implementation protocols for ground-based rodent eradications and biosecurity in the UK (Thomas, et al., 2017b). It aims to give UK organisations technical advice on eradication methodology as well as an eradication project management framework to enable greater confidence in achieving island restoration goals in invasive rodent management projects in the UK (Thomas, et al., 2017b).

THE ROLE OF COMMUNITIES

The majority of eradications around the world have occurred on uninhabited islands and it is thought that islands with significant human populations, unreceptive communities or occurrence of livestock and domestic animals are unlikely to be feasible for rat eradication (Campbell, et al., 2015). However, because invasive species are also a problem on inhabited islands, such eradications must be considered. A lack of public awareness about invasive species impacts and misunderstanding of eradication techniques from island communities are thought to have been responsible for the opposition of proposed eradications on inhabited islands around the world (Bryce, et al., 2011). The importance of the engagement and inclusion of local communities has been highlighted in a number of recent eradication and research projects, especially in regard to risk and benefit analysis and to ensure a suitable environment for eradication projects to proceed can occur (Bryce, et al., 2011; Eason, et al., 2008). Respect for the attitudes, and safety, of local communities needs to be a priority in any eradication planned for inhabited islands. The support and agreement by the community to proceed with an eradication is vital for any project on an inhabited island. This is particularly important as access into all properties is vital to effectively carry out an eradication. Involving the residents in the concept, planning, implementation and on-going biosecurity of the island was recognised as the only way such an eradication could have occurred on the islands in the UK.

Considerations to how the community view the environment, how they think the proposed eradication will affect them and other social science considerations need to be assessed for eradications planned for inhabited islands. Most importantly, all aspects of the eradication should be discussed with the community in the early stages of the proposal. Unlike eradication operators, most members of the public do not have any knowledge of the principles and techniques of eradication, particularly in regard to rodenticide choice and operational procedures. It is important that each community member understands these aspects and how they will personally be affected by the day-to-day operational requirements.

As there were staff or small communities present at four of the five previously mentioned UK eradications, almost all recent operations undertaken in the UK have had to work closely within these communities and have had to adapt to the issues and technical challenges the presence of people has on the eradications. During each of these eradications, WMIL and the local project partner worked closely with the landowner, staff and residents to understand and address concerns and questions about the operations. Where the operation occurred on staffed islands, the decision to complete an eradication had already been made by the main project partner concerned and much of the consultation with staff on the islands had already been completed by the management prior to the operation. Resident staff were generally supportive of the eradication and often viewed the eradication operational team as temporary, but separate, staff members. In comparison to those islands with resident staff, WMIL and RSPB recognised the importance of the engagement of the 85-person resident community on St Agnes and Gugh in the Isles of Scilly and started this engagement process early for the eradication of brown rats (Bell, et al., this issue a, Pearson, et al., 2019). The success of the St Agnes and Gugh eradication (Isles of Scilly Seabird Recovery Project, IOSSRP) showed how the community-based approach that was designed to develop local networks and use existing community structures to build support for the project worked extremely well. The vision and benefits of the project were shared by the community and the residents were part of the decision-making process and management of the project.

An open and transparent operating system has worked well in all these five previously mentioned eradications in the UK. Information covering details on rodenticide type, bait station design, anticoagulant poisoning symptoms and treatment, contact numbers and project management was provided to all residents, stakeholders and interested parties. The project team was permanently present on each of the islands throughout the eradication to implement the operation, answer any questions and deal with any issues. Project updates were provided to the community and stakeholders each week, which gave the residents the opportunity to observe the operational procedures and results as the eradication proceeded. Real-time bait-take maps were provided as part of this process. A 24-hour contact telephone number was provided for immediate response to any issues that a resident may have.

BIOSECURITY

With the eradication of rats from islands, the priority is to ensure that they do not become re-established. Biosecurity is a critical aspect of any eradication and should be designed, implemented and tested prior to the completion of the eradication and departure of the eradication team. Prevention of an accidental rat reintroduction should be the primary aim. Precautions need to be taken not only in obvious situations such as with visitors or boat movements, or when high-risk items like stock feed or hay are being delivered to the island, but also when the risk may be mistakenly thought to be negligible.

The long-term legacy of these five UK eradication projects was important to the implementing agencies involved as well as the communities and agency staff on the island. As such, practical biosecurity strategies were established for the community and supporting agencies; measures that have been designed to reduce the risk of rats being reintroduced to a minimum, without being a hindrance to the daily lives of the staff, community or visitors to the island. A range of biosecurity strategies were proposed to the residents or agency staff on each island and, following discussions about the protocols of each strategy, suitable measures for each island were selected and implemented. Public awareness and education leaflets have been developed for every eradication to ensure that the public are aware of the rat-free status of each island and ways they can assist in keeping the islands rat-free. Residents and staff members from the project partners have been trained in all relevant biosecurity measures and they will maintain regular monitoring checks on the islands in perpetuity. Funding for on-going biosecurity has been provided by partner agencies and completed by staff or in the case of St Agnes and Gugh, funds will be provided by the community through fundraising and grants (Pearson, et al., 2019). In some instances, such as on St Agnes and Gugh, community coordinators will maintain liaison between the residents and the supporting partner agencies (Pearson, et al., 2019).

DISCUSSION

Rat eradications have been undertaken on islands around the world for the past 65 years and in the UK for the past 50 years. International rat eradication projects over this time have used a range of methods but most recently focused on the aerial application of rodenticides. However, due to legislative limitations upon the outdoor-use of rodenticides and application methods, and although derogations can be issued to allow aerial operations, ground-based methods are likely to remain the predominant rat eradication technique in the UK (and other European counties). Developments from five eradications in the UK have streamlined operating procedures and eradication techniques for the next eradication. Using plastic corrugated drainage pipe as the main bait station type has enabled the design to be adapted to exclude large or problematic non-target species such as rabbits and crows. The positioning of bait stations using GPS and GIS-linked systems has streamlined recording bait take by rats and non-target species and enabled this to be monitored in real time. Constant monitoring throughout the operations starting with the bait take and progressing through to using a range of monitoring devices, such as flavoured wax, allowed for each operation to adapt to deal with high risk areas or ‘fussy’ rats to maximise the likelihood of eradication success. This intensive level of monitoring allows any issues that may arise with bait to be addressed directly with the manufacturers and rectified early in the operational timeframe.

Ground-based eradications have been completed on islands ranging in size from <1 ha to 3,100 ha (Taylor, et al., 2000; DIISE, 2015). Although an island’s size and terrain may prevent a ground-based bait station operation being completed, it would be perfectly feasible to eradicate rats from even larger inhabited and uninhabited islands assuming there were enough resources (including staff and funding) and commitment and support from all involved. The feasibility assessment for any proposed eradication needs to investigate the costs and benefits of all possible methods before deciding on the final operational techniques. In many cases, a combination of aerial and ground-based operations may also be suitable or preferred by communities on large inhabited islands, as shown by recent eradication plans such as for Lord Howe Island (Wilkinson & Priddel, 2011; Walsh, 2019).

Over 85% of rat eradications around the world have been completed on uninhabited islands (n = 721 out of 820 eradications; DIISE, 2015). However, many are now either being investigated or planned for islands with resident communities (Oppel, et al., 2011; Russell & Broome, 2016; Stanbury, et al., 2017). Eradications on inhabited islands raise social, economic, conservation and technical challenges for the operation (Moon, et al., 2015). The experience in the UK shows that to ensure an island restoration project runs successfully the support and agreement from the community must be secured. The community must share the project’s vision and feel that they are one of the beneficiaries. To do this, they will need to be included and play an integral role in the decision-making process, planning preparation and implementation and management of the project. In this way, the legacy of the project will be much stronger. Those proposing the eradication need to ensure that the community is aware of the effects of invasive rats on the native biodiversity of their island and how the proposed eradication can benefit those species as well as explaining the process of the eradication operation itself. However, project partners and eradication operators also need to realise that for a number of residents the biodiversity and environmental reasons to eradicate rats may be of no interest; as such, social and economic benefits should also be outlined during the planning stages as these may be more important to the communities themselves. It is important for operators to realise that communities may not have the same understanding of eradication processes and each aspect of the project may have to be explained.

The larger the community the longer, potentially, the project managers will need to ensure that the residents are all at the same position of understanding through the various stages of the project. Archipelagos or groups of islands bring additional stakeholders and interested parties that need to be engaged compared to single islands. From my experience, ten years is not an unreasonable timescale depending upon the starting point, the value placed upon seabirds by the community and the strength of the project partnership. In my view, and in agreement with others such as Moon et al. (2015), the ongoing consultation and communication with the local community and wider stakeholder groups during any eradication is essential.

As the need to prioritise islands for restoration has increased, the requirement of understanding and quantifying the costs of eradications has also increased (Martins, et al., 2006; Holmes, et al., 2015). Although general costs for eradications can be estimated if the size of the island and target species are known, and it appears that costs increase with the size of island, there are other costs from application method, permits, non-target mitigation, and biodiversity monitoring that need to be factored into an eradication operation (Martin, et al., 2006; Holmes, et al., 2015). This information is vital to be able to accurately determine the complete costs for future eradications and it is important that project costs are reported.

The defining factors underpinning the success of the eradication operations on inhabited UK islands were the professional management of the eradication, dedicated and passionate volunteer involvement, efficient and systematic monitoring, adapting to local conditions and ensuring a community-inclusive approach.

This model of consultation, engagement and community-involvement developed on these inhabited islands eradications in the UK can offer valuable information, advice and direction for eradication operations planned on islands with larger communities in the UK and around the world. The eradication of brown rats from St Agnes and Gugh could be used as a valuable education tool to show other communities that it is possible to safely eradicate rats and implement suitable biosecurity measures to reduce the risk of reinvasion without impacting on the lives of the residents, as reported by Pearson, et al. (2019). This model, and future techniques developed during other eradications on inhabited islands, will be even more important if restrictions on application measures and outdoor-use of rodenticides expand to countries outside of the UK. It is important for eradication operators to realise that even if aerial application methods are possible at the location, the community on the island may not approve or permit that type of method. As such, the use of ground-based bait station techniques will have a vital part to play and this option should be assessed as part of any original feasibility assessment.

Island restoration on UK islands has led to the dramatic recovery of seabird populations. Manx shearwaters on Ramsey and Lundy Islands have increased nearly ten-fold in the ten to fifteen years since the eradication of brown and black rats and the recolonization of European storm petrels and other small burrowing species has been recorded after long absences (Brown, et al., 2011; Morgan, 2012; Booker & Price 2014; Bell, et al., 2019b). These types of results have helped develop a legacy for many of the projects, with the residents and agency personnel on the islands committing to and doing their part to maintain important biosecurity measures. These results can also be used to help explain the benefits of completing this type of eradication project on other islands, even those with larger communities or a complex of target species. Providing safe breeding habitat and creating and then maintaining rodent-free status at important island sites, will be an important part of the long-term legacy of protection for UK seabirds.

It is important that when eradication projects are being designed and assessed that operators and project partners factor in on-going biosecurity after the completion of the project, particularly in relation to equipment, capacity and long-term funding requirements. It is one of the most vital aspects of an eradication project and agencies must recognise the requirement that biosecurity is required *in perpetuity*. For eradications that occur on inhabited islands, this makes the engagement of, and commitment from, the communities to undertake biosecurity measures, even more important to ensure the legacy of any eradication project.

Detailed prioritisation exercises such as Brooke, et al. (2007), Ratcliffe, et al. (2009) and Stanbury, et al. (2017) have identified a number of UK and UK Overseas Territories' islands as being pre-eminent sites for rat eradication because of their importance to seabirds. Twenty of the 25 islands identified in the most recent prioritisation exercise have resident human populations which increases the challenges for any eradication proposed for those sites (Stanbury, et al., 2017). One of the most important lessons identified by completing eradication operations on inhabited islands is that the community needs to be engaged as early as possible, preferably in the concept and development process. As important, all stages of the eradication need to be completely open and transparent, with community members involved throughout the implementation of the project and into the future to ensure the sustainability of the on-going biosecurity for the island. The newly developed Best Practice for UK islands (Thomas, et al., 2017b) which has built on all the lessons learnt from these eradications that have occurred over the past 50 years in the UK should help make these future eradication operations more likely to succeed on both uninhabited and inhabited islands.

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REFERENCES

Appleton, D., Booker, H., Bullock, D., Cordrey, L. and Sampson, B. (2006). 'The seabird recovery project: Lundy Island'. *Atlantic Seabirds* 8: 51–59.

Atkinson, I.A.E. (1985). 'The Spread of Commensal Species of *Rattus* to Oceanic Islands and their Effects on Island Avifaunas'. In: P.J. Moors (ed.) *Conservation of Island Birds*, pp. 35–81. ICBP Technical Publication 3. Cambridge, UK: International Council for Bird Preservation (ICBP).

Bell, E. (2004). *Lundy Seabird Recovery Project 2002–2004: The Eradication of Black and Brown Rats from Lundy Island*. New Zealand: Wildlife Management International Ltd.

Bell, E., Boyle, D., Floyd, K., Garner-Richards, P., Swann, B., Luxmoore, R., Patterson, A. and Thomas, R. (2011). 'The Ground-based Eradication of Norway Rats (*Rattus norvegicus*) from the Isle of Canna, Inner Hebrides, Scotland'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 269–274. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Bell, E., Bell, B. and Merton, D. (2016). 'The legacy of Big South Cape: rat irruption to rat eradication'. *New Zealand Journal of Ecology* 40: 212–218.

Bell, E., Floyd, K., Boyle, D., Pearson, J., St Pierre, P., Lock, L., Mason, S., McCarthy, R. and Garratt, W. (2019a). 'The Isles of Scilly seabird restoration project: the eradication of brown rats (*Rattus norvegicus*) on the inhabited islands of St Agnes and Gugh, Isles of Scilly'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 88–94. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Bell, E., Bell, M., Morgan, G. and Morgan, L. (2019b). 'The recovery of seabird populations on Ramsey Island, Pembrokeshire, Wales, following the 1999/2000 rat eradication'. In: C.R. Veitch, M.N. Clout, A.R Martin, J.C Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 539–544. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Bentley, E. (1959). 'The distribution and status of *Rattus rattus* L. in the United Kingdom in 1951 and 1956'. *Journal of Animal Ecology* 28: 299–308.

Booker, H. and Price, D. (2014). 'Manx shearwater recovery on Lundy: population and distribution change from 2001 to 2013'. *Journal of the Lundy Field Society* 4: 105–116.

Brooke, M. (1990). *The Manx Shearwater*. Calton, UK: T. & A.D. Poyser.

Brooke, M., Hilton, G. and Martins, T. (2007). 'Prioritising the world's islands for vertebrate-eradication programmes'. *Animal Conservation* 10: 380–390.

Broome, K., Brown, D, Cox, A., Cromarty, P., McClelland, P., Golding, C., Griffiths, R. and Bell, P. (2011). *Current Agreed Best Practice for Rat Eradication - poison bait in bait stations (Version 1.3).* New Zealand Department of Conservation internal document DOCDM-839096. Wellington, New Zealand: Department of Conservation.

Brown, A., Price, D., Slader, P., Booker H., Lock, L. and Deveney, D. (2011). 'Seabirds on Lundy: their current status, recent history and prospects for the restoration of a once-important bird area'. *British Birds* 104: 139–158.

Bryce, R., Oliver, M., Davies, L., Gray, H., Urquhart, J. and Lambin, X. (2011). 'Turning back the tide of American mink invasion at an unprecedented scale through community participation and adaptive management'. *Biological Conservation* 144: 575–583.

Campbell, J. (1892). 'On the appearance of the brown rat (*Mus decumanus*, Pallas) on Ailsa Craig'. *Annals of Scottish Natural History* 1892: 132–134.

Campbell, K., Beek, J., Eason, C., Glen, A., Godwin, G., Gould, F., Holmes, N., Howald, G., Madden, F., Ponder, J., Threadgill, D., Wegmann, A. and Baxter, G. (2015). 'The next generation of rodent eradications: Innovative technologies and tools to improve species specificity and increase their feasibility on islands'. *Biological Conservation* 185: 47–58.

Corbet, G. and Southern, H. (1977). '*The Handbook of British Mammals*'. Oxford: Blackwell Scientific Publications.

Courchamp, F., Chapius, J-L. and Pascal, M. (2003). 'Mammal invaders on islands: impacts, control and control impact'. *Biological Review* 78: 347–383.

Cromarty, P., Broome, K., Cox, A., Empson, R., Hutchinson, W. and McFadden, I. (2002). 'Eradication planning for invasive alien animal species on islands – the approach developed by New Zealand Department of Conservation'. In: C.R. Veitch and M.N. Clout (eds.). *Turning the tide: the eradication of invasive species*, pp. 85–91. IUCN, Gland, Switzerland and Cambridge, UK.

CRRU. (2015). *CRRU UK Code of Best Practice: Best Practice and Guidance for Rodent Control and the Safe Use of Rodenticides.* Yorkshire, UK: CRRU UK.

Dawson, J. Oppel, S., Cuthbert, R., Homes, N., Bird, J., Butchart, S., Spatz, D. and Tershey, B. (2015). 'Prioritizing islands for the eradication of invasive vertebrates in the United Kingdom overseas territories'. *Conservation Biology* 29: 143–153.

DIISE. (2015). 'The Database of Island Invasive Species Eradications, developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand'. <http://diise.islandconservation.org>.

Eason, C., Ogilvie, S., Miller, A., Henderson, R., Shapiro, L., Hix, S., MacMorran, D. and Murphy, E. (2008). 'Smarter Pest Control Tools with Low-residue and Humane Toxins'. In: R.M. Timm and J.M. O'Brien (eds.) *Proceedings of the 23rd Vertebrate Pest Conference*, pp. 148–153. Davis, California, USA: University of California.

Hayhow, D., Bond, A., Douse, A., Eaton, M., Frost, T., Grice, P., Hall, C., Harris, S., Havery, S., Hearn, R., Noble, D., Oppel, S., Williams, J., Win, I. and Wotton, S. (2017). *The State of the UK's Birds 2016*. Sandy, Bedfordshire: The RSPB, BTO, WWT, DAERA, JNCC, NE, NRW and SNH.

Heaney, V., Ratcliffe, N., Brown, A., Robinson, P. and Lock, L. (2002). 'The status and distribution of storm-petrels *Hydrobates pelagicus* and Manx shearwaters *Puffinus puffinus* on the Isles of Scilly'. *Atlantic Seabirds* 4: 1–16.

Holmes, N., Campbell, K., Keitt, B., Griffiths, R., Beek, J., Donlan, C. and Broome, K. (2015). 'Reporting costs for invasive vertebrate eradications'. *Biological Invasions* 17: 2913–2925.

Howald, G., Donlan, C., Galvan, J., Russell, J., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

IUCN. (2017). 'The IUCN Red List of Threatened Species. Version 2017-1'. <http://www.iucnredlist.org>. Accessed 12 May 2017.

Johnstone, I., Gray, C. and Noble, D. (2005). *The State of Birds in Wales 2004*. Cardiff, UK: RSPB Cymru.

Jones, H., Tershy, B., Zavaleta, E., Croll, D., Keitt, B., Finkelstein, M. and Howald, G. (2008). 'Review of the global severity of the effects of invasive rats on seabirds'. *Conservation Biology* 22: 16–26.

Lock, L. (2006). 'Eradication of brown rats *Rattus norvegicus* and black rats *Rattus rattus* to restore breeding seabird populations on Lundy Island, Devon, England'. *Conservation Evidence* 3: 111–113.

Long, J. (2003). *Introduced Mammals of the World: Their History, Distribution and Influence*, Wallingford, UK: CABI.

Lorvelec, O. and Pascal, M. (2005). 'French attempts to eradicate non-indigenous mammals and their consequences for native biota'. *Biological Invasions* 7: 135–140.

Main, C., Bell, E., Floyd, K., Tayton, J., Ibbotson, J., Whittington, W., Taylor, P., Reid, R., Varnham, K., Churchyard, T., Bambini, L., Douse, A., Nicolson, T. and Campbell, G. (2019). 'Scaling down (cliffs) to meet the challenge: the Shiants black rat eradication'. In: C.R. Veitch, M.N. Clout, A.R Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 138–146. Gland, Switzerland: IUCN.

Martins, T., Brooke, M., Hilton, G., Farnsworth, S., Gould, J. and Pain, D. (2006). 'Costing eradications of alien mammals from islands'. *Animal Conservation* 9: 439–444.

Matheson, C. (1962). *Brown Rat*. London: Sunday Times Publications. (*Animals of Britain, No. 16*).

Mavor, R., Heubeck, M., Schmitt, S. and Parsons, M. (2008). *Seabird Numbers and Breeding Success in Britain and Ireland, 2006*. Peterborough: Joint Nature Conservation Committee. (*UK Nature Conservation, No. 31*).

McCann, J. (2005). 'The introduction of the brown rat (*Rattus norvegicus*)'. *Somerset Archaeology and Natural History Journal* 2005: 139–141.

Mitchell, P., Newton, S., Ratcliffe, N. and Dunn, T. (2004). *Seabird Populations of Britain and Ireland: Results of the Seabird 2000 Census (1998–2002).* London: T. & A.D. Poyser.

Moon, K., Blackman, D. and Brewer, T. (2015). 'Understanding and integrating knowledge to improve invasive species management'. *Biological Invasions* 17(9): 2675–2689.

Moors, P.J. and Atkinson, I.A.E (1984). 'Predation on Seabirds by Introduced Animals and Factors Affecting its Severity'. In: J.P. Croxall, P.G.H. Evans and R.W. Schreiber (eds.) *Status and Conservation of the World's Seabirds,* pp. 667–690. ICBP Technical Publication 2. Cambridge, UK: International Council for Bird Preservation (ICBP).

Moors, P.J., Atkinson, I.A.E. and Sherley, G.H. (1992). 'Reducing the rat threat to island birds'. *Bird Conservation International* 2: 93–114.

Morgan, G. (2012). 'The bird populations of Ramsey and Grassholm'. *British Birds* 105: 716–732.

Oppel, S., Beaven, B., Bolton, M., Vickery, J. and Bodey, T. (2011). 'Eradication of invasive mammals on islands inhabited by humans and domestic animals'. *Conservation Biology* 25: 232–240.

Parks and Wildlife Service. (2008). *Macquarie Island Pest Eradication Plan. Part A: Overview*. Hobart, Tasmania: Department of the Environment, Parks, Heritage and the Arts. <http://www.parks.tas.gov.au/index.aspx?base=6186>.

Parks and Wildlife Service. (2014). *Evaluation Report: Macquarie Island Pest Eradication Project, August 2014*. Hobart, Tasmania: Department of Primary Industries, Parks, Water and Environment. <http://www.parks.tas.gov.au/file.aspx?id=31160>.

Pearson, J., St Pierre, P., Lock, L., Buckley, P., Bell, E., Mason, S., McCarthy, R. Garratt, W., Sugar, K. and Pearce, J. (2019). 'Working with the local community to eradicate rats on an inhabited island: securing the seabird heritage of the Isles of Scilly'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 670–678. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Ratcliffe, J. and Sandison W. (2001). 'Puffin Island - will removing rats bring back the puffins?'. *Natur Cymru* 1: 23–26.

Ratcliffe, N., Mitchell, I., Varnham, K., Verboven, N. and Higson, P. (2009). 'How to prioritise islands for the benefit of petrels: a case study of the UK, Channel Islands and Isle of Man'. *Ibis* 151: 699–708.

RSPB. (1924). 'Ailsa Craig'. *Bird Notes and News*.11: 27–28.

RSPB. (1925a). 'Ailsa Craig'. *Bird Notes and News* 11: 99–100.

RSPB. (1925b). 'Ailsa Craig'. *Bird Notes and News* 11: 150.

Russell, J. and Broome, K. (2016). 'Fifty years of rodent eradications in New Zealand: another decade of advances'. *New Zealand Journal of Ecology* 40: 197–204.

Stanbury, A., Thomas, S., Aegerter, J., Brown, A., Bullock, A., Eaton, M., Lock, L., Luxmoore, R., Roy, S., Whitaker, S. and Oppel, S. (2017). 'Prioritising islands in the United Kingdom and crown dependencies for the eradication of invasive alien vertebrates and rodent biosecurity'. *European Journal of Wildlife Research* 63: 31. <https://doi.org/10.1007/s10344-017-1084-7>.

Stoneman, J. and Zonfrillo, B. (2005). 'The eradication of brown rats from Handa Island, Sutherland'. *Scottish Birds* 25: 17–23.

Swann, R., Alton, D., Foster, S., Graham, K., Graham, A., Ramsay, A. and Young, A. (2007). 'The impact of brown rats on Canna's breeding seabirds'. *Proceedings of the Conference: Tackling the Problem of Alien Mammals on Seabird Colonies*. Edinburgh, Scotland: National Trust for Scotland.

Taylor, R., Kaiser, G. and Drever, M. (2000). 'Eradication of Norway rats for recovery of seabird habitat on Langara Island, British Columbia'. *Restoration Ecology* 8: 151–160.

Thomas, C. (1985). *Exploration of a Drowned Landscape: Archaeology and History of the Isles of Scilly*. London: Batsford Ltd.

Thomas, B. and Taylor, R. (2002). 'A history of ground-based rodent eradication techniques developed in New Zealand, 1959–1993'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 301–310. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Thomas, S., Brown, A., Bullock, D., Lock, L., Luxmoore, R., Roy, S., Stanbury, A. and Varnham, K. (2017a). 'Island restoration in the UK – past present and future'. *British Wildlife* 28: 231–242.

Thomas, S., Varnham, K., and Havery, S. (2017b). *Current Recommended Procedures for UK (Bait Station) Rodent Eradication Projects (Version 4.0)*. Sandy, Bedfordshire: Royal Society for the Protection of Birds.

Towns, D. and Broome, K. (2003). 'From small Maria to massive Campbell: Forty years of rat eradications from New Zealand islands'. *New Zealand Journal of Zoology* 30: 377–398.

Towns, D., Atkinson, I. and Daugherty, C. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?'. *Biological Invasions* 8: 863–891.

Twigg, G. (1992). 'The black rat *Rattus* in the United Kingdom in 1989'. *Mammal Review* 22: 33–42.

Walsh, A. (2019). 'Winning the hearts and minds – proceeding to implementation of the Lord Howe Island rodent eradication project: a case study'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 522–530. Occasional Paper SSC no. 62. IUCN, Gland, Switzerland.

Wilkinson, I. and Priddel, D. (2011). 'Rodent Eradication on Lord Howe Island: challenges posed by people, livestock, and threatened endemics'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 508–514. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Zonfrillo, B. (2001). *Wildlife Conservation on Ailsa Craig.* The Thomas Duncan Memorial Lectures. Girvan, Scotland: Friends of the McKechnie Institute.

Zonfrillo, B. (2002). 'Puffins return to Ailsa Craig'. *Scottish Bird News* 66: 1–2.

The Isles of Scilly seabird restoration project: the eradication of brown rats (*Rattus norvegicus*) from the inhabited islands of St Agnes and Gugh, Isles of Scilly

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Abstract As part of the Isles of Scilly Seabird Recovery Project, and directed by Wildlife Management International Ltd, the eradication of brown rats (*Rattus norvegicus*) from the inhabited islands of St Agnes & Gugh, Isles of Scilly was completed between October 2013 and April 2014 with the assistance of volunteers, and staff from the Royal Society for the Protection of Birds, Isles of Scilly Wildlife Trust and Natural England. Bait stations with cereal-based wax blocks containing bromadiolone at 0.005% w/w were established on a 40–50 metre grid over the island. With the presence of 85 residents on the 142 ha islands, this is the largest community-based brown rat eradication globally to date. Given the fact that a community is based on these islands, community engagement and advocacy was a vital and fundamental part of the eradication. Consultation for eradication began three years prior to the operation to explain the requirements for the proposed project and to assess support, but this built on many years of wider community engagement with seabird conservation. All of the residents supported the eradication of rats and vision of the project. The consultation and inclusion of the community in decision-making and management of the Isles of Scilly Seabird Recovery Project was a critical part of the operation and key to the success of the eradication. The community took ownership of the project and has committed to the on-going biosecurity requirements following the eradication of rats. The removal of brown rats from St Agnes and Gugh was a major achievement and provided the opportunity to restore the islands' communities of seabirds and other native species. This project provided an example of the effectiveness of ground-based rodent eradication techniques on an inhabited island and the lessons learnt during this operation can be used to help proposed eradications on other islands with communities and with terrain suitable for ground-based techniques.

Keywords: brown rat, community, eradication, Isles of Scilly, *Rattus norvegicus*, St Agnes and Gugh

INTRODUCTION

The eradication of invasive species from islands has become one of the most important tools in conservation in recent times. It offers the opportunity that, following an initial investment, significant long-term benefits can be achieved. The eradication of rats is a recognised prerequisite for the restoration of many seabird colonies on islands. Rodents have been successfully eradicated from over 700 islands around the world, including at least 10 UK islands (Moors & Atkinson, 1984; Atkinson, 1985; Taylor, et al., 2000; Zonfrillo, 2001; Towns & Broome, 2003; Appleton, et al., 2006; Howald, et al., 2007; Jones, et al., 2008; Bell, et al., 2011; Parks & Wildlife Service, 2014; DIISE, 2015; Thomas, et al., 2017; Bell, 2019; Bell, et al., 2019; Pearson, et al., 2019), However, most of these islands have been uninhabited. Many consider that islands with significant human populations, unreceptive communities or occurrence of livestock and domestic animals are unlikely to be feasible for eradication (Oppel, et al., 2011; Campbell, et al., 2015; Russell & Broome, 2016; Stanbury, et al., 2017). However, an increasing number of eradications are being considered on inhabited islands and the importance of the engagement and inclusion of local communities has been highlighted in a number of recent eradication and research projects, especially in regard to risk and benefit analysis (Eason, et al., 2008; Bryce, et al., 2011; Oppel, et al., 2011). It should be noted that the greatest conservation benefit to be gained from future eradications in the UK, and in other parts of the world, is predominantly from inhabited islands (Stanbury, et al., 2017). As such, it is vital that techniques and protocols developed during eradications on islands with even small communities should be assessed, utilised or adapted for these islands with larger communities.

The Isles of Scilly are a nationally and internationally important location for seabirds, particularly Manx shearwater (*Puffinus puffinus*), European storm petrel (*Hydrobates pelagicus*) and black-backed gull (*Larus fuscus*) (Lock, et al., 2006). Both Manx shearwaters and European storm petrels are amber listed under the United Kingdom Birds of Conservation Concern threat categorisation (Eaton, et al., 2015). A partnership of organisations (Royal Society for the Protection of Birds (RSPB), Natural England (NE), Isles of Scilly Wildlife Trust (IOSWT) and Isles of Scilly Bird Group (IOSBG)) produced the Isles of Scilly Seabird Conservation Strategies 2005–2008 and 2009–2013 which described the national and international status and context of the seabird populations on the Isles of Scilly and identified priority actions and strategic goals for management. These included current and future measures to improve the available habitat for seabirds through rat control and eradication (Lock, et al., 2006; Lock, et al., 2009). St Agnes and Gugh have a number of important land areas designated for seabirds as Special Protected Areas (SPA), Sites of Special Scientific Interest (SSSI) and Ramsar (Lock, et al., 2009). The eradication of brown rats (*Rattus norvegicus*) from St Agnes and Gugh was identified as a priority in these strategies as it would remove predation pressure on Manx shearwaters and storm petrels and provide the opportunity for other seabirds to colonise the islands (Lock, et al., 2006; Lock, et al., 2009). These strategies also recognised the social, economic and health benefits for the local community (Lock, et al., 2006; Lock, et al., 2009).

The Isles of Scilly Seabird Recovery Project (IOSSRP) was established in 2010 and was managed by a coalition of

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groups including RSPB, IOSWT, NE, Duchy of Cornwall (DC), the Isles of Scilly Area of Outstanding Natural Beauty (AONB) partnership and a representative from St Agnes and Gugh, with support from the IOSBG. The IOSSRP partnership identified the need to assess the possibility of eradicating brown rats from St Agnes and Gugh to protect and enhance the islands' seabirds and protect Annet from re-invasion. Annet is the most important uninhabited island for seabirds in the Isles of Scilly as it has always been rat-free (excluding an incursion in 2004, probably from neighbouring St Agnes) and holds the main populations of Manx shearwaters and European storm petrels (Lock, et al., 2006). The partnership commissioned a feasibility assessment in 2010 (Bell, 2011). A formal IOSSRP Steering Group made up of representatives from all Project Partners was established in 2012. Wildlife Management International Ltd. (WMIL) directed the eradication with the assistance of volunteers and RSPB, IOSWT and NE staff. The eradication was completed between October 2013 and April 2014 (Bell, et al., 2014). This paper covers the technical aspects of the St Agnes and Gugh brown rat eradication and complements the Pearson, et al., (this issue) paper on the community aspect of the eradication.

STUDY AREA AND METHODS

St Agnes and Gugh

St Agnes and Gugh (49.89267°N, 6.34073°W) are two islands in the Isles of Scilly archipelago off the Cornish coast, in south-west England (Fig. 1). St Agnes (105 ha) and Gugh (37 ha) are connected by a rock and sand bar at low tide (Fig. 1). St Agnes and Gugh are separated from St Mary's by a deep channel (St Mary's Sound) that is 1.1 kilometres at the closest point (via stepping stone islands) or 1.3 km from shore to shore (Fig. 1). There are 85 residents, only two of whom live on Gugh. Brown rats were accidentally introduced to the Isles of Scilly from shipwrecks in the 1700s, and were widespread and abundant across both islands, as well as many other islands in the archipelago (Matheson, 1962; McCann, 2005). Tourism is one of the islands' major sources of income, particularly between April and October.

There are approximately 40 homes on the island, but at least 150 buildings (holiday lets, farm buildings, sheds, etc.) scattered across the whole island. There are six farms (including a chicken farm and dairy), a campground, a school, a restaurant, a pub, two cafes, a post office and store. There are cattle, chickens, ducks, geese, two ponies and pigs on St Agnes. Many families have pet cats and dogs. There is a main quay where passengers and freight are landed, and a smaller slipway used mainly by residents. These factors increased the number of challenges such as providing alternative food and shelter for rats, risk to non-target species and biosecurity.

Fig. 1 Location of St Agnes and Gugh, Isles of Scilly, United Kingdom.

The main habitats on St Agnes are farmland, mainly flower farms and low intensity cattle grazing, characterised by small fields with extensive hedges and stone walls, ponds, maritime grassland, invasive *Pittosporum*, rocky shores and sandy beaches (Parslow, 2007). St Agnes and Gugh are home to the only known populations in the British Isles of a number of rare plants, including least adder's-tongue fern (*Ophioglossum lusitanicum*) (Parslow, 2007).

Rabbits (*Oryctolagus cuniculus*), Scilly shrews (*Crocidura suaveolens cassiteridum*) and pipistrelle bats (*Pipistrellus* spp.) are the only other known species of mammal found on St Agnes and Gugh, apart from livestock and pets. House mice (*Mus musculus domesticus*) were present on St Agnes and Gugh, but have not been seen in at least 15 years, though mice are still present on most of the other main islands in the Scillies (Howie, et al., 2007).

Eradication operation

The eradication programme ran from 11 October 2013 to 11 April 2014 and included establishing the bait station grid, poisoning, monitoring and biosecurity establishment. This phase took 1,593 person days. Long-term monitoring ran monthly between May 2014 and December 2015. The final check, species monitoring, and rat-free declaration ran from 6 January to 18 February 2016. This phase took 250 person-days. All IOSSRP personnel wore blaze-orange hats (with the IOSSRP logo) to be easily recognisable to the community and visitors. Each operational task was undertaken and completed as follows:

Pre-eradication

Due to the presence of a community on the island and the selected method of bait stations, different pre-eradication preparation tasks were required compared to aerial baiting methods. Preparation tasks included, but were not limited to: consultations with the community about operational techniques; timing of each aspect of the project and confirming access to land and buildings; testing rats for resistance to rodenticides; getting the community to cease using rodenticides on the island six months prior to the eradication (i.e. to prevent bait aversion, avoid rats becoming accustomed to bait and to prevent resistance); removal of waste, alternative food and harbourage (including cleaning up farm sheds and other buildings on the island); establishing waste management systems for each household and business (including provision of rodent proof wheelie-bins and compost bins); application for an extension-of-use for rodenticide use from the UK Health and Safety Executive (HSE); construction of bait stations; and delivery of all equipment to the islands.

The University of Reading completed resistance testing and DNA screening of 26 rats trapped on the islands. Of these samples, resistance (L120Q mutation) was detected in one individual (Rymer, 2013). This resistance evidence confirmed the requirement for multiple toxin and bait formulations to ensure any problem rats could be targeted successfully. An extension-of-use permission from HSE was obtained to use specific rodenticides (difenacoum and brodifacoum) at specific locations outdoors if it became necessary to target any resistant rats towards the end of the eradication.

Over 1,500 bait stations were constructed by RSPB staff and volunteers in Penzance and these and all other equipment was delivered to St Agnes in September 2013.

Bait station grid

The bait station grid was established between 12 October and 7 November 2013. Bait stations were made from 750 mm lengths of 100 mm diameter corrugated black plastic drainage pipes, wired into the ground to prevent movement by animals and/or wind. Bait was placed in the centre of the station through the access hole that is covered by an additional short section of pipe and held in place by a ‘crow clip’ (a short piece of wire wrapped around the centre of the station devised during the Lundy Island rat eradication operation which prevents the crows and gulls removing the lids (Bell, 2019)).

Bait stations were placed out on a 40 m × 50 m grid. Positions were determined by electronic Geographic Information System (GIS) and loaded onto a hand-held GPS unit. Each station was marked by a bamboo cane or flagging tape to ensure visibility in thick vegetation or poor weather.

The entire grid of 962 tube stations was positioned across the island (with an additional 74 commercial Protecta™ lockable bait stations inside all private homes, holiday rentals, public buildings and on the quay) before being individually numbered and mapped using GPS and added to a GIS-linked database (Fig. 2).

Poisoning

The main toxicant used was bromadiolone, Contrac™ (manufactured by Bell Laboratories), a 28 g, cereal-based wax block bait with 0.005% active ingredient. This bait was used between 8 November 2013–12 January 2013 and 27 January–8 March 2014 (Table 1). There were two alternative baits, both manufactured by PelGar International, available if any rats were detected that seemed to be avoiding or appeared to be resistant to the main bait: Roban Excel™, a 20 g cereal-based block bait with active ingredient difenacoum at 0.005% w/w that was used between 13–26 January 2014 (Table 1); and Vertox Oktablok II™, a 20 g cereal-based block with active ingredient brodifacoum at 0.005% w/w that was not required. Contrac™ and Roban Excel™ are dyed blue (or green/blue) to be less attractive to birds (Caithness & Williams, 1971; Hartley, et al., 1999; Weser & Ross, 2013), thus helping to further reduce risks to non-target species.

The poisoning operation commenced on 8 November 2013 and continued through to 8 March 2014. Baits were present in each station throughout the poisoning programme and replaced as required; when eaten by rats, by non-target species such as invertebrates and/or damaged by weather. Between 8 and 18 November 2013 there were eight blocks of bait in each station. This was reduced to four blocks between 19 and 25 November 2013 and reduced again to two blocks from 26 November 2013 to 26 January 2014 (Table 1). After 27 January 2014, only one block of bait was placed in each station. Existing undamaged bait blocks were left in the stations and the extra blocks were removed. All waste and partially eaten bait was collected

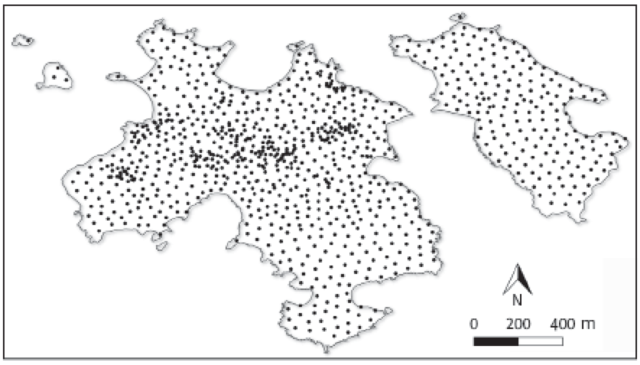

Fig. 2 Bait station grid on St Agnes and Gugh, Isles of Scilly. Bait station positions are marked by a black dot.

and incinerated in a high temperature incineration facility at the end of the operation.

Bait was loose in the stations between 8 and 25 November 2013 (so that rats can take bait back to their burrows to feed nursing females or young) and after 26 November all bait was wired into the stations (which could be used to confirm the presence of rats due to teeth marks being recorded on partially eaten blocks in the stations) (Table 1).

Excluding the stations in the houses (which were checked once a week), all other bait stations on St Agnes and Gugh were checked and serviced at intervals between one to seven days (a total of 56 bait checks over 120 days) depending on the stage of the operation (Table 2). To present the data on bait-take gained from these varied bait station checks we grouped the data into 27 periods or checks (mean (±SEM) = 1.9 ± 0.2 days between checks, range 1–7 days) shown as days from baiting (Fig. 3).

Bait-take was recorded in field notebooks by bait station number and the species believed to have consumed

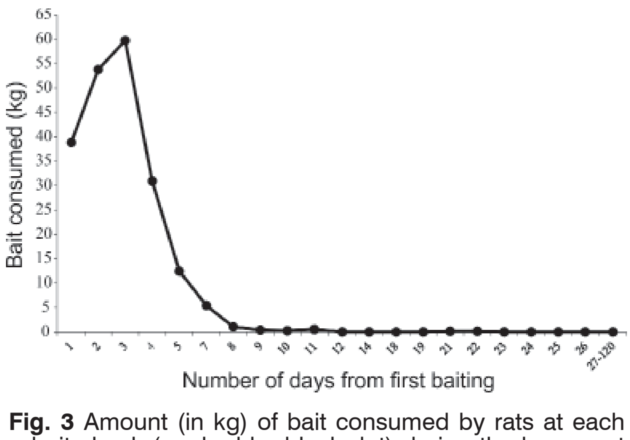

Fig. 3 Amount (in kg) of bait consumed by rats at each bait check (marked by black dot) during the brown rat (*Rattus norvegicus*) eradication on St Agnes and Gugh, Isles of Scilly.

Table 1 Baiting regime during the brown rat (*Rattus norvegicus*) eradication on St Agnes and Gugh, Isles of Scilly, United Kingdom.

Date	Bait type	No of blocks	Bait loose or wired into station
8–18 Nov 2013	Contrac™ (bromadiolone)	8	Loose in station
19–25 Nov 2013	Contrac™ (bromadiolone)	4	Loose in station
26 Nov 2013 to 12 Jan 2014	Contrac™ (bromadiolone)	2	Wired into station
13–26 Jan 2014	Roban Excel™ (difenacoum)	2	Wired into station
27 Jan to 8 Mar 2014	Contrac™ (bromadiolone)	1	Wired into station

Table 2 Number of bait station checks during the brown rat (*Rattus norvegicus*) eradication on St Agnes and Gugh, Isles of Scilly, United Kingdom.

Date	Checks per week
8–20 November 2013	6
21 Nov to 13 Dec 2013	5
14 Dec 2013 to 8 Mar 2014	3

or removed the bait. These data were entered into a GIS-linked database and maps showing active stations were produced in real-time to enable the team to effectively monitor bait-take activity and target any "hot spots".

Searches for carcasses were completed during all checks. Any carcasses that were found, were collected, necropsied to determine cause of death (where possible) and incinerated to reduce risk for non-target scavengers. It was expected that very few rat carcasses would be found on the surface as most rats die underground in their burrows. Five rat carcasses were found on the surface during the Lundy Island rat eradication and three during the Isle of Canna rat eradication (Bell, 2004; Bell, et al., 2006). Any non-target species that were collected during the operation were also necropsied and assessed for anticoagulant poisoning (i.e. blood in body cavity, bruising, discolouration of organs). Non-target species have been affected during other eradications: 77 non-target species' carcasses (greater black-backed gull *Larus marinus*, carrion crow *Corvus corone*, house sparrow *Passer domesticus*, short-eared owl *Afio flammeus* and rabbit *Oryctolagus cuniculus)* were found on the surface during the Lundy Island rat eradication and seven non-target species carcasses (wood mouse *Apodemus sylvaticus*, and pygmy shrew *Sorex minutus*) were found during the Isle of Canna operation. Of these, only 15 showed evidence of anticoagulant poisoning and the remainder had died of starvation (rabbit, shrew) or either natural (short-eared owl, crow) or unknown causes (greater black-backed gulls) (Bell, 2004; Bell, et al., 2011).

Monitoring

Three distinct periods of monitoring were undertaken as the project progressed. Intensive monitoring using 2,500 stations at 25 m spacing was carried out from 19 November 2013 to 8 March 2014 to detect rats surviving through the poisoning phase. This was followed by a 21-month period of long-term monitoring using 87 biosecurity stations and six rodent motels (wooden boxes designed to provide an attractive, alternative 'burrow' for rats during an incursion) from 9 March 2014 to 5 January 2016. These biosecurity stations were established at high risk areas on the island; around the coast, at the quay and other boat landing sites and at seabird breeding sites (Bell, et al., 2014). The final monitoring check, using 448 stations, was carried out between 6 January and 18 February 2016 (Bell & Cropper, 2016). WMIL and RSPB staff and volunteers carried out the intensive and final checks and IOSSRP staff, St Agnes and Gugh residents and volunteers maintained the long-term monitoring. Monitoring stations consisted of materials attractive to rats that would also clearly show teeth marks (e.g. chocolate, peanut or coconut flavoured wax, candles and soap), tracking tunnels and trail cameras (Bushnell™). All were individually numbered and any evidence of activity (e.g. teeth marks or foot prints) was recorded in field notebooks by station number and the species believed to have consumed or marked the monitoring item.

Monitoring items were placed inside and outside each station as well as halfway between each station during the intensive monitoring phase and final monitoring check. During these monitoring phases, each monitoring site was checked regularly 3–5 times a week (depending on weather), either separately or – during the poisoning phase – together with the poisoning bait station grid. Monitoring items were placed inside the biosecurity stations only during the long-term monitoring phase and these were checked monthly. Checks for active rat runs and activity at high-risk sites (i.e. stone walls, farms, seabird colonies, etc.) were also undertaken throughout all three monitoring phases. Any rat and non-target species sign found on any monitoring detection device at any stage of the monitoring phase was recorded and added to the database.

RESULTS

Bait acceptance and take

Bait acceptance was excellent with no evidence of bait avoidance. Green/blue rat droppings appeared within three days and rats accounted for 203.6 kg of Contrac™ bait taken (estimated 1,600–2,500 rats).

The bait-take pattern was typical of other rat eradication campaigns (Thomas & Taylor 2002; Bell, et al., 2011). It was very high in the immediate days after original baiting (checks 1–3) and dropped to a relatively low level eight days after original baiting (check 8) (Fig. 3). A small increase was recorded at day 21 after the original baiting (check 15) but dropped away, reaching zero bait-take on day 23 after the original baiting (check 17) (Fig. 3).

Throughout the poisoning phase, 62% of bait stations were visited by rats, with 42.7% active within the first three days of the original baiting. This level of activity was similar to the Lundy and Isle of Canna eradications which had 42.5% and 62% of bait stations visited by rats, respectively (Bell, 2004; Bell, et al., 2011). The high number of active bait stations during the first two bait checks shows that the rats quickly accepted the bait across St Agnes and Gugh. It is likely that the small grid size and intensive baiting regime targeted the rats effectively within a short timeframe.

The average number of blocks taken by rats was 4.3 (± 0.1) blocks per active station (range 0–41 blocks). Again, this level of activity was similar to the Lundy and Isle of Canna eradications which had 3.2 and 8 blocks taken by rats by per active station, respectively (Bell, et al., 2004; Bell, et al., 2011). This also indicates that rats were quickly removed from most sites across St Agnes and Gugh. As shown by Fig. 4, bait-take was not evenly distributed over both islands, with the greatest level of bait-take on the coastal areas of both islands and each of the offshore

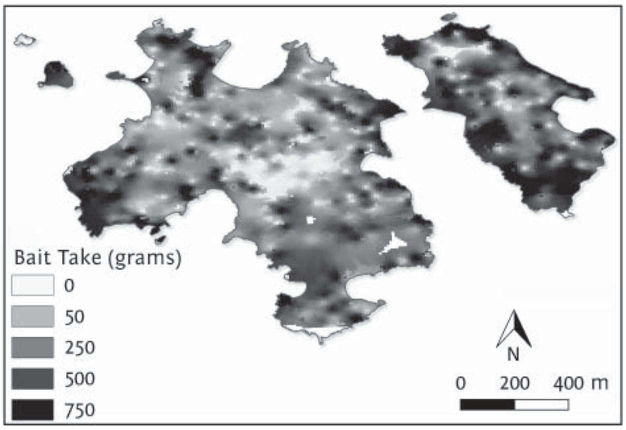

Fig. 4 Distribution of total bait-take (g) by rats consumed per station during the brown rat (*Rattus norvegicus*) eradication on St Agnes and Gugh, Isles of Scilly.

rock stacks connected to the main islands at low tide. The distribution of rats and density on Gugh was likely to be having an impact on Manx shearwaters and other seabirds and land bird and invertebrate populations present on St Agnes and Gugh.

There were 19 rat carcasses collected on the surface during the operation. These were collected and incinerated to prevent availability to non-target species.

There were low levels of interference by non-target species with nearly 54 kg of bait being consumed; cattle kicked up stations and ate a small amount of bait (1.4 kg), slugs and other insects consumed 51.9 kg and shrews consumed 0.4 kg. The weather conditions also complicated the operation and accounted for 3.4 kg of bait that had to be replaced due to the loss of 54 bait stations in storms. Carcasses of a water rail (*Rallus aquaticus*), a song thrush (*Turdus philomelos*), a blackbird (*T. merula*) and nine Scilly shrews were found. There was no evidence that any of these non-target species was affected by the rodenticide.

Monitoring

Monitoring for rat presence continued island-wide for two years after the end of the poisoning operation. The last rat was detected on chocolate flavoured wax on 29 November 2013 during the overlap between the poisoning and intensive monitoring phases and this rat was successfully targeted using the main bait, Contrac™, by 2 December 2013. No rats or sign were detected during any phase of the long-term or final check monitoring. St Agnes and Gugh were declared rat-free in February 2016.

Cattle, shrews and birds interfered with 899 monitoring stations (by eating the flavoured wax or soap, marking tracking plates or, in the case of cattle, by removing the monitoring wires) a total of 12,156 times between 21 November 2013 and 26 February 2014. There were 127 stations affected 1,384 times by cattle, 60 (82 times) by birds, 5 (8 times) by insects, 9 (9 times) by rabbits and 454 (2084 times) by shrews. Interference by birds, shrews and rabbits was limited to teeth or beak marks on the soap or flavoured wax or footprints on tracking plates. Cattle removed wires and ate flavoured wax and soap, so monitoring points had to be moved or hidden in those areas with cattle.

DISCUSSION

The success of the St Agnes and Gugh brown rat eradication shows that a well-planned, adequately resourced, well-executed programme, with the complete support of the community, local agencies and government and directed by an experienced operator with dedicated workers, can eradicate rats from inhabited islands using a ground-based bait station operation. The project on St Agnes and Gugh is the largest community-led (with 85 residents) brown rat eradication anywhere in the world. Most other eradications on inhabited islands either have smaller communities (e.g. Isle of Canna, 12 residents; Bell, et al., 2011; Rakino in New Zealand, 16 residents; Bassett, et al., 2016) or have staff or a military population (e.g. Bird, Denis, Curieuse and Fregate Islands in the Seychelles, Merton, et al., 2002; Lundy Island, Bell, 2004; Wake Island, Brown, et al., 2013) and have not had direct involvement of the community during and after the eradication or leaving the community responsible for all biosecurity measures (Pearson, et al., this issue).

However, the success of the eradication was dependent on the participation and support of the entire local community. The community maintained an integral role and was consulted extensively in the planning, preparation and implementation of the eradication programme. As such, it is vital that techniques and protocols developed during eradications on islands with even small communities should be assessed, utilised or adapted for islands with larger communities. The opinions and safety of local communities need to be a priority in any eradication planned for inhabited islands.

Stock and chicken feed provided a possible alternative food source for rats, but all the farmers were fully supportive of the project and stored all the unopened feed on pallets (with bait stations and/or traps underneath) or in rodent-proof containers and any opened feed was stored in large plastic, metal or wooden sealed bins. Where possible, farm buildings were kept clean to ensure fresh sign was quickly noted. All these methods meant that the sheds were cleared of rats and any roaming rats which re-invaded the area could be noted quickly. The presence of a large chicken farm could have been a major problem as their runs provide excellent rat habitat and alternative food. The owner of the chicken farm strictly managed his chickens and feeding regime throughout the rat eradication operation which made targeting rats and monitoring for any survivors on this farm easier.

Rubbish can be the most serious issue on an inhabited island wanting to eradicate rats. This was discussed comprehensively with the community before the project commenced. As a result, rat-proof wheelie bins and Green Johanna compost bins were provided to the residents and all rubbish was stored in these prior to removal to St Mary's. Rubbish was removed regularly (generally weekly) from St Agnes to St Mary's by the Isles of Scilly Council. In October and early November 2013, with the permission and assistance of residents, a number of sheds, farm buildings and outhouses were cleared and tidied by the IOSSRP team to ensure bait stations could be placed along all the walls.

St Agnes and Gugh were cleared of rats within three weeks (23 days from original baiting). Bait-take showed that the rat population appeared to be low (approximately 2,000 rats) and was not evenly distributed across the islands. There were high concentrations of rats on Gugh and around the coastal areas on St Agnes where the burrow-nesting seabird colonies are present, meaning rats were likely to have been having an effect on these breeding seabirds (Moors & Atkinson, 1984; Atkinson, 1985; Jones, et al., 2008).

The interference by cattle was another major factor affecting the operation, with cattle kicking up or crushing stations, but cooperation by the farmers to move stock around different paddocks, as well as altering the bait station positions, wiring the bait or lids into position in addition to the crow clip or weighting the stations down with rocks, meant this problem was quickly dealt with. Many of the monitoring stations were removed from, and then replaced back into, certain areas (such as Covean and Wingletang) as the cattle were rotated between paddocks.

Importantly, there were no known non-target species affected by this operation. Although a small number of Scilly shrews (n = 9) were found dead and necropsied during the eradication, proof of poisoning could not be confirmed (i.e. no symptoms of anticoagulant poisoning such as blood in body cavity, bruising or discolouration of organs). However, no liver or tissue samples were taken from non-target species for further analysis. It should be noted that, in certain cases, bait-take by shrews subsequently stopped in nearby stations suggesting these animals had died due to primary poisoning. Although there is no information on the LD50 for shrews, using LD50 data from other small mammals (voles and mice), it is likely that shrews would have to eat between 0.2–1.25 mg/kg to be affected by bromadiolone. This amounts to 0.001 blocks of bait and this level of bait take by shrews occurred at 83 different stations between 22 November 2013 and 5 March 2014 suggesting that approximately 83 shrews may have been affected by the baiting phase (totalling to 0.4 kg of bait). However, it is thought that as Scilly shrews have small home ranges (< 50 m^2; Spencer-Booth, 1963; Rood, 1965), excluding those with a bait station in their immediate home range, most shrews would not encounter bait stations or poisoned invertebrates using the 40 m × 50 m grid. This means that even if a small number of individuals was killed, the overall population would survive. The risk to the shrew population was considered minimal, but the potential for a small number of individuals to be affected was acknowledged (Bell, 2011). Calculations of bait-take indicate that more shrews than anticipated may have been at risk, but extensive searches for carcasses and the necropsies performed do not support this; there was no definitive evidence of any shrew death being attributable to the rodenticide. Scilly shrew numbers have increased to population levels higher than those before the eradication (IOSSRP, unpublished data).

A large quantity of bait was consumed or damaged by slugs and other insects. Bait was changed often to ensure there was always the most attractive and palatable bait available to rats. Contrac™ was more durable than expected, compared to earlier experience on Lundy Island where it deteriorated within one to two days (Bell, 2004), meaning it lasted better in the St Agnes and Gugh environment. Occasionally it was difficult to interpret sign on the blocks during the important monitoring phase of the operation, owing to the nature of the block and ridges, but the Contrac™ bait successfully targeted all rats on St Agnes and Gugh within three weeks.

There was no evidence that any other non-target species were affected by the rodenticide, traps or monitoring tools used in the operation. Following necropsy of shrews and other non-target species carcasses (water rail, thrush and blackbird), there was no bait found in the stomach or symptoms of anticoagulant poisoning (i.e. blood in the body cavity, bruising or haemorrhaging or discoloured organs). Although 19 dead rats were found on the surface (1.1% of estimated rat population on St Agnes and Gugh), there was no evidence of any other animal scavenging these carcasses. There were no observations of pet cats, crows, gulls or raptors eating dead or dying rats on St Agnes and Gugh.

Weather also affected the eradication when storms removed or dislodged stations, but this generally was limited to coastal areas.

The eradication of invasive species such as rats from islands has become one of the most important tools in avian conservation worldwide. It was recognised that for the restoration and protection of seabird colonies on St Agnes and Gugh, the eradication of rats was required. This operation has already benefited key seabird species on the islands as well as the Scilly shrew as shown by comparisons between the pre- and post-eradication biodiversity monitoring. Manx shearwaters were recorded successfully breeding within one year of the eradication and 73 pairs were recorded in 2016 compared to 22 pairs and no fledged chicks in 2013 (Pearson, 2016). European storm petrels were first recorded on St Agnes in 2015, with 9 pairs in 2016, and the Scilly shrew population has increased to levels higher than the pre-eradication levels since rats have been eradicated (IOSSRP, unpublished data; Pearson, 2016; Thomas, et al., 2017).

Although eradicating rats from St Agnes and Gugh is a considerable and significant achievement, it is important to stress that keeping these islands rat-free will require constant vigilance and commitment from the whole community, partner agencies and visitors in order to prevent, detect and respond to any incursions. Prevention of an accidental rat re-introduction should be the primary aim. The greatest risk is via service and private vessels traveling between all of the inhabited islands in the Isles of Scilly, especially if delivering farming equipment, hay, stock feed, equipment or food to St Agnes. There is also a small risk from visiting yachts and general tourism. Permanent biosecurity stations have been established on St Agnes and Gugh; these will be maintained indefinitely by trained community members and IOSSRP personnel. A detailed biosecurity plan has been developed to prevent, detect and respond to possible incursions. Residents have been trained in these biosecurity measures, identification of rodents and rodent sign, and methods to reduce the risk of accidentally introducing rodents, demonstrating the commitment of the St Agnes and Gugh community to the restoration of their islands.

It is important to stress that the eradication of brown rats from St Agnes and Gugh is a valuable education tool to show other island communities that it is possible to safely eradicate rats without unduly impacting on the lives and habits of the local residents. The successful eradication of brown rats from St Agnes and Gugh demonstrates how the techniques of ground-based bait station operations can be utilised on inhabited islands throughout the UK and the world where this technique is feasible and where the community is involved and supportive.

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REFERENCES

Appleton, D., Booker, H., Bullock, D., Cordrey, L. and Sampson, B. (2006). 'The seabird recovery project: Lundy Island'. *Atlantic Seabirds* 8: 51–59.

Atkinson, I.A.E. (1985). 'The Spread of Commensal Species of *Rattus* to Oceanic Islands and their Effects on Island Avifaunas'. In: P.J. Moors (ed.) *Conservation of Island Birds,* pp. 35–81. ICBP Technical Publication 3. Cambridge, UK: International Council for Bird Preservation (ICBP).

Bassett, I., Cook, J., Buchanan, F. and Russell, J. (2016). 'Treasure Islands: biosecurity in the Hauraki Gulf Marine Park'. *New Zealand Journal of Ecology* 40: 250–266.

Bell, E. (2004). *Lundy Seabird Recovery Project 2002–2004: The Eradication of Black and Brown Rats from Lundy Island*. Unpublished client report to the Royal Society for the Protection of Birds, South-West Regional Office, Devon, UK. Blenheim, New Zealand: Wildlife Management International Ltd.

Bell, E., Boyle, D., Garner–Richards, P. and Floyd, K. (2006). *Canna Seabird Recovery Project: Phase 1: Eradication of Brown Rats 2005–06*. Unpublished client report to the National Trust for Scotland, Edinburgh, Scotland. Blenheim, New Zealand: Wildlife Management International Ltd.

Bell, E (2011). *Isles of Scilly Seabird Recovery Project: Part 3: Assessing the Feasibility of Removing Brown Rats from St Agnes and Gugh, Isles of Scilly*. Unpublished client report to Isles of Scilly Seabird Recovery Project Partnership. Blenheim, New Zealand: Wildlife Management International Ltd.

Bell, E., Boyle, D., Floyd, K., Garner–Richards, P., Swann, B., Luxmoore, R., Patterson, A. and Thomas, R. (2011). 'The ground-based eradication of Norway rats (*Rattus norvegicus*) from the Isle of Canna, Inner Hebrides, Scotland'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 269–274. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Bell, E, Boyle, D., Tayton, J. and Floyd, K. (2014). *The Isles of Scilly Seabird Recovery Project: Technical Report for the Rat-removal Phase on St Agnes and Gugh, October 2013 to April 2014*. Unpublished client report to the Isles of Scilly Seabird Recovery Project Partnership. Blenheim, New Zealand: Wildlife Management International Ltd.

Bell, E. and Cropper, A. (2016). *The Isles of Scilly Seabird Recovery Project: Final Check Report Following the Eradication of Brown Rats (*Rattus norvegicus*) from St Agnes & Gugh, 6 January to 18 February 2016*. Unpublished client report to the Isles of Scilly Seabird Recovery Project Partnership. Blenheim, New Zealand: Wildlife Management International Ltd.

Bell, E.A. (2019). 'It's not all up in the air: the development and use of ground-based rat eradication techniques in the UK'. In: C.R. Veitch, M.N. Clout, A.R Martin, J.C Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 79–87. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Bell, E., Bell, M., Morgan, G. and Morgan, L. (2019). 'The Recovery of Seabird Populations on Ramsey Island, Pembrokeshire, Wales Following the 1999/2000 Rat Eradication'. In: C.R. Veitch, M.N. Clout, A.R Martin, J.C Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 539–544. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Brown, D., Pitt, W. and Tershy, B. (2013). *A Review of the Wake Island Rat Eradication Project*. Honolulu, Hawaii: US Fish and Wildlife Service.

Bryce, R., Oliver, M., Davies, L., Gray, H., Urquhart, J. and Lambin, X. (2011). 'Turning back the tide of American mink invasion at an unprecedented scale through community participation and adaptive management'. *Biological Conservation*. 144: 575–583.

Caithness, T. and Williams, G. (1971). 'Protecting birds from poisoned baits', *New Zealand Journal of Agriculture* 122: 38–43.

Campbell, K., Beek, J., Eason, C., Glen, A., Godwin, G., Gould, F., Holmes, N., Howald, G., Madden, F., Ponder, J., Threadgill, D., Wegmann, A. and Baxter, G. (2015). 'The next generation of rodent eradications: Innovative technologies and tools to improve species specificity and increase their feasibility on islands'. *Biological Conservation* 185: 47–58.

DIISE. (2015). The Database of Island Invasive Species Eradications, developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand. <http://diise.islandconservation.org>.

Eason, C., Ogilvie, S., Miller, A., Henderson, R., Shapiro, L., Hix, S., MacMorran, D. and Murphy, E. (2008). 'Smarter Pest Control Tools with Low-residue and Humane Toxins'. In: R.M. Timm and J.M. O'Brien (eds) *Proceedings of the 23rd Vertebrate Pest Conference* pp. 148–153. Davis, California, USA: University of California.

Eaton, M., Aebischer, N., Brown, A., Hearn, R., Lock, L., Musgrove, A., Noble, D., Stroud, D. and Gregory, R. (2015). 'Birds of conservation concern 4: the population status of birds in the UK, Channel Islands and Isle of Man'. *British Birds* 108: 708–746.

Hartley, L., O'Connor, C., Waas, J. and Matthews, L. (1999). 'Colour preferences in North Island robins (*Petroica australis*): Implications for deterring birds from poisonous baits'. *New Zealand Journal of Ecology* 23: 255–259.

Howald, G., Donlan, C., Galvan, J., Russell, J., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Howie, A., Jelbert, K. and Doyle, J. (2007). 'Mammals in Cornwall and the Isles of Scilly: Project Findings'. In: S. Board (ed.) *Celebrating Cornwall's Mammals – from Dormice to Dolphins*, pp. 1–70. Cornwall, UK: Environmental Records Centre for Cornwall and Isles of Scilly (ERCCIS).

Jones, H., Tershy, B., Zavaleta, E., Croll, D., Keitt, B., Finkelstein, M. and Howald, G. (2008). 'Review of the global severity of the effects of invasive rats on seabirds'. *Conservation Biology* 22: 16–26.

Lock, L., Brown, A., Clitherow, J., Mawer, D. and St Pierre, P. (2006). *Isles of Scilly Seabird Conservation Strategy*. Unpublished client report to the Isles of Scilly Seabird Conservation Group. Exeter, UK: Royal Society for the Protection of Birds (RSPB).

Lock, L., Brown, A., Clitherow, J., Mawer, D. and St Pierre, P. (2009). *Isles of Scilly Seabird Conservation Strategy*. Unpublished client report to the Isles of Scilly Seabird Conservation Group. Exeter, UK: Royal Society for the Protection of Birds (RSPB).

Matheson, C. (1962). *Brown Rats*. London: Sunday Times Publications. (*Animals of Britain, No 16*).

McCann, J. (2005). 'The introduction of the brown rat (*Rattus norvegicus*)'. *Somerset Archaeology and Natural History Journal 2005*: 139–141.

Merton, D., Climo, G., Laboudallon, V., Robert, S. and Mander, C. (2002). 'Alien mammal eradication and quarantine on inhabited islands in the Seychelles'. In: C.R. Veitch and M.N. Clout (eds.). *Turning the tide: the eradication of invasive species,* pp. 182–198. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Moors, P.J. and Atkinson, I.A.E. (1984). 'Predation on Seabirds by Introduced Animals and Factors Affecting its Severity'. In: J.P. Croxall, P.G.H. Evans and R.W. Schreiber (eds.) *Status and Conservation of the World's Seabirds,* pp. 667–690. ICBP Technical Publication 2. Cambridge, UK: International Council for Bird Preservation (ICBP).

Oppel, S., Beaven, B., Bolton, M., Vickery, J. and Bodey, T. (2011). 'Eradication of invasive mammals on islands inhabited by humans and domestic animals'. *Conservation Biology* 25: 232–240.

Parks and Wildlife Service. (2014). *Evaluation Report: Macquarie Island Pest Eradication Project, August 2014*. Hobart, Tasmania: Department of Primary Industries, Parks, Water and Environment. <http://www.parks.tas.gov.au/file.aspx?id=31160>.

Pearson, J. (2016). 'Manx shearwater and storm petrel numbers increase on St Agnes and Gugh'. *Seabirds South West* 8: 2.

Pearson, J., St Pierre, P., Lock, L., Buckley, P., Bell, E., Mason, S., McCarthy, R. Garratt, W., Sugar, K. and Pearce, J. (2019). 'Working with the local community to eradicate rats on an inhabited island: securing the seabird heritage of the Isles of Scilly'. In: C.R. Veitch, M.N. Clout, A.R Martin, J.C Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 670–678. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Parslow, R. (2007). *The Isles of Scilly*. London, UK: HarperCollins.

Rood, J. (1965). 'Observations on the home range and activity of the Scilly shrew'. *Mammalia* 29(4): 507–516.

Russell, J. and Broome, K. (2016). 'Fifty years of rodent eradications in New Zealand: another decade of advances'. *New Zealand Journal of Ecology* 40: 197–204.

Rymer, D. (2013). *Genetic Resistance to Anticoagulant Rodenticides in Norway Rats (Rattus norvegicus): Findings from St. Agnes and Gugh, the Isles of Scilly, June 2013*. Unpublished technical report prepared for IOSSRP partners. Reading, UK: University of Reading.

Spencer-Booth, Y. (1963). 'A coastal population of shrews *C. s. cassiteridum*'. *Proceedings of the Zoological Society of London* 140: 322–326.

Stanbury, A., Thomas, S., Aegerter, J., Brown, A., Bullock, A., Eaton, M., Lock, L., Luxmoore, R., Roy, S., Whitaker, S. and Oppel, S. (2017). 'Prioritising islands in the United Kingdom and crown dependencies for the eradication of invasive alien vertebrates and rodent biosecurity'. *European Journal of Wildlife Research* 63: 31. < https://doi.org/10.1007/s10344-017-1084-7>.

Taylor, R., Kaiser, G. and Drever, M. (2000). 'Eradication of Norway rats for recovery of seabird habitat on Langara Island, British Columbia'. *Restoration Ecology* 8: 151–160.

Thomas, B. and Taylor, R. (2002). 'A history of ground-based rodent eradication techniques developed in New Zealand, 1959–1993'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 301–310. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Thomas, S., Brown, A., Bullock, D., Lock, L., Luxmoore, R., Roy, S., Stanbury, A. and Varnham, K. (2017). 'Island restoration in the UK – past present and future'. *British Wildlife* April 2017: 231–242.

Towns, D. and Broome, K. (2003). 'From small Maria to massive Campbell: Forty years of rat eradications from New Zealand islands'. *New Zealand Journal of Zoology* 30: 377–398.

Weser, C. and Ross, J.G. (2013). 'The effect of colour on bait consumption of kea (*Nestor notabilis*): Implications for deterring birds from toxic baits'. *New Zealand Journal of Zoology* 40(2): 137–144.

Zonfrillo, B. (2001). Wildlife Conservation on Ailsa Craig. The Thomas Duncan Memorial Lectures. Girvan, Scotland: Friends of the McKechnie Institute.

Rat eradication in the Pitcairn Islands, South Pacific: a 25-year perspective

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Abstract This essay offers a 25-year overview of efforts to remove Pacific rats (*Rattus exulans*) from the four islands of the Pitcairn group. Following the 1991–1992 discovery that rats were severely reducing breeding success of gadfly petrels (*Pterodroma* spp.), Wildlife Management International proposed eradication. Eradication success was achieved using ground-based baiting on the small atolls of Ducie and Oeno in 1997, and there is now evidence of petrel recovery on Oeno, but two eradication attempts on inhabited Pitcairn (1997 and 1998) failed. By the early 2000s, the development of aerial baiting through the 1990s placed an eradication operation on the fourth island, Henderson, within reach. Preparatory fieldwork in 2009 allayed doubts in two key areas: the feasibility of maintaining a captive "back-stop" Henderson rail (*Porzana atra*) population, and bait uptake by crabs (*Coenobita* spp.). Royal Society for the Protection of Birds (RSPB) expertise secured the necessary funding of £1.5 million, and 75 tonnes of brodifacoum-containing bait were dropped in August 2011. Despite extensive mortality of free-living rails, the population, supplemented by released captive birds, returned to pre-operational levels in 2–3 years. Meanwhile those tending captive rails saw no rat sign before leaving Henderson in November 2011. Unfortunately, a rat was sighted in March 2012, and continuing rat presence confirmed in May 2012. Subsequently rat numbers have returned to pre-operational levels without any sign of population 'overshoot' as observed on Pitcairn. Genetic analysis suggests around 80 rats, roughly 1 in 1,000, survived the bait drop. With no evidence of imperfect bait coverage or deficiencies in bait quality or brodifacoum resistance, it seems some animals chose not to eat bait. Choice tests on Henderson Island rats suggest some rats prefer natural foods over bait. This adverse situation may have been exacerbated because, in August 2011, natural fruits were more abundant than anticipated due to drought earlier in the year. To overcome rat preference for natural food, any second Henderson attempt might benefit from more attractive bait. Without such developments, a second attempt risks another failure. Henderson's biota will survive the delay.

Keywords: brodifacoum, Ducie, Henderson, Henderson rail, Oeno, *Pterodroma*

INTRODUCTION

The Sir Peter Scott Commemorative Expedition to the Pitcairn Islands of 1991–1992 involved 35 personnel in the field over a span of 15 months. While short periods were spent on the sole inhabited island of Pitcairn (500 ha) and the low atolls of Oeno (c. 60 ha) and Ducie (c. 75 ha), Henderson Island (4300 ha) was the principal study site. Since Henderson had been designated a World Heritage Site in 1988 "as one of the last near-pristine limestone islands of significant size in the world" (<http://whc.unesco.org/en/list/487>), it had been appreciated that the natural history of the island was incompletely documented. The expedition aimed to rectify this omission, bringing together expertise in archaeology, geology and many branches of natural history.

One of the Expedition's unexpected findings was the very low breeding success of gadfly petrels (*Pterodroma* spp.) on Henderson: ca. 5% among Murphy's petrels (*P. ultima*), 10% in Kermadec petrels (*P. neglecta*), and 15–20% in Herald (*P. heraldica*) and Henderson petrels (*P. atrata*) (Brooke, 1995). This was especially concerning in the case of Henderson Petrels, split from Herald Petrels as a result of expedition work (Brooke & Rowe, 1996), endemic to Henderson and therefore without any source of immigrants to rescue the situation, and potentially on a downward trajectory to extinction within a few centuries (Brooke, et al., 2010a).

Field observations showed that the cause of this low breeding success was predation by Pacific rats (*Rattus exulans*), introduced to the island by Polynesians settlers about 700–800 years ago (Weisler, 1994). Hatching success was apparently not substantially reduced by rats. Rather, the problem arose in the first week after hatching, especially when the chick moved from under to beside the parent. Then the rats approached, pulled the chick away from the nest site, even in the presence of a brooding parent, and ate it (Brooke, 1995).

Observations on the atolls of Oeno and Ducie were too intermittent to establish whether rats there had a similar impact on the breeding success of petrels. However, the fact that petrel densities were 1–2 orders of magnitude higher on Oeno and Ducie than on Henderson prior to the eradications on the atolls suggested that rat impact was less, if not negligible. Probably because of the presence of rats and feral cats (*Felis catus)*, petrels do not breed on Pitcairn.

After these findings had entered the public domain via the expedition report (Pitcairn Islands Scientific Expedition, 1992) and a special volume of the Biological Journal of the Linnean Society (Benton & Spencer, 1995), the late Brian Bell of Wildlife Management International contacted the author to propose rat eradication in the Pitcairn Islands (Bell & Bell, 1998). At this time, the mid-1990s, an eradication on Henderson was not feasible using ground-based methods. Therefore, the proposal was for eradications on Oeno and Ducie using tested ground-based methods to benefit three gadfly petrel species but, crucially, not the Henderson Petrel which was not confirmed as a nesting species on either atoll.

ACTIONS

Oeno and Ducie

The modest extent and flat accessible topography of the atolls meant that the proposed eradication campaigns were likely to be successful, given prior achievements elsewhere (Towns & Broome, 2003). The eventual source of funding was the UK's Department for International Development (DfID) whose interest lay principally in Pitcairn Island and its people. For this reason, the programme linked eradications on Oeno and Ducie, offering clear biodiversity gains with limited risk of failure, to an eradication attempt on Pitcairn where the risks of failure were higher

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because of the rugged and heavily vegetated topography and the complications associated with human presence. Nonetheless the project proceeded in late 1997 with approximately £100,000 of funding for Pitcairn and Oeno from DfID and a further £20,000 for Ducie from the World Wide Fund for Nature (Bell & Bell, 1998).

Success was duly achieved on Oeno and Ducie by hand-laying of bait (baiting rate unspecified) on a 25 m grid (Bell & Bell, 1998). The Oeno eradication has been followed by growth of the population of the seabird species most easily censused, Murphy's petrel, at an annual rate of 6% (Brooke, et al., 2017). There are no post-eradication census data from Ducie.

Pitcairn

Eradication was not achieved on Pitcairn in 1997. There, preceding bait laying, the endeavour of cutting a 25 m grid of paths through the dense scrub cloaking the island's extremely severe terrain taxed the endurance of the WMIL team, especially since, in the absence of prior reconnaissance, the severity of the task ahead had not been appreciated. Coverage of the cliffs was probably incomplete. A lesson was learnt: future operations of this magnitude must involve prior on-site reconnaissance by key personnel.

The WMIL team departed shortly after the completion of bait laying (overall baiting rate not specified), entrusting the task of follow-up monitoring to the Pitcairn Islanders (Bell & Bell, 1998). Given the many calls on the islanders' time, and their lack of appropriate expertise, this strategy was probably a mistake. With the benefit of hindsight, it would have been better if extra costs had been incurred and logistical difficulties overcome to allow some dedicated team members to remain on Pitcairn to detect any residual rat presence. While this change in protocol would not have guaranteed a successful outcome, it could only have increased the probability of success.

WMIL returned in 1998 to attempt to rectify the 1997 eradication failure. Unfortunately, the outcome reprised that of 1997 despite more intensive monitoring after the initial baiting, coupled with spot-laying of bait wherever rat sign was detected (Bell, 1998).

A striking feature of these failures was not simply the rapidity with which rats recovered to their pre-bait levels which, the reports of Pitcairners suggested, happened within 18–24 months. There was also a universal impression among the islanders and indeed myself on a visit in 2000 that numbers overshot the status quo ante, to a startling extent. For example, rats were frequently encountered in homes, even in cooking ovens left ajar. A possible explanation of this 'overshoot', that cannot be confirmed by any formal existing trapping or density data, is that, after the reduction in rat numbers due to baiting, a large amount of food accumulated, for example on or below Pitcairn's abundant fruit trees. This surfeit possibly nourished the extreme increase in rat numbers.

Henderson

Following the successful eradication of rats from several large New Zealand islands using aerial baiting techniques during the 1990s (Towns & Broome, 2003) and from 113 km^2 Campbell Island in 2001 (McClelland & Tyree, 2002), the possibility of an eradication project on Henderson Island using aerial baiting moved up the agenda. A feasibility report delivered a favourable verdict, subject to two caveats (Brooke & Towns, 2008). The first was that, in the areas of high land crab (*Coenobita* spp.) density behind Henderson's beaches, it should be demonstrated that sufficient bait could be scattered so that, even after substantial bait removal by crabs, enough bait remained to permit all rats to consume a fatal quantity. The second concerned the endemic flightless Henderson rail (*Porzana atra*). Given the recorded susceptibility of rails to brodifacoum in cereal bait (Eason, et al., 2002) – as would be used in a Henderson operation – there was a need to demonstrate that Henderson rails could be caught and then kept healthy in captivity. In the worst-case scenario, the elimination of the wild population during the eradication operation, the captives, once released after the disappearance of bait, would become the founders of the new wild population.

Both these issues were successfully addressed by a field expedition in August/September 2009 (Brooke, et al., 2010b; Cuthbert, et al., 2012), paving the way for an eradication operation in 2011. The feasibility report (Brooke & Towns 2008) suggested the late winter months of September/October as the period of lowest food availability and therefore the most suitable for bait-laying. This suggestion was based on a 1-year study of plant phenology (Brooke, et al., 1996), and drew on the fact that *Rattus exulans* includes a proportion of vegetable material in its diet. In the absence of any data whatsoever on the intra-annual variation in the availability of invertebrates and their contribution to the rats' diet, this potential factor

Table 1 Summary table of rat eradication operations on the four Pitcairn Islands. Details from Bell & Bell (1998), Bell (1998), Torr & Brown (2012) and E. Bell (pers. comm.).

Island	Type	Method	Year baited	Month(s) baited	Bait type	No. baitings	Successful?
Pitcairn	Volcanic	Hand broadcast	1997	June – August	Pestoff 20R; wax-covered chocolate bait for 3rd baiting	3	No
Pitcairn	Volcanic	First two: hand broadcast. Then bait stations and spot-laying	1998	April – July	Pestoff 20R. Later baitings supplemented by wax-covered chocolate bait	3+	No
Oeno	Atoll	Hand broadcast	1997	July – August	Pestoff 20R	2	Yes
Ducie	Atoll	Hand broadcast	1997	November	Pestoff 20R	2	Yes
Henderson	Makatea	Aerial	2011	August	Pestoff 20R	2	No

could not be addressed in project planning. In the event, late August 2011 became the provisional project date. Fund-raising for the £1.5 million budget proceeded apace under the aegis of the Royal Society for the Protection of Birds (RSPB).

The operation was logistically complex involving the 298-tonne Alaskan crab-fishing vessel, the *Aquila*, sailing from the United States. Carrying two helicopters, the *Aquila* undertook other rat eradications in the central Pacific (Palmyra Atoll followed by Enderbury and Birnie in the Phoenix Islands) before loading the 76 tonnes of bait required for Henderson in Samoa. She then sailed east to Henderson.

Meanwhile the rail-catching team were landed on the island on 8 July 2011. The team immediately noticed that fruit was more abundant than expected – of which more anon. Catching of rails proceeded satisfactorily but adapting birds to captivity proved more problematical than in 2009, and 22 died before the solution was found, enticing the birds to the food bowls with live bait such as immobilised moths (Oppel, et al., 2016). In retrospect, it appears that, by chance, the smaller 2009 batch of rails (26 caught: two died) simply included few birds reluctant to adapt to captivity (Brooke, et al., 2010b; Brooke, et al., 2012).

The losses meant that the number of captive rails, 75, at the time of the *Aquila*'s arrival on 14 August, was lower than the target of 100 birds, but not so much lower as to cause a postponement or cancellation of baiting. The details of bait spreading are covered in the report of the project leaders (Torr & Brown, 2012). Overall the process went remarkably smoothly, with bait buckets filled on board the *Aquila*, obviating the need for any onshore storage of bait. GPS mapping of the island, prior to the first bait drop, revealed the area to be 43 km^2, an enlargement over the 37 km^2 that had been the basis for planning. Fortunately there was sufficient contingency bait that this unexpected expansion necessitated no adjustment of planned bait densities.

Excluding enhanced bait application in the areas of high crab density (Cuthbert, et al., 2012) and in the coconut groves, the application rate was 10 kg/ha of pellets (brodifacoum concentration of 20 ppm) over the majority of the island for the first drop carried out between 15 and 17 August, and 6 kg/ha during the second bait drop on 21 and 22 August. The 5-day interval between drops was slightly less than originally planned because settled weather prompted a decision to proceed immediately, rather than delay until the planned interval of seven days (Torr & Brown, 2012).

The immediate impact of the bait drop on the wild free-living rails was dramatic – as it was on rats. Sixteen of 16 rails that were radio-tagged, and whose fate could therefore be determined with certainty, died. However, mortality island-wide was not total. The best estimate is that 93 percent of free-living rails died, leaving c. 500 survivors (Oppel, et al., 2016). A few weeks after the drop, these birds began breeding. Their numbers were supplemented in October and November by the release of the captive birds, and the population has since completely recovered (Oppel, et al., 2016). Although, in the event, the captive birds were not essential for the species' persistence, the outcome was in doubt in the anxious days after the bait drops, and there is no question that a similar captive rail population must be established, should there be another eradication attempt in the future. This recommendation only gains force if, for example, the bait drops occur over a longer time period, or there are three drops instead of two. No other bird species is known to have been adversely affected by the bait drops on Henderson.

At the time the team caring for the captive rails left Henderson in November, three months after the bait drop, no signs of surviving rats had been noticed. Disastrously, a surviving rat was seen and captured on video by a visitor in March 2012. A follow-up visit, in May, confirmed continuing rat presence and, as expected, rat numbers had returned to 'normal' about two years later with no sign of the overshoot noted on Pitcairn (Bond, et al., 2019).

The eradication failure immediately prompted a review of the operation and a search for possible operational errors. None has been discovered (Internal RSPB documents). There were no apparent gaps in bait coverage, and none of the batches of bait, deliberately retained for post-operational testing, was shown to have incorrect toxin loading. Such post-hoc testing cannot absolutely exclude the remote possibility that some bags of bait did not have toxic baits, a factory error. Finally, fieldwork on Henderson in 2013 tested the rats, presumably animals descended by several generations from the actual survivors, for resistance to brodifacoum. No such resistance was found (Churchyard, et al., 2015).

Genetic studies after failure excluded the possibility that Pitcairn or other islands elsewhere in the Pacific had been a source of rats that had somehow reached Henderson and re-populated the island. In any case, knowledge of boat traffic made this scenario extremely unlikely. Thus, there had been a failure of eradication and not a re-introduction. Because rat samples had been secured before the operation, and were then obtained afterwards, it was possible to use the change in microsatellite allele frequency to estimate how many rats survived (Amos, et al., 2016). The answer was about 80 individuals, very roughly one in a thousand of the rats present on Henderson before the operation (Brooke, et al., 2010b). It is a total compatible with the absence of observations of living rats for around seven months after the bait drops.

Can this total, neither indicating a tiny number of survivors that might be ascribed to chance nor several hundreds, even thousands, indicating serious deficiencies in operational protocol, suggest improvements that might be made for a second attempt?

Mention has already been made of the fact that the rail team encountered more fruit than expected on Henderson in July 2011. This was probably a delayed consequence of a drought that afflicted Pitcairn, and presumably also Henderson, from November 2010 to March 2011. When this drought broke, it is likely that the trees became greener, flowered and then fruited, at a time that was inopportune for the rat eradication, especially if flowering and fruiting were accompanied by increased numbers of invertebrates. Although there has been one year-long study of the leafing, flowering and fruiting phenology of Henderson's plants (Brooke, et al., 1996), this is clearly inadequate to understand how plant phenological schedules may change from year to year, and how they are altered by annual variations in weather. That would require around 20 years of study, an impossible task on isolated Henderson. Thus, tailoring a rat eradication to a particular window of plant food scarcity will always be difficult, if not impossible. And no subsequent findings have altered the cautious recommendations of the feasibility study (Brooke & Towns, 2008), derived from the Brooke, et al. (1996) plant phenology study, that September or a month either side is the most suitable period.

Compounding this problem is that the operation must be set in train – boats chartered, bait ordered and so forth – at least six months before baiting (Parkes & Fisher, 2017). It would, in theory, be possible to cancel an operation at a late stage, for instance if there were reports of a surge in fruit abundance, but the penalties for such a late cancellation could well approach £500,000.

Following their helicopter flights across the island in 2011, the pilots reported, to universal surprise, a few tens of coconut trees (*Cocos nucifera*) emerging from the canopy growing on the raised atoll lagoon. Since the ground is about 30 m above sea level, these trees must have involved human intervention. They were certainly not planted by members of the Sir Peter Scott Commemorative Expedition of 1991–1992. There are two other known possibilities. The first is that the Pitcairners who, during World War II, cut a network of paths across the island, some several kilometres from the coast, were responsible. Another possibility is that the helicopter presence associated with the visit of the USS Sunnyvale in 1966 provided an opportunity for coconuts to be 'bombed' from overhead.

However the coconuts arrived, it is not surprising that they have been growing unknown for decades since most parts of this impenetrable island have remained unvisited for centuries. The relevance of these observations is that the research visit of 2013 (Churchyard, et al., 2015) conducted captive trials to test which natural foods, if any, were preferred by rats to bait pellets. Given a four-way choice between coconut (removed from its shell), *Myrsine* fruits, *Pandanus* nuts and Pestoff bait pellets, coconut was preferred, with pellets second. Moreover 11 of 30 rats ate no pellets whatsoever in a 3-day trial (details in Churchyard, et al., 2015). These findings were confirmed by further similar research in 2015 that also indicated the preference for natural food could not be overcome by increasing the relative abundance of bait pellets, an experimental adjustment equivalent to increasing the bait application rate during helicopter operations (Lavers, et al., 2016).

Although the coconut groves behind the North and North-West Beaches received deliberately high applications of bait pellets (Torr & Brown, 2012), this was not the case for the unknown isolated trees in mid-island. However, there are no data bearing on where on the island the 80 surviving rats lived and whether their home ranges were in the vicinity of coconuts.

It is evident that an absence of coconuts is not a sine qua non of a successful rat eradication. Success was achieved on Oeno (coconuts present) and Ducie (no coconuts). Projects failed on Henderson and Pitcairn, both with coconuts. More generally, numerous islands with coconuts have been cleared of rats, including the island of Palmyra (</www.fws.gov/refuges/news/PalmyraAtollRatFree.html>) visited by the *Aquila* two months before it reached Henderson.

Although Henderson's coconuts could have contributed to the project's failure (Holmes, et al., 2015), removing this possible cause would not be easy. Reaching every mid-island coconut would require a helicopter to insert a small group of "coconut destroyers" close to each tree, perhaps via a winch. Their task would be to destroy all the nuts and possibly the tree as well. That would still leave the coastal coconuts. It is unlikely that their total destruction would be countenanced by the Pitcairn Islanders and, in any case, their flowers are a significant food of the endemic Stephen's lorikeet (*Vini stepheni*) (Trevelyan, 1995). Even destroying or removing off-island all the fallen nuts, weighing several tens of tonnes, would not be easy. But the practicalities should be explored.

The discussion has reached the stage where the 2011 eradication appears to have failed, not because of any operational blemishes and not because of any brodifacoum-resistance but because a small number of rats failed to consume a fatal dose, approximately one pellet, of bait. Instead they chose to eat natural food in preference to bait (Keitt, et al., 2015). This picture is entirely compatible with the more general observation that tropical rodent eradications are less likely to be successful than those on temperate islands (Russell & Holmes, 2015)

If a second eradication attempt is to have an improved chance of success, some aspects of the protocol may have to change. The impracticalities of guaranteeing that a bait drop occurs at a time of minimal food abundance have already been discussed. The challenge of reducing the availability of coconuts needs further thought. Finally, I strongly advocate consideration of a further option, the development of a more attractive bait formulation that will entice even those rats that might have shunned the pellets used in 2011 to eat bait. It will probably never be known whether these crucial rats did not eat bait pellets because a more palatable natural food was available, and/or whether illness or pregnancy affected their appetite for novel foods (neophobia). Altering the formulation of bait pellets by the addition of such flavours as chocolate or peanut has already been trialled by Orillion, the manufacturers of PestOff pellets (Bill Simmons, pers. comm.). However, it remains uncertain whether these changes would demonstrably reduce the risk to an operation of such rat behaviours as neophobia.

Although modest alteration of pellets may not engender regulatory problems in UK Overseas Territories (Bill Simmons, pers. comm.), the development of pellets of enhanced attractiveness could pose technical problems. For example, any additives must not make the pellets more 'sticky' and liable to clog the hoppers underslung from bait-distributing helicopters. But, optimistically, such developments will occur as New Zealand develops the expertise to rid itself of alien predators by 2050, as other countries follow New Zealand's lead, and as the relative intractability of tropical islands is addressed.

Meanwhile, from my 25-year perspective, Henderson will probably not change greatly in the next decade. A patient approach will hugely increase the likelihood that any second rat eradication attempt on Henderson is made when the chances of success are demonstrably higher. It will also avoid the mistake made on Pitcairn, of undertaking an eradication project because money was available rather than because a rational, even hard-nosed, assessment confirmed that the chances of success and the biodiversity gains of success outweighed the costs and risks of failure.

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REFERENCES

Amos, W., Nichols, H.J., Churchyard, T. and Brooke, M. de L. (2016). 'Rat eradication comes within a whisker! A case study of a failed project from the South Pacific'. *Royal Society Open Science*. DOI: 10.1098/rsos.160110.

Bell, B.D. and Bell, E. (1998). *Habitat restoration: Pitcairn Islands, South Pacific. Eradication of rats and feral cats, April to December 1997*. Unpublished report. Wellington, New Zealand: Wildlife Management International.

Bell, D. (1998). *Habitat restoration: Pitcairn Islands, South Pacific. Eradication of rats and feral cats, April to September 1998*. Unpublished report. Wellington, New Zealand: Wildlife Management International.

Benton, T.G. and Spencer T. (eds.) (1995). *The Pitcairn Islands: Biogeography, Ecology and Prehistory*. London, U.K.: Linnean Society and Academic Press.

Bond, A.L., Cuthbert, R.J., McClelland, G.T.W., Churchyard, T., Duffield, N., Havery, S., Kelly, J., Lavers, J.L., Proud, T., Torr, N., Vickery, J.A., and Oppel, S. (2019). 'Recovery of introduced pacific rats following a failed eradication attempt on Subtropical Henderson Island, South Pacific Ocean'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 167–174. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Brooke, M.de L. (1995). 'The breeding biology of the gadfly petrels *Pterodroma* spp. of the Pitcairn Islands: Characteristics, population sizes and controls'. *Biological Journal of the Linnean* Society 56: 213–231.

Brooke, M. de L., Jones, P.J., Vickery, J.A. and Waldren, S. (1996). 'Seasonal patterns of leaf growth and loss, flowering and fruiting on a sub-tropical Central Pacific Island'. *Biotropica* 28: 164–179.

Brooke, M.de L. and Rowe, G. (1996). 'Behavioural and molecular evidence for specific status of dark and light morphs of the Herald petrel *Pterodroma heraldica'. Ibis* 138: 420–432.

Brooke, M. de L. and Towns, D.R. (2008). *A Feasibility Study for the Eradication of Kiore* Rattus exulans *from Henderson Island*. Unpublished report. Sandy, UK: Royal Society for the Protection of Birds.

Brooke, M. de L., O'Connell, T.C., Wingate, D., Madeiros, D., Hilton, G.M. and Ratcliffe, N. (2010a). 'The potential for rat predation to cause decline of the globally threatened Henderson petrel *Pterodroma atrata*: Evidence from population modelling, the field and stable isotopes'. *Endangered Species Research* 11: 47–59.

Brooke, M. de L., Cuthbert, R. J., Henricson, A., Torr, N., Warren, P. and O'Keefe, S. (2010b). *Towards Rat Eradication on Henderson Island: Fieldwork Report August–September 2009*. Unpublished report. Sandy, UK: Royal Society for the Protection of Birds.

Brooke, M. de L., Harrison, G. and Cuthbert, R. J. (2012). *The Capture, Husbandry and Captive Breeding of Henderson Rails During the Henderson Island Rat Eradication and the Impact of the Project on Native Landbirds.* Unpublished report. Sandy, UK: Royal Society for the Protection of Birds.

Brooke, M. de L., Bonnaud, E., Dilley, B.J., Holmes, N.D., Jones, H.P., Provost, P., Rocamora, G., Ryan, P.G., Surman, C. and Buxton R.T. (2017). 'Seabird population changes following mammal eradications on islands'. *Animal Conservation* 21: 3–12.

Churchyard, T., Proud, T., Brooke, M. de L., O'Keefe, S., Warren, P. and Rodden, M. (2015). *Henderson Island Trip Report: 30th June–31st August 2013*. Unpublished report. Sandy, UK: Royal Society for the Protection of Birds.

Cuthbert, R.J., Brooke, M. de L. and Torr, N. (2012). 'Overcoming hermit crab interference during rodent baiting operations: A case study from Henderson Island, South Pacific'. *Wildlife Research* 39: 70–77.

Eason, C.T., Murphy, E.C., Wright, G.R.G. and Spurr E.B. (2002). 'Assessment of risks of brodifacoum to non-target birds and mammals in New Zealand'. *Ecotoxicology* 11: 35–48.

Holmes, N.D., Griffiths, R., Pott, M., Alifano, A., Will, D., Wegmann, A.S. and Russell, J.C. (2015). 'Factors associated with rodent eradication failure'. *Biological Conservation,* 185: 8–16.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation,* 185: 17–26.

Lavers, J.L., McClelland, G.T.W., MacKinnon, L., Bond, A.L., Oppel, S., Donaldson, A.H., Duffield, N.D., Forrest, A.K., Havery, S.J, O'Keefe, S., Skinner, A., Torr, N., and Warren, P. (2016). 'Henderson Island Expedition Report: May–November 2015'. *RSPB Research Report* 57. Sandy, UK: Royal Society for the Protection of Birds.

McClelland, P. and Tyree, P. (2002). 'Eradication – the clearance of Campbell Island'. *New Zealand Geographic*. 58: 86–94.

Oppel, S., Bond, A.L., Brooke, M. de L., Harrison, G., Vickery, J.A. and Cuthbert, R.J. (2016). 'Temporary captive population and rapid population recovery of an endemic flightless rail after a rodent eradication operation using aerially distributed poison bait'. *Biological Conservation* 204: 442–448.

Parkes, J. and Fisher, P. (2017). 'Review of the Lehua Island Rat Eradication Project 2009'. *Pacific Cooperative Studies Unit Technical Report* 195. Honolulu, HI: University of Hawai'i at Mānoa, Department of Botany.

Pitcairn Islands Scientific Expedition. (1992). *Sir Peter Scott Commemorative Expedition to the Pitcairn Islands, 1991–1992*. Unpublished expedition report. University of Cambridge, UK: Department of Zoology.

Russell, J.C. and Holmes N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Torr, N. and Brown, D. (2012). *Henderson Island Restoration Project: Post-operational Report.* Unpublished report. Sandy, UK: Royal Society for the Protection of Birds.

Towns, D.R. and Broome, K.G. (2003).' From small Maria to massive Campbell: Forty years of rat eradications from New Zealand islands'. *New Zealand Journal of Zoology* 30: 377–398.

Trevelyan, R. (1995). 'The feeding ecology of Stephen's lory and nectar availability in its food plants'. *Biological Journal of the Linnean Society* 56: 185–197.

Weisler, M.I. (1994). 'The settlement of marginal Polynesia: New evidence from Henderson Island'. *Journal of Field Archaeology* 24: 83–102.

House mice on islands: management and lessons from New Zealand

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Abstract The impacts of house mice (*Mus musculus*), one of four invasive rodent species in New Zealand, are only clearly revealed on islands and fenced sanctuaries without rats and other invasive predators which suppress mouse populations, influence their behaviour, and confound their impacts. When the sole invasive mammal on islands, mice can reach high densities and influence ecosystems in similar ways to rats. Eradicating mice from islands is not as difficult as previously thought, if best practice techniques developed and refined in New Zealand are applied in association with diligent planning and implementation. Adopting this best practice approach has resulted in successful eradication of mice from several islands in New Zealand and elsewhere including some of the largest ever targeted for mice; in multi-species eradications; and where mouse populations were still expanding after recent invasion. Prevention of mice reaching rodent-free islands remains an ongoing challenge as they are inveterate stowaways, potentially better swimmers than currently thought, and prolific breeders in predator-free habitat. However, emergent mouse populations can be detected with conventional surveillance tools and eradicated before becoming fully established if decisive action is taken early enough. The invasion and eventual eradication of mice on Maud Island provides a case study to illustrate New Zealand-based lessons around mouse biosecurity and eradication.

Keywords: biosecurity, eradication, impacts, invasive rodents, Maud Island

INTRODUCTION

The house mouse (*Mus musculus*) established in New Zealand (NZ) around 1830, about 550 years after the first rodent to arrive, the Pacific rat or 'kiore' (*Rattus exulans*), 60 years after Norway rats (*R. norvegicus*) and 30 years before ship rats (*R. rattus*) (Atkinson, 1973). Mice in New Zealand have traces of ancestry from three subspecies – *Mus musculus domesticus*, *M. m. castaneus* and *M. m. musculus* – however *M. m. domesticus* is the dominant subspecies (King, et al., 2016; Veale, et al., 2018). The hybridisation of subspecies could have occurred before or after the mice arrived in NZ (Veale, et al., 2018).

Today mice are widespread and common throughout NZ but not as common as ship rats. Mice increase in numbers quickly in response to pulses of food and reductions in ship rat abundance (Elliott & Kemp, 2016).

Rodent colonisations of smaller islands in the NZ archipelago have different histories influenced by past human visitation and proximity to the largest islands 'North' and 'South' considered 'mainland' by New Zealanders. Of the 1065 islands >1 ha (excluding the mainland), mice established on about 42 of them (Ruscoe & Murphy 2005; Department of Conservation (DOC), unpublished data).

Action against mice for biodiversity protection goals began with efforts by NZ Wildlife Service with rodent-proof packaging of stores destined for rodent-free islands. The first eradication of mice in NZ occurred in 1984 on 2 ha Whenuakura Island, although the project targeted Norway rats, not mice (Veitch & Bell 1990).

In 1989 the first deliberate attempts to eradicate mice from islands occurred on Mana 217 ha (Hook & Todd, 1992), Rimariki 22 ha (Veitch & Bell, 1990), and Allports 16 ha, (Brown, 1993). We can identify 36 attempts to remove mice from NZ islands larger than 1 ha, 28 of them succeeded and eight failed (Appendix 1). Mice have reinvaded seven of the 28 from which they were eradicated. Some of the eradication failures could possibly be attributed to reinvasion. These figures update NZ data presented by MacKay, et al., (2007) and Howald, et al. (2007) who included eradication attempts worldwide where the eradication of mice was not always a stated goal and where the presence of mice on the island prior to eradication remained unproven.

In this paper, we explore three questions related to the management of mice on islands for biodiversity protection:

1. What do we know about the impacts of mice on NZ island ecosystems?
2. What have we learnt about eradicating mice from islands and what do we now consider best practice in NZ?
3. What have we learnt about preventing mice from establishing new populations on NZ islands?

We use the invasion of Maud Island by mice in 2013 and their successful eradication in 2014 as a case study to illustrate our lessons.

IMPACTS OF MICE

Mice often inhabit islands with other invasive species which can confound efforts to quantify mice impacts. Predators, particularly rats, can have a marked influence on the behaviour and densities of mice while simultaneously reducing and masking mice impacts (Bridgman, 2012). Removal of mice in these situations often requires simultaneous removal of other invasive mammals, thereby continuing the confusion over how to attribute recovery to the absence of mice and not the other species involved.

On islands where mice are the only invasive mammal present they usually attain higher densities, exhibit different behaviours and therefore have more conspicuous impacts on native biodiversity (Angel, et al., 2009).

Mice as bird predators

Mice eat small bird's eggs. Frogley (2013) filmed them eating quail (*Coturnix japonica*) (30 × 24 mm), zebra finch (*Taeniopygia guttata*) (14 × 9 mm) and canary (*Serinus canaria*) eggs (16 × 11 mm) from unattended used nests placed on the forest floor. Fewer of the quail eggs tested were eaten, suggesting they are near the size limit for mice to break into. Over 400 hours of filming six natural forest bird nests in podocarp-broadleaved forest at Maungatautari resulted in observation of only a single mouse visit (Watts, et al., 2017).

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Smaller seabirds such as some storm petrel species appear more vulnerable to egg and sometimes chick predation by mice although some studies suggest this has little effect on productivity (Campos & Granadeiro, 1999). Shore plover (*Thinornis novaeseelandia*) breed very successfully on Waikawa Island with mice at high densities. There is no evidence of egg predation on shore plover (egg size 37 × 26 mm) or white-faced storm petrels (*Pelagodroma marina*) (egg size 36 × 26 mm) on Waikawa Island in the presence of high mouse numbers (H. Jonas & J. Dowding pers. comm.).

The evidence for other impacts on birds in NZ is more circumstantial, for example differences in abundance of snipe (*Coenocorypha aucklandica*) and black-bellied storm petrels (*Fregetta tropica*) on Antipodes Island with mice and rodent-free islands such as Adams and Bollons (Miskelly, et al., 2006; Imber, et al., 2005).

Mice as reptile predators

On Mana Island removal of grazing livestock led to an increased mouse population due to improved habitat from rank grass. The McGregor's skink (*Oligosoma macgregori*) population declined and mice were seen eating skinks in pitfall monitoring traps. Following the eradication of mice in 1989 McGregor's skink numbers increased and they became more conspicuous (Newman, 1994).

Norbury, et al. (2014) followed the fate of translocated Otago skinks (*Oligosoma otagense*) in a fenced site which contained mice as the only mammalian predator. They observed mice attacking 25 cm adult skinks but noted skink survival rates were adequate for population persistence.

Romijn (2013) compared the capture rates of ornate skinks (*Oligosoma ornata*) between sites with and without mice present (without other predators). The site with mice had periodic control of mice to maintain densities below 21 per100 trap-nights. He found population increases at both sites but significantly higher rates in the site with no mice.

Mice were implicated in the suppression of recruitment in a shore skink (*O. smithi*) population at Tawharanui fenced sanctuary (Wedding, 2007).

Mice as invertebrate predators

Invertebrates are an important part of the broad diet of mice (Ruscoe & Murphy, 2005). St Clair (2011) compiled the known impacts of invasive rodents on island invertebrates including a range of NZ species influenced by mice.

Watts, et al. (2017) conducted a large-scale treatment switch experiment at Maungatautari in 2011–2016. Two fenced enclosures in forest had all mammalian pests removed except mice. At one site they eradicated mice and at the other allowed mice to increase. Results suggested mice suppressed beetles, spiders, earthworms and weta in both abundance and size.

Mice impacts on vegetation

Williams, et al. (2000) found mice destroy all seed they eat, rather than acting as seed dispersers. On the New Zealand mainland, seed predation by mice may affect regeneration of kauri (*Agathis australis*) (Badan, 1986), pingao (*Desmoschoenus spiralis*) and sand tussock (*Poa triodioides*) (Miller & Webb, 2001). Mouse predation on mountain beech (*Fuscospora cliffortioides*) and rimu (*Dacrydium cupressinum*) seeds not only reduces rates of seedling establishment, but may also alter the composition of forests over time (Wilson, et al., 2007). Seed predation by mice may also impede ecological restoration efforts, for example inhibiting a tree planting programme on Mana Island (Hook & Todd, 1992).

Watts, et al. (2017) found no significant impact of mice on forest seedling establishment over their five-year study. However, they noted their (predator fenced) mainland study site has been subject to modification by a range of introduced mammals for hundreds of years prior to the beginning of the study.

Other biodiversity impacts by mice

Two studies reported observations of mice eating the eggs of a NZ native fish, inanga (*Galaxias maculatus*) (Baker, 2006; Hickford, et al., 2010).

Besides the direct impacts discussed above, mice also influence other predators who use them as a food source. For example, stoats (*Mustela erminea*) will include mice in their diet. In beech (*Fuscospora* spp.) dominated forest, mast seeding events lead to high populations of mice followed by increased stoat populations with consequent impacts on native species (King & Murphy, 2005).

Mice may also provide an important year-round food resource for larger predators on islands with strongly seasonal primary food resources such as colonial nesting seabirds. They may therefore 'artificially' sustain higher predator populations through the non-seabird nesting periods.

MOUSE ERADICATION

Since 1989 developments in mouse eradication methodologies in New Zealand mirrored those of rat eradications (Towns & Broome, 2003; Broome, 2009; Russell & Broome, 2016). Aerial broadcast baiting was consistently chosen for eradications targeting mice on islands larger than 40ha (Appendix 1).

Mouse susceptibility to brodifacoum is highly variable. For example, Cuthbert, et al. (2011) had two Gough Island mice survive doses of 2.44 and 5.41 mg/kg, respectively. These individuals were subsequently offered more bait in no-choice tests and died after ingesting 12.2 and 7.14 mg/kg. Three (of 10) mice from Lord Howe Island survived doses of 5.2 mg/kg in a no-choice bait test (D. Priddel pers. comm.). A subsequent trial using 30 wild-caught Lord Howe mice allowed to feed *ad libitum* for three days resulted in 100% mortality (A. Walsh pers. comm.).

Mice usually die from about five days following the first application. For example, MacKay, et al. (2007) found no sign of surviving mice on Adele Island eight days after bait application. However, they can survive much longer (see case study) and in one laboratory trial, a warfarin-resistant mouse survived a total of 65 days after first feeding on brodifacoum laced bait (Rowe & Bradfield, 1976).

Bridgman (2012) studied the behaviour of mice in the presence of ship rats. She found ship rats strongly influenced the movements of mice, reducing home ranges and nutrition levels. This has implications for eradication projects targeting both rats and mice, reinforcing the need for comprehensive bait coverage and well-spaced multiple bait applications to allow for the dominant rats to die off and theoretically 'free up' the movement of any mice remaining.

Some projects failed to eradicate mice because they did not explicitly target them. For example, on Mokoia Island in 1989 an eradication project targeting Norway rats using bait stations spaced at 50 × 50 m subsequently found mice on the island (P. Jansen, pers. comm.). Because the eradication was designed around the home range of Norway rats, mice survived and became detectable after the rat population had crashed.

Eradications of mice on islands in NZ progressed through the 1990s with mixed success (MacKay, et al., 2007). The review of mouse eradication projects by MacKay, et al., in 2007 could not find a consistent operational factor contributing to eradication failure but recommended robust planning of future projects to rule out operational errors, thereby providing better insight into the cause of failures.

Following this recommendation, a project to eradicate mice from three islands (Adele, Tonga, Fisherman) in Tasman Bay in 2007 strictly adhered to the current agreed best practice methodology for mouse eradications (Golding, 2010). The Island Eradication Advisory Group (IEAG), a technical advisory group of the NZ Department of Conservation, updates and maintains a document providing technical advice to project managers in the planning, implementation and monitoring of rat eradications on islands (Broome, et al., 2017a).

The IEAG consider best practice for mouse eradications to be similar to that used for rats with the following changes:

Bait applications use 50% overlap on both the first and second application (cf. for rats where 50% overlap is recommended for the first application and 25% for the second) (Fig. 1).

Bucket flow rates remain at or above 4 kg/ha (cf. for rats where bucket flow rates of 3 kg/ha are permissible). With 50% overlaps as in 1 above, this means applying a minimum of 8 kg/ha on the ground in each application.

The interval between applications is extended to a minimum of 14 days (cf. for rats where more flexibility in timing of the second application is permissible).

The IEAG has recently developed a best practice document incorporating these elements with other advice borrowed from the rat best practice (Broome, et al., 2017b). Since the Tasman Bay project, all subsequent mouse eradications following this advice have succeeded. including one of the largest (Macquarie 12,800 ha); multi-species eradications (Macquarie, and Rangitoto/Motutapu 3,809 ha) and a still-establishing mouse population (Maud 309 ha – see case study).

Changes 1 and 2 recognise the smaller territories of mice than rats and strive to ensure all mice encounter bait. Relatively few mouse home range studies have occurred on NZ islands (Ruscoe & Murphy, 2005). MacKay, et al. (2011) measured home ranges varying from 0.15–0.48 ha on Saddle Island. Radio-tracking found animals living in areas with dense shrub and grass cover had smaller ranges and mean nightly movements than those living in areas with tall canopy and minimal ground cover. Elsewhere on the NZ mainland in the absence of other mammalian predators and competitors, Goldwater, et al. (2012) estimated densities of 160 mice/ha in rank kikuyu grass (*Pennisetum clandestinum*) immediately after other mammals were eradicated, but density has since greatly declined.

Eradication designs must cater for not only the smallest home range (rather than the mean) but also the smallest foraging movements by mice over the limited period that bait is available in palatable condition. At 8 kg/ha the 2 g baits used in NZ would in theory be on the ground at 0.4 baits/m^2 providing ample opportunity for mice to encounter baits, especially after a second application.

Keeping bucket flow rates relatively high (possum control operations using the same equipment routinely use rates around 1 kg/ha), reduces the risk of interruptions in bait flow out of the bucket. Such interruptions in flow are potentially fatal to eradication success as they would not be mapped by the helicopter's GPS navigation recording system, and therefore could go unnoticed.

Change 3 acknowledges mice as light and erratic feeders compared to rats (Clapperton, 2006). Extending the period of bait availability, compared to a rat eradication, is desirable to ensure all mice have access to lethal doses before bait is consumed by other fauna or environmentally degraded. Brown (1993) found mice initially reluctant to take bait presented in bait stations on Allports and Motutapu Islands. They often 'sampled' small portions of baits over several nights before full-scale consumption ensued. He described a gradual spread of consumption from a focal point, speculating that social interactions between mice encouraged more to try the new food resource presented.

To counter the risk of mice being present but undetected in the presence of rats, some projects have deliberately designed their baiting strategy to mice eradication standard. For example, the rodent eradication (ship rats and kiore) on Great Mercury Island was designed to mouse eradication best practice standards despite no confirmed evidence of mice. The island operated as a pastoral farm with minimal biosecurity precautions for over 50 years so it was difficult to believe mice had not arrived during this time. The project sponsors found it cost effective risk management to assume mice were present and design the project accordingly (Corson & Hawkins, 2016).

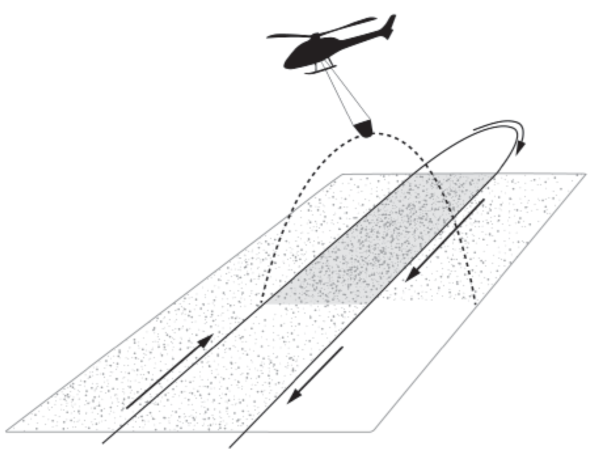

Fig. 1 50% overlap when aerially sowing bait. Arrows indicate centres and direction of two consecutive sowing lines. The dark shaded area shows the area of overlap between the first and the (half-completed) second line.

MOUSE BIOSECURITY

Keeping islands free of mice presents ongoing challenges in quarantine, surveillance and responding to arrivals. Pathways for invasion include cargo and personal luggage landed on the island, vessels and aircraft of all sizes, and swimming or rafting to islands.

Vulnerabilities to these pathways differ between islands but some islands may also be less susceptible to establishment of a mouse population following incursion. For example, Secretary, Kapiti, Stewart, Raoul and Campbell Islands have records of mice arriving, without evidence of meaningful action to respond, and yet failing to subsequently establish populations (DOC unpublished data). At the time all of these large islands had rats or stoats present, potentially providing a form of biological defence against mouse establishment. Weka (*Gallirallus australis*) may also play a role where they occur on islands. For example, on rat-free Tarakaipa Island mice were barely detectable in the presence of weka (DB pers. obs.). Weka held in captivity eagerly attacked mice entering their pen (CG pers. obs.). Conversely, the subsequent eradication of such predators could, in theory, increase the vulnerability of the island to invasion by mice. Further research into this phenomenon is warranted.

The probability of establishment can relate to propagule pressure (Lockwood, et al., 2005). Because rodent populations fluctuate seasonally in NZ with peaks in late summer, the risk of invasion could increase at this time of year. Additionally, mast seeding events in some forests can produce superabundant populations of mice which increase propagule pressure on nearby islands. For example, mice were successfully eradicated from Adele Island in 2007 and a biosecurity system installed. In the 2014/15 summer a significant mast seeding event occurred in the adjacent Abel Tasman National Park where mice became abundant. In February 2015 they were discovered on Adele. Attempts to eliminate them by localised trapping around points of detection failed and a population re-established (CG pers. obs.).

Mice as stowaways

Mice are inveterate stowaways with numerous records of their discovery in cargo destined for islands. The DOC invasion incidents database has 24 records of mice reaching islands amongst cargo between January 2010 and June 2017. Two more were intercepted on vessels en route to pest-free islands. Mice have been discovered in visitor day packs, in kayaks and nesting in under-seat dingy flotation. Container, building and vessel openings must be <6 mm to restrict mouse access. Of equal importance is the vigilance required to ensure doors, lids and hatches remain closed when not in use.

Quarantine measures to prevent mice reaching islands require constant vigilance by people involved. Careful checking of cargo, using rodent-proof containers for transport and control measures on board vessels are key components. These precautions can be enhanced by good rodent management and habitat control at ports and minimising the quantity of equipment transferred to islands (e.g. by having field equipment remain on-island).

Mice swimming to islands

Mice are often thought of as poor swimmers relative to rats (Russell & Clout, 2005). However, Evans, et al. (1978) found mice would readily enter water and swim. A fisherman saw a number of mice 600 m from shore in Lake Monowai while night fishing during the 2009 mouse plague (CG pers. comm..). Fishermen anecdotally report them in trout guts (James & Fox, 2017) and they have been found live in coastal flood debris (DB pers. obs.). The maximum distance over water that mice can cross unassisted remains unknown and therefore the pathway should not be assumed unimportant when considering biosecurity risks for an island.

Pomona and Rona Islands in Lake Manapouri were both assumed a 'safe' distance offshore (500 m and 600 m respectively) but both were reinvaded by mice within a decade of successful eradication, probably by swimming or rafting on flood debris. These re-invasions coincided with beech masting events when mice reached high abundance on the mainland.

Detection methods

We can readily detect mice at low densities, in the absence of other rodent species, using a range of tools including footprint tracking tunnels, chew cards and other bait interference methods, snap traps and trained detection dogs. Nathan, et al. (2013) studied mouse detection on Saddle Island (6 ha) during an experimental invasion event in which a male and a female mouse were released on the rodent-free island. They readily detected mice by both tracking tunnels and wax tags, even during the initial phases of the invasion.

Invading mice can move large distances. For example, pairs of mice sequentially released at opposite ends of Saddle Island (approximately 400 m apart), increased their nightly movements two-fold, and range sizes ten-fold, relative to movements on this island prior to the mouse eradication. This allowed them to rapidly and reliably encounter each member of the opposite sex (MacKay, 2011).

A mouse invading pest-free Moturua Island initially tracked inked footprint tracking cards in October 2011 and was finally trapped in late 2011. On one occasion this animal travelled at least 750 m between tracking tunnels over a 36-hour period (KB unpublished data).

Mice established on islands in relatively high numbers can hinder the detection of newly invading rats by 'swamping' detection tools. For example, they cover ink tracking cards on Waikawa Island within a few nights which can obscure the footprints of an invading rat. Mice usually do not trigger DOC200 stoat and rat traps but steal the bait, rendering the trap less attractive. These mouse-induced limitations delayed the detection of a Norway rat incursion on Waikawa Island in 2012, indicated by a dramatic decline in the critically endangered NZ shore plover. The rat was never caught and only retrospectively identified with the help of a rodent detection dog by the discovery of a nest containing bird remains and Norway rat fur and droppings (EM unpublished data).

Incursion response

Responding to the discovery of invading mice on a pest free island is challenging due to the potential delay between incursion and discovery through periodic surveillance checks. Nathan, et al. (2015) demonstrated the urgency of responding to a mouse invasion by experimentally releasing one male and one female mouse on Saddle Island. They subsequently bred and the mouse population reached the island's carrying capacity within five months. Routine surveillance discovered invading mice on Adele Island in February 2015, potentially months after arrival. Despite intensive trapping around points of detection the incipient population could not be eliminated.

CASE STUDY MAUD ISLAND

Biosecurity

Before 2013 rodents had never established on Maud Island (309 ha) in the Marlborough Sounds. Consequently, it has some highly rodent-vulnerable native species including some not found elsewhere, such as the Maud Island frog (*Leiopelma pakeka*), and others restricted to a handful of nearby pest-free islands.

Keeping pests from reaching Maud has long been a priority. Landing is restricted and DOC staff are present year-round. Stoats are considered the biggest invasive threat because they can swim the 900 m from the mainland and have done so on at least three occasions. Traps targeting stoats and rats are throughout the island and checked regularly. A quarantine store at the mainland DOC ranger station is used to check cargo destined for Maud or other pest free islands. Extra precautions are taken to prevent chytrid fungus – a pathogen implicated in the worldwide decline of frog populations (Berger, et al., 1999) – from reaching Maud.

In 2006, a mouse was killed by the Maud Island resident ranger when turning garden compost. An incursion response using mouse traps and a trained rodent detection dog failed to find further sign of mice after several weeks.

In October 2013, a mouse was captured in visitor accommodation on the island. An incursion response immediately deployed traps, detection devices and a rodent detection dog. Several mice were trapped around the buildings. The dog handler reported mice in several places across the island. Breeding was confirmed from necropsied animals. The youngest mice were in age class 1 (0–1 months in age) and the eldest in age class 6 (8–10 months) suggesting the first invaders arrived about a year previously and they had bred through the winter, which is uncommon in NZ.

DNA analyses found the Maud Island population highly inbred, suggesting the population arose from a single incursion. Although the mice were a genetic subset of the mainland population, their point of origin could not be established (E.M. & R. Fewster, unpublished data).

With an emerging picture of an established mouse population across the island, the incursion response team were forced to admit their efforts had begun too late and a whole island eradication was required.

To understand how mice had reached the island and remained undiscovered for long enough to establish, an independent review of biosecurity procedures was undertaken (Kennedy & Chappell, 2013). This found several weaknesses, including a lack of devices capable of killing or detecting mice on the island or on the ranger's boat, that was pulled onto a slipway on the island when not in use. The focus on stoats and rats allowed mice to go unnoticed. Some staff regularly visiting the island bypassed quarantine standards.

The review could not identify the pathway for the mouse incursion but made many recommendations for improvement which were actioned prior to the eradication. The island's biosecurity plan has recently been re-written to capture these new practices and give more authority to biosecurity rangers to enforce standards.

Eradication

In 2014, mouse eradication best practice was successfully applied to eradicating the newly established population of mice on Maud Island. Challenges included the abundance of natural food available to the expanding mouse population, and the presence of residential buildings requiring careful management of domestic foodstuffs and waste to minimise access to alternative food after toxic baiting.

A helicopter applied 8 kg/ha on 23 July 2014 followed by 8 kg/ha 23 days later (15 August) with strict adherence to the current agreed best practice described above. Two mice were trapped on Maud on 19 August, 27 days after the first bait application. Both had bait in their stomachs. A badly decayed male mouse was taken from a snap trap on 22 September and a female trapped the next day. This sexually mature female showed no signs of past or present breeding and appears to have survived about 60 and 37 days after the first and second bait applications, respectively. An intensive trapping grid (10 m × 10 m) was installed around each capture site covering about one hectare. No further mice were caught.

We estimated the age (from tooth eruption and wear) of the last mouse caught to be five months, meaning it could have lived through all bait applications. Bait was freely available from July to October, so these individuals must have encountered it. Although a range of trap baits were used, the snap traps which caught each mouse were baited with a Pestoff 20R pellet as used for the aerial baiting, indicating no aversion to the bait.

Testing of all four trapped mice revealed brodifacoum liver residues in three of them of 4.65–8.82 mg/kg. Considering liver values probably resulted from higher doses due to losses through excretion and metabolism (Eason & Wickstrom, 2001), these mice probably received many times the published LD50 for mice of 0.52 mg/kg (O'Connor & Booth, 2001). Maggots from the more decomposed male caught 22 September contained 2.35 mg/kg. DNA testing found these mice to be clearly from the original Maud invasion, not a new independent invasion.

Extensive monitoring over the subsequent two years no further survivors but a further incursion in 2018 has once again established a mouse population on the island. Mouse trapping on the island after bait application was intended as indicative monitoring only and had limited coverage of the island. We assume other mice survived in un-trapped areas long after bait application. These animals presumably acquired a lethal dose of brodifacoum and died without reproducing.

The successful eradication of an expanding population of mice from Maud is an indication of high bait acceptance despite other natural food being available in relative abundance. Camera footage from some of the buildings on Maud showed mice taking large quantities of bait placed in trays during the eradication and presumably caching it (CB pers. obs.).

CONCLUSION

Mice remain on many large islands in New Zealand and around the world. The techniques used in NZ to eradicate mice have been successful and could readily be applied to other temperate islands of similar size with a good chance of success. Biosecurity measures to protect islands from mouse invasion are challenging and mice must be considered a real threat to all rodent free islands, regardless of previous invasion history.

Biosecurity lessons:

Quarantine standards must apply to everyone to be effective. The pre-eminence of biosecurity over other duties of island staff and managers needs regular reinforcement to create an organisational culture which can sustain high biosecurity standards over time.

All potential threats and all potential pathways need to be assessed and multiple layers of protection established: i.e. quarantine checking, pest proof containerisation, hygiene of transportation, targeted surveillance, capability and readiness for incursion response.

Independent review of procedures can give valuable insights into opportunities for improvements and should be done proactively and routinely.

The risk of successful mouse invasions may be influenced by island predators (or lack thereof) and mouse abundance at potential source populations.

Eradication lessons:

The current agreed best practice used in NZ has a very good track record of success (>90% in known outcomes) against mice on temperate islands. This is far better than previously published review figures which did not present data on the quality of planning and delivery or discriminate between operations deliberately targeting mice and those targeting other species where mice also occur.

Mice can take a long time to succumb to the cumulative effects of small doses of brodifacoum and some individuals may require significantly higher doses than others. A baiting strategy which prolongs the availability of toxicant to mice has a better chance of success. In NZ this is usually achieved with two well-spaced bait applications but a third application is also an option.

Bait application rates need to allow for other bait consumers when multiple target species are involved and must not fall below the ability of sowing equipment to spread bait 100% reliably.

Where the presence of mice is likely but unproven due to suppression by other species, it is prudent to design the eradication assuming their presence, rather than discover that they have survived a rat eradication and thrived in the absence of rats or other predators.

Eradication is feasible against newly established and expanding populations of invading mice, especially if current agreed best practice is followed.

REFERENCES

Atkinson, I.A.E. (1973). 'Spread of the ship rat (*Rattus r. rattus* L.) in New Zealand'. *Journal of the Royal Society of New Zealand* 3: 457–472.

Angel, A. Wanless, R. and Cooper, J. (2009). 'Review of impacts of the introduced house mouse on islands in the Southern Ocean: are mice equivalent to rats?' *Biological Invasions* 11: 1743–1754.

Badan, D. (1986). 'Diet of the house mouse (*Mus musculus* L.) in two pine and a kauri forest'. *NZ Journal of Ecology* 9: 137–141.

Baker, C.F. (2006). 'Predation of inanga (*Galaxias maculatus*) eggs by field mice (*Mus musculus*)'. *Journal of the Royal Society of New Zealand* 36: 143–147.

Bancroft, B. (2004). *Patiti (Banded) Island rodent eradication 2004*. Rotorua, New Zealand: Waimangu Volcanic Valley Ltd and Department of Conservation.

Berger, L. Speare, R. and Hyatt A. (1999). 'Chytrid Fungi and Amphibian Declines: Overview, Implications and Future Directions'. In: Campbell, A. (ed.) *Declines and Disappearances of Australian Frogs*, pp. 23–33. Canberra, ACT: Environment Australia.

Bowie, M. Kavermann M. and Ross, J. (2011). 'The Quail Island Story – Thirteen Years of Multi-species Pest Control: Successes, Failures and Lessons Learnt'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 157–161. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Bridgman, L.J. (2012). 'Interactions between ship rats and house mice'. PhD thesis. Hamilton, New Zealand: University of Waikato. Retrieved from <http://hdl.handle.net/10289/6396>.

Broome, K. (2009). 'Beyond Kapiti – a decade of invasive rodent eradications from New Zealand islands'. *Biodiversity* 10: 14–24.

Broome, K., Golding, C., Brown, K., Corson, P. and Bell, P. (2017a). *Rat Eradication Using Aerial Baiting: Current Agreed Best Practice Used in New Zealand (Version 3.1).* Wellington, NZ: New Zealand Department of Conservation internal document DOC-29396.

Broome, K., Golding, C., Brown, K., Horn, S., Corson, P. and Bell, P. (2017b). *Mouse Eradication Using Aerial Baiting: Current Agreed Best Practice Used in New Zealand (Version 1.0).* Wellington, NZ: New Zealand Department of Conservation internal document DOC-3034281.

Brown, D. (1993). 'Eradication of mice from Allports and Motutapu Islands'. *Ecological Management* 1: 19–30. Wellington, NZ: Department of Conservation.

Brown, D. (2013). *Coal Island (Te Puka-Hereka) Restoration Plan. Department of Conservation Te Anau.* <http://www.doc.govt.nz/Documents/conservation/land-and-freshwater/offshore-islands/coal-island-restoration-plan.pdf>. DOCDM-12333366.

Campos, A.R. and Granadeiro, J.P. (1999). 'Breeding biology of the white-faced storm-petrel on Selvagem Grande Island, north-east Atlantic'. *Waterbirds: The International Journal of Waterbird Biology* 22: 199–206.

Chappell, R. (2008). *Eradication of Ship Rat, Kiore, House Mouse and Rabbit from Ohinau and Motutapere Islands – Final Report*. Department of Conservation internal report DOC-5449724.

Clapperton, B.K. (2006). 'A Review of the Current Knowledge of Rodent Behaviour in Relation to Control Devices'. *Science for Conservation* 263. Wellington, NZ: Department of Conservation. 55 p.

Clout, M.N. and Russell, J.C. (2006). 'The Eradication of Mammals from New Zealand Islands'. In: F. Koike, M.N. Clout, M. Kawamichi, M. De Poorter and K. Iwatsuki (eds.) *Assessment and Control of Biological Invasion Risks*, pp.127–141. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Corson, P. and Hawkins, K. (2016). *Post Operational Report for the Eradication of Kiore, Ship Rats and Cats from Ahuahu – Great Mercury Island, New Zealand*. Department of Conservation internal report DOCDM-1477863.

Cuthbert, R., Visser, P, Louw, H. and Ryan, P. (2011). 'Palatability and efficacy of rodent baits for eradicating house mice (*Mus musculus*) from Gough Island, Tristan da Cunha'. *Wildlife Research* 38(3): 196–203.

Department of Conservation (2011). *Indian Island Rodent Eradication*. Department of Conservation internal report DOCDM-752616.

Eason C and Wickstrom, M. (2001). *Vertebrate Pesticide Toxicology Manual (Poisons)*. Wellington, NZ: Department of Conservation.

Elliott, G. and Kemp, J. (2016). 'Large scale pest control in New Zealand beech forests'. *Ecological Management and Restoration* 17(3): 200–209.

Evans, R.L., Katz, E.M., Olson, N.L. and Dewsbury, D.A. (1978). 'A comparative study of swimming behaviour in eight species of muroid rodents'. *Bulletin of the Psychonomic Society* 11: 168–170.

Fraser, Island, Wilcken, J. and Buley, K. (2013). *Rotoroa Wildlife Management and Translocation Plan 2013-2038. Version 1.4.* Auckland, NZ: Auckland Zoo.

Frogley, K. (2013). 'Invasive Mice and their Effects on Birds: Do Mice (*Mus musculus*) Within Fenced Sanctuaries Prey on Bird Eggs? *University of Otago Wildlife Management Report* 270. 30 p.

Glassey, M. (2006). 'A Historical and Ecological Survey of Hauturu Island, Whangamata'. BAppSc thesis. Auckland University of Technology / Bay of Plenty Polytechnic, Tauranga.

Golding, C. (2010). 'House mouse (*Mus musculus*) eradication by aerial bait application on Adele, Tonga and Fisherman Islands, Abel Tasman National Park, New Zealand'. *Conservation Evidence* 7: 62–68.

Goldwater, N., Perry, G.L.W. and Clout, M.N. (2012). 'Responses of house mice to the removal mammalian predators and competitors'. *Austral Ecology* 37: 971–979.

Griffiths, R., Buchanan, F., Broome, K., Neilsen, J., Brown, D., Weakley, M. (2015). 'Successful eradication of invasive vertebrates on Rangitoto and Motutapu Islands'. *Biological Invasions* 17: 1355–1369.

Hickford, M.J., Cagnon, M. and Schiel, D.R. (2010). 'Predation, vegetation and habitat-specific survival of terrestrial eggs of a diadromous fish, *Galaxias maculatus* (Jenyns, 1842)'. *Journal of Experimental Marine Biology and Ecology* 385: 66–72.

Hook, T. and Todd, P. (1992). 'Mouse Eradication on Mana Island'. In: C.R. Veitch, M. Fitzgerald, J. Innes and E. Murphy (eds.) *National Predator Management Workshop*, pp. 33. *Threatened Species Occasional Publication* 3, Wellington, NZ: Department of Conservation.

Horn, S. Russell J. and Simmons, B. (these proceedings). Antipodes Island mouse eradication.

Howald, G., Donlan, C.J., Galvan, J.B., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradications on islands'. *Conservation Biology* 21: 1258–1268.

Imber, M.J., Bell, B.D. and Bell E.A. (2005) 'Antipodes Islands birds in autumn 2001'. *Notornis* 52:125–132.

James, G. and Fox, S. (2017). 'Mouse Eating Trout'. <http://www.bestofnzflyfishing.com/articles/mouseeatingtrout.html>. Accessed June 2017.

Kennedy, E. and Chappell, R. (2013). Audit of DOC's *Quarantine and Surveillance Arrangements for the Islands of the Marlborough Sounds. November 2013*. Christchurch, NZ: Department of Conservation internal report DOC-1318761.

King, C., Alexander A., Chubb T., Cursons R., MacKay J., McCormick H., Murphy E., Veale A. and Zhang H. (2016). 'What can the geographic distribution of mtDNA haplotypes tell us about the invasion of New Zealand by house mice *Mus musculus*?' *Biological Invasions* 18 (6):1551–1565.

King, C. and Murphy, E. (2005). 'Stoat'. In: C. King (ed.) *The Handbook of New Zealand mammals,* pp. 261–287. Oxford: Oxford University Press.

Lockwood, J.L., Cassey, P. and Blackburn, T. (2005). 'The role of propagule pressure in explaining species invasions'. *Trends in Ecology and Evolution* 20(5): 223–228.

MacKay, J.W.B., Russell, J.C. and Murphy, E.C. (2007). 'Eradicating house mice from islands: successes, failures and the way forward'. In: G.W. Witmer, W.C. Pitt and K.A. Fagerstone (eds.) *Managing Vertebrate Invasive Species: Proceedings of an International Symposium*, pp. 294–304. Fort Collins, Colorado, USA: USDA/APHIS Wildlife Services, National Wildlife Research Center.

MacKay, J.W.B. (2011). 'Improving the Success of Mouse Eradication Attempts on Islands'. PhD thesis, Auckland, NZ: University of Auckland.

MacKay, J.W.B., Murphy, E.C., Anderson, S.H., Russell, J.C., Hauber, M.E., Wilson, D.J. and Clout, M.N. (2011). 'A Successful Mouse Eradication Explained by Site-specific Population Data'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 198–203. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

McKinlay, B. (1999). 'Eradication of Mice from Mou Waho, Lake Wanaka'. *Ecological Management 7*: 1–5. Wellington, NZ: Department of Conservation.

Miller, A.P. and Webb, P.I. (2001). 'Diet of house mice (*Mus musculus* L.) on coastal sand dunes, Otago, New Zealand'. *New Zealand Journal of Zoology* 28: 49–55.

Miskelly, C.M., Walker, K.J. and Elliott, G.P. (2006). 'Breeding ecology of three subantarctic snipes'. *Notornis* 53: 361–374.

Nathan, H.W., Clout, M.N., Murphy, E.C. and MacKay, J.W.B. (2013). 'Strategies for detection of house mice (*Mus musculus*) on a recently invaded island'. *New Zealand Journal of Ecology* 37(1): 26–32.

Nathan, H.W., Clout, M.N., MacKay, J.W.B., Murphy, E.C. and Russell, J.C. (2015). 'Experimental island invasion of house mice (*Mus musculus*)'. *Population Ecology* 57(2): 363–371.

Newman, D.G. (1985). *The Apparent Loss of the Whenuakura Island Tuatara Population, Whangamata Islands Wildlife Sanctuary*. Wellington, NZ: Department of Conservation internal report, file WIL 35/2/13.

Newman, D.G. (1994). 'Effects of a mouse, *Mus musculus*, eradication programme and habitat change on lizard populations of Mana Island, New Zealand, with special reference to McGregor's skink, *Cyclodina macgregori'*. *New Zealand Journal of Zoology* 21: 443–456.

Norbury, G., van den Munckhof, M., Neitzel, S., Hutcheon, A., Reardon, J. and Ludwig, K. (2014). 'Impacts of invasive house mice on post-release survival of translocated lizards'. *New Zealand Journal of Ecology* 38: 322–327.

O'Connor, C.E. and Booth, L.H. (2001). 'Palatability of rodent baits to wild house mice'. *Science for Conservation* 184. Wellington, NZ: Department of Conservation. 11 p.

Owen, K. (1998). 'Removal and Reintroduction of North Island Weka (*Gallirallus australis greyi*) to Mokoia Island as a Result of a Talon 7/20 Cereal Based Aerial Poison Drop'. *Ecological Management 6*: 41–47. Wellington, NZ: Department of Conservation.

Ritchie, J. (2000). *Matakohe – Limestone Island Scenic Reserve Restoration Plan*. Auckland, NZ: Natural Logic Environmental Management Ltd.

Romijn R.L. (2013). 'Can Skinks Recover in the Presence of Mice? BSc (Hons) thesis. Wellington, NZ: Victoria University of Wellington.

Rowe, F.P. and Bradfield, A. (1976). 'Trials of the anticoagulant rodenticide WBA 8119 against confined colonies of warfarin-resistant house mice (*Mus musculus*)'. *Journal of Hygiene* 77: 427–431.

Ruscoe, W.A. and Murphy, E.C. (2005). 'The House Mouse'. In: C.M. King (ed.) *The Handbook of New Zealand Mammals.* Oxford: Oxford University Press.

Russell, J.C., and Clout, M.N. (2005). 'Rodent Incursions on New Zealand Islands'. In: J. Parkes, M. Statham and G. Edwards (eds.) *Proceedings of the 13th Australian Vertebrate Pest Conference*, pp. 324–330. Lincoln, New Zealand: Landcare Research.

Russell, J.C. and Broome, K.G. (2016) 'Fifty years of rodent eradications in New Zealand: another decade of advances'. *New Zealand Journal of Ecology* 40: 197–204.

Shaw, V., Torr, N. (2011). 'Eradicating mammal pests from Pomona and Rona Islands in Lake Manapouri, New Zealand: a focus on rodents'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 356–360. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

St Clair, J.J.H. (2011). 'The impacts of invasive rodents on island invertebrates.' *Biological Conservation* 144: 68–81.

Torr, N. (2002). 'Eradication of rabbits and mice from Subantarctic Enderby and Rose Islands'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 319–328. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group, IUCN, Gland, Switzerland and Cambridge, UK.

Towns D.R. and Broome, K.G. (2003). 'From small Maria to massive Campbell: forty years of rat eradications from New Zealand islands'. *NZ Journal of Zoology* 30: 377–398.

Veale, A.J., Russell, J.C., King, C.M. (2018). The genomic ancestry, landscape genetics, and invasion history of introduced mice in New Zealand. *Royal Society Open Science* 5: 170879. <http://dx.doi.org/10.1098/rsos.170879>.

Veitch, C.R. and Bell, B.D. (1990). 'Eradication of Introduced Animals from the Islands of New Zealand'. In: D.R. Towns, C.H. Daugherty and I.A.E. Atkinson (eds.) *Ecological Restoration of New Zealand Islands*, pp. 137–146. Wellington, NZ: Department of Conservation.

Veitch, C.R. (2002a). 'Eradication of Norway rats (*Rattus norvegicus*) and house mice *(Mus musculus*) from Browns Island, New Zealand'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 350–352. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group, IUCN, Gland, Switzerland and Cambridge, UK.

Veitch, C.R. (2002b). 'Eradication of Norway rats (*Rattus norvegicus*) and house mice *(Mus musculus*) from Motuihe Island, New Zealand'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 353–356. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group, IUCN, Gland, Switzerland and Cambridge, UK.

Watts, C., Innes, J., Wilson, D., Fitzgerald, N., Bartlam, S., Thornburrow, D., Smale, M. and Barker, G. (2017). *Impacts of Mice Alone on Biodiversity: Final Report of a Waikato Field Trial*. Landcare Research Contract Report LC2747. Prepared for Waikato Regional Council. 33 p.

Wedding, C. (2007). 'Aspects of the Impacts of Mouse (*Mus musculus*) Control on Skinks in Auckland, New Zealand'. MSc thesis. Auckland, NZ: Massey University.

Williams, P.A., Karl, B.J., Bannister, P. and Lee, W.G. (2000). 'Small mammals as potential seed dispersers in New Zealand'. *Austral Ecology* 25: 523–532.

Wilson, D.J., Wright E.F., Canham C.D. and Ruscoe W.A. (2007). 'Neighbourhood analyses of tree seed predation by introduced rodents in a New Zealand temperate rainforest'. *Ecography* 30: 105–119.

Appendix 1 Mouse eradications on NZ islands >1ha. F or R = failed or reinvaded and S (R) = successful but subsequently reinvaded.

Region	Mice targeted	Island	Area (ha)	DOC best practice	Eradication start date	Eradication status	Primary baiting method	Secondary baiting method	References
Coromandel	Y	Whenuakura	2	NA	1983	Successful	Bait station	NA	Veitch & Bell, 1990; Newman, 1985
Cook Strait	Y	Mana	217	NA	1989	Successful	Bait station	Aerial broadcast	Hook & Todd, 1992; Newman, 1994
Marlborough	Y	Allports	16	NA	1989	Successful	Bait station	NA	Brown, 1993
Marlborough	Y	Motutapu	2	NA	1989	Successful	Bait station	NA	Brown, 1993
Lake Rotorua	N	Mokoia	136	NA	1989	Failed	Bait station	NA	MacKay, et al., 2007
Kaituna Bay	Y	Rimariki	22	NA	1989	Successful	Bait station	NA	Veitch & Bell, 1990
Kaipara Harbour	Y	Moturemu	5	NA	1992	Successful	Bait station	NA	Clout & Russell, 2006
Subantarctic	N	Enderby	710	NA	1993	Successful	Aerial broadcast	NA	Torr, 2002
Coromandel	N	Hauturu	10	NA	1993	F or R	Bait station	Hand broadcast	Glassey, 2006
Hauraki Gulf	Y	Te Haupa	6	NA	1993	Failed	Bait station	NA	Clout & Russell, 2006
Coromandel	Y	Motutapere	46	NA	1994	Successful	Aerial broadcast	Bait station	Clout & Russell, 2006

Appendix 1 (continued) Mouse eradications on NZ islands >1ha. F or R = failed or reinvaded and S (R) = successful but subsequently reinvaded.

Region	Mice targeted	Island	Area (ha)	DOC best practice	Eradication start date	Eradication status	Primary baiting method	Secondary baiting method	References
Hauraki Gulf	Y	Browns	60	NA	1995	Successful	Aerial broadcast	NA	Veitch, 2002a
Lake Wanaka	Y	Mou Waho	140	NA	1996	Successful	Aerial broadcast	NA	McKinlay, 1999
Whangarei Harbour	Y	Matakohe	37	NA	1996	F or R	Aerial broadcast	NA	MacKay, et al., 2007
Lake Rotorua	Y	Mokoia	136	NA	1996	F or R	Aerial broadcast	Hand broadcast	Clout & Russell, 2006; Owen, 1998
Hauraki Gulf	Y	Motuihe	179	NA	1997	Successful	Aerial broadcast	NA	Veitch, 2002b
Whangarei Harbour	Y	Matakohe	37	NA	1997	F or R	Aerial broadcast	NA	MacKay, et al., 2007; Ritchie, 2000
Whangarei Harbour	Y	Matakohe	37	NA	1998	F or R	Aerial broadcast	NA	Clout & Russell, 2006; Ritchie, 2000
Lake Rotorua	Y	Mokoia	136	NA	2001	S (R)	Aerial broadcast	Hand broadcast	MacKay, et al., 2007
Whangarei Harbour	Y	Matakohe	37	NA	2001	F or R	Bait station	NA	Clout & Russell, 2006
Canterbury	N	Quail	85	NA	2002	Failed	Bait station	Hand broadcast	Bowie, et al., 2011
Lake Rotomahana	Y	Patiti	13	NA	2004	Failed	Bait station	NA	Bancroft, 2004
Marlborough	Y	Blumine	377	NA	2005	Successful	Aerial broadcast	NA	MacKay, et al., 2007
Coromandel	Y	Ohinau	46	NA	2005	Successful	Aerial broadcast	NA	Chappell, 2008
Marlborough	Y	Pickersgill	96	NA	2005	Successful	Aerial broadcast	NA	MacKay, et al., 2007
Lake Manapouri	Y	Rona	60	Y	2007	S (R)	Aerial broadcast	NA	Shaw & Torr, 2011
Lake Manapouri	Y	Pomona	262	Y	2007	S (R)	Aerial broadcast	NA	Shaw & Torr, 2011
Tasman Bay	Y	Adele	88	Y	2007	S (R)	Aerial broadcast	NA	Golding, 2010
Tasman Bay	Y	Fisherman	4	Y	2007	S (R)	Aerial broadcast	NA	Golding, 2010
Tasman Bay	Y	Tonga	8	Y	2007	S (R)	Aerial broadcast	NA	Golding, 2010
Hauraki Gulf	Y	Te Haupa	6	N	2008	Successful	Bait station	Hand broadcast	MacKay, et al., 2011
Fiordland	Y	Coal	1,163	Y	2008	Successful	Aerial broadcast	NA	Brown, 2013
Hauraki Gulf	Y	Motutapu	1,509	Y	2009	Successful	Aerial broadcast	Bait station	Griffiths, et al., 2015
Hauraki Gulf	Y	Rangitoto	2,311	Y	2009	Successful	Aerial broadcast	Bait station	Griffiths, et al., 2015
Canterbury	Y	Quail	85	Y	2009	F or R	Aerial broadcast	Hand broadcast	Bowie, et al., 2011
Hauraki Gulf	Y	Te Haupa	6	N	2010	Successful	Trapping	Bait station	MacKay, et al., 2011
Dusky Sound	Y	Indian	167	Y	2010	Successful	Aerial broadcast	NA	Department of Conservation, 2011
Hauraki Gulf	Y	Rotoroa	140	Y	2013	Successful	Aerial broadcast	NA	Fraser, et al., 2013
Marlborough	Y	Maud	309	Y	2014	Successful	Aerial broadcast	NA	This paper
Subantarctic	Y	Antipodes	2,012	Y	2016	Successful	Aerial broadcast	NA	Horn, et al. (these proceedings)

Simultaneous rat, mouse and rabbit eradication on Bense and Little Bense Islands, Falkland Islands

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Abstract Bense and Little Bense Islands (144 ha total area) have, for over a century, supported populations of three introduced pest mammals: Norway rat (*Rattus norvegicus*), house mouse (*Mus musculus*), and European rabbit (*Oryctolagus cuniculus*). An operation to eradicate these mammals simultaneously was undertaken in winter 2016. Cereal pellets laced with brodifacoum (25 ppm) were hand-broadcast on both islands in two applications with 3,900 kg of bait applied in total. Baiting transects were spaced at 20 m intervals and bait-throwing positions located every 20 m along each transect. The coastline was also baited at 20 m intervals. Precision bait coverage was aided by programming GPS units to give off an audible alarm when staff reached each correct bait-throwing position. Application 1 resulted in an average bait density of 15.3 kg/ha. Application 2 commenced 10 days later and resulted in an average baiting density of 11.7 kg/ha. Reduced availability of field staff resulted in coverage in Application 2 being less complete than in Application 1 and only the most important mammal habitats were baited a second time. These were: all tussock areas, all coastlines, and some inland heath areas. Areas with no vegetation (e.g. burned zone on Bense) and some inland heath communities were not treated, although all of these retained unconsumed bait from Application 1. Some non-target mortality was recorded, with dolphin gulls (*Larus scoresbii*) being the most common victims. This was also the only species observed to consume bait pellets. Consumption of poisoned mammals or gulls may have killed three turkey vultures (*Cathartes aura jota*), one striated caracara (*Phalcoboenus australis*), and one short-eared owl (*Asio flammeus*). The removal of invasive species is part of a broader ecological restoration plan for these islands and will hopefully lead to an increase in native biodiversity, including the re-establishment of the endemic passerines Cobb's wren (*Troglodytes cobbi*) and blackish cinclodes (*Cinclodes antarcticus*).

Keywords: ecological restoration, invasive species

INTRODUCTION

Like the natural biodiversity on most islands, the native plants and animals of the Falkland Islands are vulnerable to catastrophic impacts when non-native mammals are introduced (Tabak, et al., 2014; Carey, 2015). Prior to the arrival of humans, the Falklands had only one species of terrestrial mammal – the Falklands fox, or warrah (*Dusicyon antarcticus*). While people quickly hunted this sole native mammal to extinction by 1876, they also introduced a further nine alien species which have since established feral populations. These are: Norway rat (*Rattus norvegicus*), black rat (*R. rattus*), house mouse (*Mus musculus*), European rabbit (*Oryctolagus cuniculus*), eastern cottontail rabbit (*Sylvilagus* sp.), brown hare (*Lepus capensis*), Patagonian grey fox (*Dusicyon griseus*), domestic cat (*Felis catus*), and guanaco (*Lama guanicoe*), as well as domestic dogs (*Canis lupus*), poultry, and livestock (Strange, 1992; Woods & Woods, 2006). These invasive species have had negative impacts on the native birds (Tabak, et al., 2015) and invertebrates (St Clair, 2011) through direct predation and competition for food.

The Falkland Islands are located in the south-west Atlantic Ocean, approximately 500 km east of Argentina. Spanning 51°–53° S and 57°–62° W, there are 778 islands in the archipelago (FITB, 2016). Eleven islands are permanently inhabited, although only the two largest of these are home to more than one family. The Falklands are unique among subantarctic islands in that much of the land is privately owned, and conservation-minded landowners have been at the forefront of environmental work in the islands (for example Strange, 2007; Poncet, et al., 2011). Invasive species eradications began in 2001 with the removal of Norway rats from two small islands (Brown, et al., 2001). Rats have since been successfully cleared from a further 66 islands, while the Patagonian grey fox was eradicated from one island in 2008 (Poncet, et al., 2011; FIG, 2015). The project covered here is the first Falklands attempt to eradicate mice and rabbits, and the first to attempt the simultaneous removal of three species: *R. norvegicus, M. musculus,* and *O. cuniculus*. These were the only introduced mammal species on the Bense islands.

Although conservation gains can be made by eradicating a single mammal species where more than one invasive species is present (Helmstedt, et al., 2016), eradication attempts that simultaneously target all invasive species are desirable when logistically and financially feasible. Simultaneous multi-species eradications can avoid magnifying the problems caused by one pest species when another is removed. On subantarctic Macquarie Island, the removal of cats prior to the eradication of rabbits may have contributed to a population increase of the latter species, which in turn exacerbated grazing pressure on plants and soil erosion (Bergstrom, et al., 2009; but see Springer (2016) for a discussion of the role of rabbit population fluctuations).

METHODS

Site description

Bense (c. 107 ha) and Little Bense (c. 37 ha) Islands are found in Port North, in the north-west Falkland Islands at 51°29'S 61° 31'W. These two islands have been home to Norway rats, house mice, and European rabbits for more than 100 years. Rabbits were deliberately introduced by whalers whereas rats and mice arrived as stowaways on vessels anchored in the nearby harbour or used to move livestock (R. Napier pers. comm.). The islands are joined by a rocky reef, exposed at low tide, and therefore were treated as a single island for eradication purposes (Fig. 1).

The vegetation is broadly similar across the two islands, with at least 20 species of vascular plants recorded (Table 1). The coastal zone is maritime tussock formation, with lush stands of tussock grass (*Poa flabellata*) growing to 3 m in height. The interior is low-growing oceanic heath

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Fig. 1 Bense Island (bottom left), Little Bense Island (top) and West Falkland Island (bottom right). Note the dark, burned area along much of the east coast of Bense. At low tide, Bense and Little Bense Islands are connected by a rocky reef. Bense is 750 m away from West Falkland at its closest point.

formation, dominated by diddle-dee (*Empetrum rubrum*) (Moore, 1968). Bense Island has had greater grazing pressure with horses, cattle, and sheep wintering on the island at various times during the 20th century. These same species were also placed on Little Bense but would quickly migrate to Bense Island as Little Bense has no water on it. (W. Goodwin, pers. comm.) This may explain why palatable species such as boxwood (*Hebe elliptica*) are more prevalent on Little Bense, and why there are also greater expanses of dense tussock on the smaller island. Both islands have been free of livestock since 1985. Also in 1985, a fire burned about 20% of Bense Island. The scorched area remains an unvegetated barren zone of peat and ash, with loose peat creeping downwind and smothering some areas of unburned vegetation.

The western coast of Bense Island has vertical cliffs up to c. 25 m in height. The terrain gradually tilts lower as one moves east, with gentle cobble or sand beaches found on the east coast. Little Bense is lower (c. 18 m maximum height) with a coastline of sloping rocks in the west and north, and sand beaches in the east and south. At its closest point, the mainland of West Falkland Island is 750 m away from Bense Island.

Despite the presence of invasive mammals, the avifauna of these islands is not completely extirpated (Table 1)

Table 1 Plants and birds commonly found on Bense and Little Bense Islands.

Birds		**Plants**	
Magellanic penguin	*Spheniscus magellanicus*	Tussock grass	*Poa flabellata*
Rock shag	*Plalacrocorax magellanicus*	Couch grass	*Agropyron pubiflorum* (*magellanicum*)
Imperial shag	*Phalacrocorax atriceps albiventer*	Common bent grass	*Agrostis tenuis*
Black-crowned night heron	*Nycticorax nycticorax falklandicus*	Hair grass	*Aira* sp.
Upland goose	*Chloephaga picta*	Small fern	*Blechnum penna-marina*
Kelp goose	*Chloephaga hybrida*	Chickweed	*Cerastium arvense*
Ruddy-headed goose	*Chloephaga rubidiceps*	Wavy hair grass	*Deschampsia flexuosa*
Falklands steamer duck	*Tachyeres brachypterus*	Diddle-dee	*Empetrem rubrum*
Crested duck	*Lophonetta specularioides*	Tufted fescue grass	*Festuca cirrosa (erecta)*
Turkey vulture	*Cathartes aura jota*	Cudweed	*Gamochaeta nivalis*
Variable hawk	*Geranoaetus polyosoma*	Pig vine	*Gunnera magellanica*
Striated caracara	*Phalcoboenus australis*	Native boxwood	*Hebe elliptica*
Magellanic oystercatcher	*Haematopus leucopodus*	Mountain berry	*Pernettya pumila*
Blackish oystercatcher	*Haematopus ater*	Meadow grass	*Poa* sp.
Two-banded plover	*Charadrius falklandicus*	Sheep's sorrel	*Rumex acetosella*
Magellanic snipe	*Gallinago paraguaiae magellanica*	Sea cabbage	*Senecio candicans*
Brown skua	*Catharacta antarctica*	Procumbent pearlwort	*Sagina procumbens*
Dolphin gull	*Larus scoresbii*	Groundsel	*Senecio vulgaris*
Kelp gull	*Larus dominicanus*	Christmas bush	*Baccharis magellanica*
South American tern	*Sterna hirundinacea*	Wood rush	*Luzula alopecurus*
Dark-faced ground tyrant	*Muscisaxicola maclovianus*		
Grass wren	*Cistothorus platensis*		
Falklands thrush	*Turdus falcklandii*		
White-bridled finch	*Melanodera melanodera*		
Long-tailed meadowlark	*Leistes loyca*		
Black-chinned siskin	*Spinus barbatus*		

and the islands were listed within the Falklands as a top priority for mammal eradication (Miller, 2008). While 26 land and sea bird species were commonly found on the islands, conspicuously absent were the only Falklands endemic passerines: Cobb's wren (*Troglodytes cobbi*) and blackish cinclodes (*Cinclodes antarcticus*). Neither of these species breeds on islands with rats (Tabak, et al., 2016). The islands are also bereft of burrowing seabirds such as sooty shearwater (*Puffinus griseus*) and thin-billed prion (*Pachyptila belcheri*), both of which breed on nearby rat-free islands (Woods & Woods, 1997).

Bense and Little Bense have never had a resident human population, but because they were a desirable site for wintering livestock, for much of the 20th century they were occasionally home to shepherds and farmhands for a few days at a time. A small shanty, built on Bense in 1926, was the only building found on either island until 2002, when a second shanty was built next to the original structure. All farming ceased in 1996, when Bense and Little Bense Islands (along with neighbouring Cliff Island and Bradley Islet) were purchased by the SubAntarctic Foundation for Ecosystems Research (SAFER) with a goal to restore the islands' ecology and improve them as wildlife habitat.

Index trapping

Index trapping to ascertain habitat preferences and relative abundance of rodents was conducted on Bense Island over eight visits, spanning 10 years and most seasons (i.e. November 2004, October 2006, July 2007, August 2007, September 2008, March 2010, January 2013, January 2014). Trap lines followed the methods described in Cunningham and Moors (1996), using Victor Easy Set wooden snap-traps (Woodstream Corp., Lititz, Pennsylvania, USA), with an interval of 25 m between trapping stations. A trap which caught an animal or which was sprung with no catch, was deemed to have been effective for half the night, and was therefore counted as 0.5 of an effective trap-night. Trap lines were placed in two different habitats: coastal tussock formation (1,612.5 effective trap-nights), or inland heath communities (1,077 effective trap-nights).

Eradication operation

Following basic ecological studies, including surveys of birds and invertebrates, an operation to eradicate rats, mice, and rabbits was undertaken in winter 2016. For bait distribution, local field staff were hired in Stanley, the Falklands capital. None had previous experience with hand-baiting so training was provided the day prior to the beginning of operations. The operation ran from 8 August to 3 September and was timed to coincide with the period when natural food on the islands is most scarce. Cereal pellets laced with brodifacoum at 25 ppm (25-W Conservation Pellets, manufactured by Bell Laboratories) were hand-broadcast along parallel transects in two applications, with an interval of 10 days between them.

A baiting map of the islands, comprising a series of parallel transects spaced at 20 m intervals laid over a high-resolution satellite photo, was created using QGIS software. Along each of these transects, baiting points were located every 20 m (Fig. 2). This resulted in an imaginary grid with 20m squares across both islands. Baiting points were also created at 20 m intervals along the coastlines of both islands, following the natural contours of the shoreline. Map data were loaded onto handheld GPS units (Garmin GPSMAP64) with an audible alarm set to sound whenever the unit reached a baiting point. Field personnel could then navigate to a desired transect line and follow it exactly, with the alarm telling them when they had reached a baiting point. GPS units were accurate to around 2 m.

Fig. 2 Detail of the baiting map of Bense Island. Each white or purple dot represents a baiting point. Baiting points are 20 m apart.

The walking tracks of field staff were monitored using GPS tracking and were checked each night against a base map. Any areas not covered properly were thus identified, and targeted for remedial attention the following day.

At each baiting point, five full scoops of bait were flung in five different directions as per hand broadcast best-practice (Broome, et al., 2011). Thus, coverage at each baiting point overlapped with bait thrown from neighbouring baiting points. Bait pellets were thrown with plastic scoops cut to hold 100 g when full. Staff carried the pellets in 20-litre plastic buckets, which could hold about 15 kg of bait. Rubber gloves, Tyvek coveralls, and dust masks were available to all field personnel.

Bait was transported to the islands from Stanley. It first went by barge to a protected bay on West Falkland Island, and from there it was moved to Bense and Little Bense in loads slung under a Chinook helicopter. The helicopter deposited the bait in six depots across the approximate midline of Bense Island, and at one location in the centre of Little Bense Island. A total of 4,400 kg of bait was delivered to the islands for this operation.

At the end of the operation, seven bait stations were established along the north-eastern coast of Bense Island, in areas thought to be the most likely zone of landfall for any rats that might swim from West Falkland Island. Bait was placed inside lengths of polyethylene pipe, 15 cm in diameter. Wax baits (containing 0.0005% w/w difenacoum and 0.001% w/w denatonium benzoate) were wired to the inside of the pipe and a handful of brodifacoum cereal pellets were also added. Bait stations were secured to the ground with wire staples and rocks.

Post-eradication monitoring

The islands were re-visited briefly in December 2016 (three months post-baiting) and in November 2017 (14 months post-baiting) to search for survivors of the baiting operation. During the latter visit, two hundred chewsticks (PCR Wax Tag, Pest Control Research) with peanut butter-flavoured wax attractants were installed in all coastal areas and in vegetated interior zones, with preference given to those areas known to be good rodent habitat. Chewsticks were checked for bite marks from rabbits and rodents before departure (up to 14 days after installation) and were left in place to be checked on subsequent visits to the islands. Staff actively searched for tracks, fresh droppings, and other signs of mammals throughout the visit. Daytime searches for rabbits were made by a dedicated hunter, including extensive observations by binoculars from a camouflaged position on high ground and by careful downwind stalking through areas known to be favoured by rabbits. A thermal camera (Thermapp) was used to replicate these searches at night without the use of lights that could frighten rabbits.

Weather during eradication operation

Temperatures ranged from -3° C to +7° C, with moderate to strong winds on all days. Snow and sleet showers frequently swept the islands but accumulation was slight and short-lived. No precipitation fell as rain. Weather did not prevent baiting except for one half-day during Application 1 and one full-day during Application 2, when wind speeds were too high to cast bait effectively.

RESULTS

Index trapping

Index trapping showed rats were much more prevalent in coastal tussock areas, with 82 rats caught there from 1,612.5 effective trap nights, whereas on inland heath areas, only three rats were recorded from 1,077 effective trap nights. Mice were more evenly distributed between the two habitats sampled, with 50 caught in coastal tussock, and 32 caught in inland heath.

Rabbits were not targeted with snap traps but individuals were observed on most parts of Bense, except the denuded burn-zone. Rabbits were not thought to be present on Little Bense until a single animal was observed there in February 2015. This was the only time in 18 visits that a rabbit was seen on Little Bense, suggesting that if there was a resident population on the island, it was likely much smaller than that on Bense.

Effectiveness and coverage of Application 1

For the first application (8–16 August), a team of five field staff covered Bense and Little Bense with bait, resulting in a mean density of 15.3 kg/ha. However, bait was more densely applied along the shoreline and in dense tussock, while it was applied less densely in the burn zone, which is devoid of vegetation. All cliffs were baited along their top edges and on all lower ledges that were safely accessible. Where safe access was not possible, pellets were thrown from above. Along accessible shorelines, particular attention was paid to the beach margin where vegetation began and to areas just above the high tide line where debris had accumulated.

Although Little Bense is only a third the size of Bense, baiting there proved to be much more challenging due to the extremely dense tussock grass and the fragmented, convoluted northern coast. Overland access to the many coastal chasms and rock slabs was particularly difficult since it required climbing through or over the worst of the tussock (over 2 m high). To apply bait to this northern coast more efficiently, a small boat was used. In some chasms the boat could be used as a mobile baiting platform, with pellets broadcast into the tussock from the deck. In other areas, personnel were landed to climb to the vegetated margin, then re-boarded and moved to the next position.

Effectiveness and coverage of Application 2

For the second application (26 August–3 September), a team of three field staff attempted to duplicate the coverage achieved in Application 1. However, due to the smaller team and staff injuries, this was not possible. Instead, Application 2 made selective coverage, with priority given to areas known from index trapping to be the best rodent and rabbit habitat. On Bense Island, Application 2 covered all tussock areas, all shorelines, and all areas north of the island's midline, regardless of vegetation type. Not covered were some areas of inland heath south of the midline, and the denuded burn zone. These latter areas still had intact bait remaining from Application 1.

On Little Bense Island, Application 2 covered all tussock areas and all shorelines, but did not cover inland heath areas. As on Bense, the inland heath here still had intact, uneaten bait remaining from the first application. On Little Bense, staff injuries also curtailed coverage in the tussock area: bait was applied on every second transect, meaning there was a gap of 40 m (instead of the normal 20 m) between each baiting line. To help reduce the size of the potentially un-baited space between transects, bait was thrown wider on lateral throws, and a greater quantity was thrown. The coastline was baited as in the first application, including the use of the boat to access the north coast. Mean baiting density on Application 2 was 11.7 kg/ha. Over the whole operation, c. 3,900 kg of bait were applied to the islands.

Daily reviews of the GPS tracks of workers revealed that some areas were missed in the earliest days of baiting but these were easily remedied the following day. After the first two days, all workers had mastered navigation and no further areas needed remediation.

Mammal and non-target mortality

Staff stayed on the islands from the start of Application 1 until seven days after the completion of Application 2 and during this time staff searched for animals killed in the operation. In total, 64 dead rabbits were found on Bense Island but none was found on Little Bense. All intact carcasses found were placed under heavy tussock grass or in burrows to hide them from scavenging birds. However, many carcasses were discovered after they had been scavenged, so some secondary poisoning is likely to have occurred. Three dead mice were found on Bense Island and one was found on Little Bense. No dead rats were found on either island, presumably because they died in their burrows.

Dolphin gulls (*Larus scoresbii*) were the most common non-target casualty with a total of 23 carcasses discovered. This species was observed to eat bait pellets directly, often fighting conspecifics for them. Dolphin gulls were the only species seen to eat the pellets. Three dead adult turkey vultures *(Cathartes aura jota)* were found, as was one adult striated caracara (*Phalcoboenus australis*) and one short-eared owl (*Asio flammeus*). The owl had been scavenged before discovery. Dissection of the striated caracara showed no visual evidence that it had directly ingested bait pellets, so perhaps it died from eating parts of a poisoned animal, most likely a rabbit or dolphin gull. Striated caracaras were observed playing with pellets but were never observed to ingest them. Two dead flightless steamer ducks (*Tachyeres brachypterus*) were found (one on each island). Direct consumption of bait may explain these deaths, but this species is known to eat offal occasionally (Woods, 1975) so it is also possible they were victims of secondary poisoning from eating a dead dolphin gull. Kelp gulls (*Larus dominicanus*) and snowy sheathbills (*Chionis alba*), two birds known for their curiosity and scavenging habits, were both present but were not seen to touch the pellets and no dead kelp gulls or sheathbills were recorded during the operation.

Post-operation follow-up

During the December 2016 follow-up visit, informal observations did not detect any live mammals, and no footprints were found despite careful examination of areas with soft soil or wet sand, where rabbit or rat tracks had been commonly seen in the past. The bait stations on Bense were also completely undisturbed with no evidence of gnawing on the wax baits. Three freshly-dead kelp gulls were found and evidence of pellet consumption was discovered upon dissection: the crops of two of the birds were discoloured with the bright green biomarker found in the pellets. It is thought these birds consumed bait that was inadvertently exposed during this visit when stored bait was moved near the campsite.

The more thorough post-operation visit in November 2017 did not discover any evidence of rodents or mammals on the islands. No live rodents or rabbits were seen, nor were any fresh droppings or tracks discovered. No chewsticks had been sampled by rodents, although the bite marks of striated caracaras and other birds were found on 10 sticks. Nocturnal observations with the thermal camera also found no mammals. However, bait blocks inside bait stations were found to be heavily sand-blasted and in need of replacement.

DISCUSSION

The first application of bait achieved 100% coverage as per the project design. However, Application 2 was less complete and several compromises were made, with priority given to bait the areas shown by index trapping to be the most important as habitat for invasive mammals. However, one area of concern was the dense tussock on Little Bense where the second application of baiting could have left gaps between baiting lines.

Rats have proved easier to eradicate from islands than mice, with rats successfully removed from islands in 92% of the operations attempted (Howald, et al., 2007), whereas early reports found success was achieved in only 62% of mouse operations (MacKay, et al., 2007). However, recent findings show a more optimistic picture, with mice successfully eradicated in 77% of operations in New Zealand, and this figure rises to 100% when considering only operations that followed current best-practice techniques (Broome, et al., 2019). Mice may be harder to eradicate because of behavioural traits such as aversion to cereal (Humphries, et al., 2000) or smaller home range (Clapperton, 2006; MacKay, et al., 2011). This necessitates a denser and more meticulous application of bait to ensure that all mice encounter pellets. The possible gaps in bait availability in dense tussock areas on Little Bense are thus a cause for concern.

Eradication operations carry a risk of killing non-target species through direct ingestion of poison pellets or by eating an animal that was poisoned. At South Georgia, brodifacoum pellets were consumed directly by skuas, sheathbills, and pintails, while other scavengers such as kelp gulls and giant petrels were less likely to eat baits (Lee, et al., 2013). In contrast, at Campbell (McClelland, 2011) and Macquarie Islands (Springer & Carmichael, 2012) kelp gulls were found to be extremely vulnerable to primary poisoning. In the Falklands, the death of non-target species is not well known since most islands have been without observers immediately after the completion of baiting operations. However, on Great Island, the bodies of many kelp and dolphin gulls were found following a rat eradication operation in July 2016 (T. Poole, pers. comm.). Dolphin gulls were the most common bird species poisoned on Bense and Little Bense Islands and their corpses were possibly a source of secondary poisoning of turkey vultures and striated caracara. It is suggested that future eradication operations in the Falklands plan for some personnel to remain on the island after the completion of baiting in order to improve understanding of non-target mortality.

That no evidence of mammals could be found on the island 14 months post-baiting is cause for optimism. However, the overall success of this operation will not be known until late 2018 (26 months post-baiting) after further monitoring has taken place. Elsewhere, rabbits have proven particularly difficult to eradicate using poison alone (Torr, 2002) and monitoring may reveal the need to use additional techniques on Bense and Little Bense Islands. There are no trained detection dogs in the Falklands and snares and fumigants are not advised as they could have an impact on burrowing penguins. In addition, biosecurity concerns prevent the import of rabbit-specific pathogens, thus leaving spotlight shooting as the most effective tool available for eliminating any remaining rabbits.

This Bense and Little Bense islands operation was intended to help restore native biodiversity with the potential to re-establish populations of the endemic Cobb's wren and blackish cinclodes. However, it will also contribute to future operations on other Falkland islands by allowing landowners to understand which eradication techniques do, or do not, work. As the first attempt to eradicate mice in the Falklands, the results will be especially helpful in planning for eradications on mouse-infested islands such as Steeple Jason Island, which is home to many seabird species and has been identified as an Important Bird Area. (Falklands Conservation, 2006). In the Falkland Islands, private landowners have been a driving force in many ecological restoration projects, so the training and experience gained by local residents in the course of the Bense operation also serves to increase the pool of skilled staff who can participate in future eradications on other islands.

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REFERENCES

Bergstrom, D.M., Lucieer, A., Kiefer, K., Wasley, J., Belbin, L., Pedersen, T.K. and Chown, S.L. (2009). 'Indirect effects of invasive species removal devastate World Heritage island'. *Journal of Applied Ecology* 46: 73–81.

Broome, K.G., Brown, D, Cox, A., Cromarty, P., McClelland, P., Golding, C., Griffiths, R., and Bell, P. (2011). *Current Agreed Best Practice for Rodent Eradication – Hand Broadcasting Poison Bait* (Version 1.3). New Zealand Department of Conservation internal document DOCDM-458533. Wellington, New Zealand: Department of Conservation.

Broome, K., Brown, D., Brown, K., Murphy, E., Birmingham, C., Golding, C., Corson, P. and Cox, A. (2019). 'House mice on islands: management and lessons from New Zealand'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 100–107. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Brown, D., Chadderton, L. and Brown, K. (2001). *Report on the Falklands Conservation Rat Eradication Project*. Stanley, F.I.: Falklands Conservation.

Carey, P.W. (2015). 'Remote Ocean Outposts'. In: D. Liggett, B. Storey, Y. Cook, and V. Meduna (eds.), *Exploring the Last Continent*, pp. 157–173. Heidelberg, Germany: Springer.

Clapperton, K.A. (2006). *A Review of the Current Knowledge of Rodent Behaviour*. Science for Conservation 263. Wellington, New Zealand: Department of Conservation.

Cunningham, D.M. and Moors, P.J. (1996). *Guide to the Identification and Collection of New Zealand Rodents, 3rd Edition*. Wellington, New Zealand: Department of Conservation.

Falklands Conservation (2006). *Important Bird Areas of the Falkland Islands*. London, UK: Falklands Conservation, Royal Society for the Protection of Birds, and BirdLife International.

FIG. (2015). *Rat-free Islands in the Falkland Islands Spreadsheet.* Version May 2015. Falkland Islands Government.

FITB. (2016). *Falkland Islands Tour Guide Handbook*. 3rd Edition. Stanley, F.I.: Falkland Islands Tourist Board.

Helmstedt, K.J., Shaw, J.D., Bode, M., Terauds, A., Springer, K., Robinson, S.A., Possingham, H.P. and Driscoll, D. (2016). 'Prioritizing eradication actions on islands: It's not all or nothing'. *Journal of Applied Ecology* 53: 733–741.

Howald, G.C., Donlan, J., Galvan, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A., and Tershy, B.R. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Humphries, R.E., Sibly, R.M. and Meehan, A.P. (2000). 'Cereal aversion in behaviourally resistant house mice in Birmingham, UK'. *Applied Animal Behaviour Science* 66: 323–333.

Lee, J., Black, A., Parker, G., and Rexer-Huber, K. (2013). *Report on Mortality of Non-target Species Following Year 1 Phase 2 of the South Georgia Rodent Eradication*. Stanley, F.I.: Government of South Georgia and the South Sandwich Islands.

MacKay, J.W.B., Russell, J.C., and Murphy, E.C. (2007). 'Eradicating house mice from islands: successes, failures, and the way forward'. In: G. W. Witmer, W. C. Pitt, and K.A. Fagerstone (eds.) *Managing Vertebrate Invasive Species: Proceedings of an International Symposium*, pp. 294–304. USDA/APHIS/WS, National Wildlife Research Center, Fort Collins, CO.

MacKay, J.W.B., Murphy, E.C., Anderson, S.H., Russell, J.C., Hauber, M.E., and Clout, M.N. (2011). 'A successful mouse eradication explained by site-specific population data'. In: C.R. Veitch, M.N. Clout, and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 198–203. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

McClelland, P.J. (2011). 'Campbell Island – pushing the boundaries of rat eradications'. In: C.R. Veitch, M.N. Clout, and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 204–207. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Miller, C. (2008). *Report from Rat Eradication Workshop, Stanley, Falkland Islands*. Sandy, UK: Royal Society for the Protection of Birds.

Moore, D.M. (1968). *The Vascular Flora of the Falkland Islands*. British Antarctic Survey Reports No. 60. London, UK: Natural Environment Research Council.

Poncet, S., Poncet, L., Poncet, D., Christie, D., Dockrill, C. and Brown, D. (2011). 'Introduced Mammal Eradications in the Falkland Islands and South Georgia'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 332–336. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Springer, K. (2016). 'Methodology and challenges of a complex multi-species eradication in the sub-Antarctic and immediate effects of invasive species removal'. *New Zealand Journal of Ecology* 40: 273–278.

Springer, K., and Carmichael, N. (2012). 'Non-target Species Management for the Macquarie Island Pest Eradication Project'. In: R.M. Timm (ed.) *Proceedings of the 25th Vertebrate Pest Conference*, pp. 38–48. Davis, USA: University of California, Davis.

St Clair, J.J.H. (2011). 'The impacts of invasive rodents on island invertebrates'. *Biological Conservation* 144: 68–81.

Strange, I.J. (1992). *A Field Guide to the Wildlife of the Falkland Islands and South Georgia*. London, UK: Harper Collins Publishers.

Strange, I.J. (2007). *New Island Falkland Islands: A South Atlantic Wildlife Sanctuary for Conservation Management*. Hook, UK: New Island Conservation Trust.

Tabak, M.A., Poncet, S., Passfield, K. and Martinez del Rio, C. (2014). 'Invasive species and land bird diversity on remote South Atlantic islands'. *Biological Invasions* 16: 341–352.

Tabak, M.A., Poncet, S., Passfield, K., Goheen, J.R., Martinez del Rio, C. and Griffen, B. (2015). 'Rat eradication and the resistance and resilience of passerine bird assemblages in the Falkland Islands'. *Journal of Animal Ecology* 84: 755–764.

Tabak, M.A., Anderson, O.R.J., Robb, G., Poncet, S., Passfield, K., Martinez, M.G. and Martinez Del Rio, C. (2016). 'Comparative isotopic natural history of two native passerines (*Troglodytes cobbi* and *Cinclodes antarcticus*) and the invasive rats (*Rattus norvegicus*) that extirpate them'. *Austral Ecology* 41: 622–632.

Torr, N. (2002). 'Eradication of rabbits and mice from Subantarctic Enderby and Rose Islands'. In: C.R. Veitch and M.N. Clout (eds.). *Turning the tide: the eradication of invasive species*, pp. 319–328. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Woods, R.W. (1975). *The Birds of the Falkland Islands*. Owestry, UK: Anthony Nelson.

Woods, R.W. and Woods, A. (1997). *Atlas of Breeding Birds of the Falkland Islands*. Owestry, UK: Anthony Nelson.

Woods, R.W. and Woods, A. (2006). *Birds and Mammals of the Falkland Islands*. Old Basing, UK: Wild Guides.

The history of the aerial application of rodenticide in New Zealand

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Abstract Following the incursion of rats (*Rattus rattus*) on Taukihepa (Big South Cape Island; 93.9 km²) off southern New Zealand in 1963, and the subsequent extirpation of several endemic species, the New Zealand Wildlife Service realised that, contrary to general belief at the time, introduced predators do not reach a natural balance with native species and that a safe breeding habitat for an increasing number of 'at risk' species was urgently needed. Offshore islands offered the best option for providing predator free habitat but there was a limited number of predator-free islands available and most were very small. Eradicating rodents on larger islands to provide a wider range and greater area of habitats was required and hand treating these larger areas using trapping and hand application of toxicants, the only methods available at the time, proved problematic and often impossible. Helicopters had been used to distribute bait for the control of rabbits and brushtail possums in the past but eradication of any particular predator species was considered 'not feasible'. The development of a GPS-based aircraft guidance system, a suitable bait product, specialised bait delivery systems and second-generation anti-coagulant toxicants changed that. Now islands as large as South Georgia (3,900 km²) have been treated using this method.

Keywords: aerial application, brodifacoum, eradication, helicopter, *Mus musculus* – house mouse, *Rattus exulans* – Pacific rat, *Rattus norvegicus* – Norway rat, *Rattus rattus* – ship rat

INTRODUCTION

New Zealand's terrestrial flora and fauna evolved in isolation from mammalian predators leading to many species being highly susceptible to any ground-based predators that hunt by smell and sight (e.g. Tennyson & Martinson, 2006). Since the arrival of humans, this unique environment has suffered from the deliberate or accidental introduction of a range of species that have decimated native biodiversity. This includes four species of rodent, Norway rat (*Rattus norvegicus),* ship rat *(R. rattus),* Pacific rat or kiore *(R. exulans*) and house mouse (*Mus musculus*), which continue to have a devastating impact on New Zealand's native flora and fauna (King, 2005).

Polynesians arrived in New Zealand bringing with them the Pacific rat or kiore. The rats, along with kuri or native dog (*Canis familiaris*), were brought for food and clothing and led to the first wave of extinctions in New Zealand (Tennyson & Martinson, 2006). In 1770, James Cook mentions vermin in his journals and this may refer to Norway rat (Innes, 2005). House mice had arrived in New Zealand by 1830 (Ruscoe & Murphy, 2005). Ship rats were introduced with early European settlers between 1860 and 1890 and had both cumulative and additional impacts to the rodent species that were already present (King, 2005; Tennyson & Martinson, 2006).

Invasive species have caused ecological problems around the world since humans started exploring but it was in New Zealand, where biodiversity loss was obviously due to introduced predators (Tennyson & Martinson, 2006), that organisations began to consider ways to minimise these impacts. It was not until the mid-1990s that technology advanced to a stage where this human induced disaster could be offset on any significant scale (Towns, et al., 2013).

This paper outlines the key events that led to the development of a rodent eradication tool used around the world today and discusses the role played in this process by the New Zealand agricultural aviation industry.

RECOGNITION OF THE DAMAGE RODENTS COULD DO TO NEW ZEALAND WILDLIFE

The ship rat invasion of Taukihepa (Big South Cape Island; 93.9 km²) in the early 1960s and the extinction of three species of endemic vertebrates sent shock waves through conservation circles (Bell, et al., 2016). A fourth species was saved only by transferring to it a nearby predator-free island. This disaster led to an increased interest in the ecology of rodents and their impact on native species as well as ways to control or eradicate them along with other introduced predators (Towns & Broome, 2003).

ERADICATION TOOLS AND ADVANCES

Early application of aircraft in New Zealand agriculture

Demobilised World War II pilots in New Zealand began an industry applying fertiliser and grass seed to hill country and established the skills to fly accurate parallel swath patterns. The spread of fertiliser and seed initially used fixed wing aircraft as outlined by Alexander & Tullett (1967), but the skills were later transferred to the use of helicopters.

The skill and experience of the pilots is a crucial component of any aerial baiting operation. In addition to having experience with all the systems that are to be used

Fig. 1 Auster aircraft loading rabbit bait, MacKenzie Basin 1951.

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in the operation (e.g. helicopter, bucket, GPS etc.), they are often required to fly under adverse conditions such as during poor weather, across islands with challenging topography and frequently a high risk of bird strikes. Pilots are expected to fly accurate lines in spite of these challenges whilst also monitoring the bait flow out of the bucket. It is highly desirable that the pilots are involved in the planning for an eradication as they can identify both risks and opportunities associated with the bait application.

The establishment of the Department of Conservation

The establishment of the New Zealand Government's Department of Conservation (DOC) out of the Wildlife Service, Forest Service and Department of Lands and Survey brought the various government agencies charged with protecting biodiversity under one management regime and allowed better focus on prioritising 'endangered species' programmes, including predator removal. The Department of Conservation was able to provide the financial and political support necessary to carry out this work. This was especially so with the larger projects such as Campbell Island (113.3 km²) in the New Zealand sub-Antarctic. Current operations now follow the international trend of joint venture or partnership operations with Non-Government Organisations (NGOs) and private conservation trusts.

Fig. 3 Purpose built eradication bucket 2001.

IMPROVEMENTS IN TECHNOLOGY

Development of toxins

On the mainland, compressed grain bait (pellets) suitable for dispersal through a mechanised spreader bucket (Fig. 2) were also laced with 1080 and phosphorus to target brushtailed possums (*Trichosurus vulpecula*) (Bill Simmons pers. comm.). Prior to this, aerial bait application had been predominantly diced carrot or grain.

The development of the second-generation blood anticoagulant toxicant brodifacoum in England in the mid-1970s provided a toxicant suitable for large-scale rodent eradication (Dubock & Kaudeinen, 1978). The delayed action of the anticoagulant toxicants meant that rodents would consume a lethal dose of toxicant before showing any symptoms, thus eliminating the risk of bait avoidance. Brodifacoum also has the ability to kill a rodent with a single feed, compared to the first-generation anti coagulants that required multiple feeds over several days. Brodifacoum is currently registered in over 40 countries in the form of over 100 separate registrations covering different formulations or product forms (Kaudeinen & Rampaud, 1986).

Fig. 2 Compressed cereal bait impregnated with brodifacoum.

Development of bait spreading equipment

Various New Zealand agricultural helicopter companies had been developing underslung cargo hook-mounted spreader buckets for the application of fertiliser and seed. By 1980, these spreader buckets had been modified to spread toxin-laced chopped carrot and cereal-based pellets for the control of rabbits and possums (Peter Garden, unpublished data).

Purpose-built bait-spreading buckets have continued to be developed (Fig. 3), and these now allow for a consistent swath width and density of bait application on a large scale. Buckets have been repeatedly refined to provide a wider bait swath and, most importantly, the addition of an internal deflector to direct bait just out one side minimising any bait that may go into the marine environment as well as being able to treat cliffs. Additional improvements including linking the bait flow to the flight track recording system are currently being developed.

Development of guidance and data recording equipment

Various methods to assist pilots in following straight lines have been tried. One of these, the Decca Navigation System, was used on forestry spraying operations as early as 1980 and used in a possum control operation on Rangitoto Island in 1990. Another method trialled was using reciprocal compass headings at the end of each run. This required the pilot to make calculations using compass variation, deviation and cross wind headings.

The United States military developed a constellation of global orbiting satellites in the late 1970s to provide very accurate navigation information. The Global Positioning System (GPS) relies on highly accurate time and position information transmitted by these satellites to receivers on the ground or in aircraft. The receivers use triangulation to compute three-dimensional position, direction and speed of travel information. To preserve security of this information, deliberate errors were factored in and the corrections for these errors were only available to those with security clearance to use them. This error factor was known as 'selective availability'. The civilian world was keen to access this information and several companies developed simple navigation devices that could be used for guidance within the expected error range. The error range was not consistent but was never much more than a few hundred meters, which was acceptable to support other navigational equipment. However, to be an effective guidance tool for aerial application this error could be no more than one or two metres. In 1993, attempts were made to use GPS for guiding bait spread onto Cuvier Island, but a suitable satellite triangulation system at that time was not available (D.R. Towns, pers. comm.).

In 1995, an American avionics manufacturer, Trimble Navigation, set up a facility in Christchurch New Zealand with the specific purpose of developing systems for use in aerial agricultural application that could meet the very stringent accuracy requirements of that industry. The system required the use of a 'base station' that recorded satellite signals transmitted over time and calculated the errors. The corrected information was then transmitted to the aircraft by radio telemetry.

By 2000, the US military had switched off the 'selective availability' function so the use of base stations was no longer necessary. More recently, a New Zealand based company, TracMap Ltd™, has developed a system designed specifically for aerial application – for the distribution of both agricultural products and bait (Fig. 4).

The first island eradication where GPS guidance equipment was successfully used was on Tiritiri Matangi (1.7 km²) in 1993 (Veitch, 2002d).

ERADICATION HISTORY

Early aerial application of toxicants

Rabbits (*Oryctolagus cuniculus*) were introduced for sport and as supplementary food for settlers in the 1830s (King 2005). However, the animals soon developed into plague proportions, particularly in the drier inland areas where they contributed to significant land erosion (King, 2005). Systems were developed for the aerial application of toxicants to control rabbits using fixed wing aircraft (Fig. 1). This was predominantly using either carrot pieces or grain laced with the toxin 1080 (sodium monofluoroacetate).

The first recorded island rat eradication in New Zealand was the removal of Norway rats by hand baiting from Maria Island (1 ha), Noises Islands, in 1960 (Towns & Broome, 2003). as the first in a series of unintended rodent eradications when control had been the expected outcome.

Fig. 4 TracMap™ GPS guidance equipment fitted to South Georgia Heritage Trust aircraft,2015.

The first use of bait stations was by Ian McFadden on Rurima Island (0.045 km²) in 1983, using maize laced with the anticoagulant bromadialone and the same product was used successfully on Korapuki Island (0.18 km²) in 1986. Both campaigns were against Pacific rats (and rabbits on Korapuki, McFadden & Towns, 1991). Between 1986 and 1988, commercially available Talon™ (brodifacoum) wax blocks in bait stations were used to eradicate Norway rats from Hawea (9 ha) and Breaksea (1.70 km²) islands in Fiordland (Thomas & Taylor, 2002). While this type of technique has been used on islands as large as 31 km² Langara Island, Canada (Taylor, et al., 2000), the usefulness of this method is limited by topography of the target island and logistical difficulties associated with ensuring complete coverage of the island.

Early use of aircraft targeting rodents on islands

In 1986, Moutohora Island (1.43 km²) in the Bay of Plenty was the first island in New Zealand to be treated using aerially distributed toxic bait (Talon™ 20P, active ingredient brodifacoum) to target rabbits using a fertiliser spreading bucket. As an unplanned side effect, Norway rats were also removed as part of this operation (Jansen, 1993).

The first attempt at aerially distributing rodenticide targeting rats in New Zealand occurred on the Mokohinau Islands (0.73 km²) in the Hauraki Gulf in 1990 (Towns & Broome, 2003). This operation was carried out using a 'monsoon' firefighting bucket to spread Talon™ 20P and resulted in the removal of Pacific rats. However, it was identified that the bait spread was concentrated along a narrow swath, due to the bucket not having a spinner to spread the bait out, and hand spreading was required to fill in the gaps (McFadden & Greene, 1994).

Between 1991 and 1993 a partnership was developed between DOC and ICI Crop Care, to improve the durability of Talon™ 20P (brodifacoum) and to license the product for aerial spread against rodents. An efficient means of spreading the baits also needed to be developed. By 1993, Ian McFadden of DOC and Tony Monk of Heletranz had developed a bait bucket with spinner, for use against rodents on offshore islands. The bucket was used to spread Talon™ 20P to target Pacific rats on Cuvier Island (1.81 km²) in 1993 (Towns & Stephens, 1997).

Increasing the scale

The first large scale aerial application operation specifically targeting rodents (Norway and Pacific rats) was carried out on 19.65 km² Kapiti Island (Fig. 5) off the south-west side of the North Island, New Zealand (Miskelly & Empson, 1999). The operation succeeded in removing both species. This island was four times larger than any previously attempted (Broome, 2009).

Fig. 5 Mechanical loading of bait for Kapiti Island, 1996.

Fig. 6 Hand loading bait on Codfish/Whenua Hou, 1997.

Fig. 8 Bait spreading on Mokonui Island, off Stewart Island, 2006.

Pacific rats were eradicated from Putauhina Island (1.41 km²) and Raratoka Island (0.88 km²) off southern Stewart Island in 1997 in the lead-up to rodent eradication on Whenua Hou (Codfish Is; 13.96 km²). (McClelland, 2002) Although these islands had significant conservation values in their own right, the removal of rats was largely to establish procedures and issues for the treatment of Whenua Hou in order to provide a predator free environment to establish a kakapo breeding base (Merton, et al 2006)

In August 1998, two applications of brodifacoum-laced compressed cereal bait were aerially applied to 13.96 km² Whenua Hou (Fig. 6) to remove Pacific rats (McClelland, 2011). The Kapiti project used two applications and this has become the standard methodology for aerial bait applications for eradicating rats on islands worldwide, with modifications as required for each island.

Tuhua/Mayor Island (12.83 km²) in the Bay of Plenty, New Zealand was successfully treated for the removal of Norway rats and Pacific rats in 2000, largely to test the methods required against rats and cats on the much larger and more remote Raoul Island in the Kermadecs (Williams & Jones, 2003).

Campbell Island followed on from the success of the Kapiti and Codfish/Whena Hou eradication programmes. DOC embarked on a very ambitious plan to eradicate Norway rats from this 113.31 km² island, 700 kilometres south of mainland New Zealand. The logistics of this project far exceeded anything that had been contemplated previously and required a rethink on how such operations could be streamlined to make them logistically and financially feasible. The resulting operational plan called for a single application of just 50% of the standard bait rate. This was a substantial risk but the GPS navigation and spreader bucket technology and experienced pilots gave planners confidence in being able to achieve complete coverage. A 600 ha trial involving the aerial application of non-toxic bait with a biomarker was carried out to test the proposed methodology before the full operation (Fig. 7) was started. (McClelland, 2011). Over the period 2000 to 2008, more than a dozen islands around the New Zealand coastline were treated including: Raoul (29.38 km²) in the Kermadecs (Ambrose, 2006; Little Barrier (30.83 km²) in the Hauraki Gulf (Griffiths, et al., 2019); Bench (1.21 km²) and Pearl (5.12 km²) off Stewart Island (Brent Beaven pers. comm.); Coal (11 km²) Preservation Inlet, (Brown, 2013); Pomona (2.62 km²) and Rona Islands (0.6 km²) (Shaw & Torr, 2011). Notable during this period was the Rakiura Titi Islands restoration project (McClelland, et al., 2011) which included Mokonui (0.86 km²) (Fig. 8) and Taukihepa/ Big South Cape (9.39 km²) islands. Managing non-target risks, multi-species eradications and reinvasion issues are all now part of the planning process and this culminated in the Rangitoto/Motutapu project 34.81 km² in 2009 that targeted seven species of introduced mammals including the four species of rodent (*M. musculus, R. rattus, R. norvegicus, R. exulans*). (Griffiths, et al., 2015).

Mice removal from 20.02 km² Antipodes Island 850 km south-east of Bluff (New Zealand) occurred in winter, 2016 (Horn & Hawkins, 2017) (Fig. 9). Success has been confirmed.

Fig. 7 Spreading bait on cliffs, Campbell Island, 2001.

Fig. 9 Mouse eradication operations Antipodes Island, 2016.

INTERNATIONAL PROJECTS

Exporting the technology

Because of the concern for the critically endangered Seychelles magpie robin *(Copsychus sechellarum),* an operation to carry out the eradication of Norway rats from Denis (1.43 km²), Frigate (2.19 km²) and Curieuse (2.86 km²) Islands in the Seychelles was completed in June and July 2000 (Merton, et al., 2002).

The same basic technique, usually using New Zealand-made spreader buckets and often with experienced New Zealand pilots, has been and is used to eradicate rodents on islands worldwide. Methods are modified for each island with alterations made to sowing density, number of drops, timing between drops etc., To date rodents have been eradicated from more than 300 islands using this technique, making it the most widely used and most successful technique for rodent eradications compared to bait stations, hand broadcast or traps. (Howald, et al., 2007). Whereas there are still some situations where the other techniques are the most suitable option, e.g. on islands where it is not practical to use aerial eradication methods it has allowed islands that could never previously have been considered for eradication programmes to be treated successfully. The largest island worked on to date is 3900 km² (1070 km² treated) South Georgia Island in the sub-Antarctic (Black, et al., 2013), which had Norway rats and an isolated population of mice treated in three phases over a five-year period from 2011 to 2015. Other successful international eradications using this methodology include Macquarie (128 km²) where rabbits, ship rats and mice were eradicated in 2012 (Parks and Wildlife Service, 2014) and Rat Island/Hawadax (10 km²) in the Aleutians where Norway rats were eradicated in 2008 (Buckelew, et al., 2011).

Aerial distribution of bait has now been successfully used for the eradication of rodents in more than ten countries including Australia, USA, Canada, Mexico, Japan, Italy and several smaller Pacific Island nations.

CONCLUSION

The aerial dispersal of rodenticide has been a 'game changer' allowing large and geographically challenging islands and tracts of land to be treated quickly and efficiently. The advent of GPS guidance and recording equipment and purpose-built distribution systems (spreader buckets) has given project managers confidence that a lethal dose of toxic bait can be delivered into each home range of the target species, maximising the chances of eradication.

Many organisations and islands around the world have benefited from the developments carried out in New Zealand since the availability of second-generation anticoagulant toxicants. Now NGOs and Government departments in all corners of the globe are using this information to carry out their own projects. These in turn are now providing feedback to advance the knowledge base needed to carry out ever more complex and challenging projects.

While aerial application of toxic bait has been a major advancement in habitat restoration, ground based techniques – bait stations and hand broadcast – are still used where relevant. These methods tend to be used on smaller, more accessible islands as well as around dwellings on inhabited islands during aerial operations. However, the ability to treat large areas in a short space of time and the lower overall cost per hectare of treatment make aerial application a valuable tool in the continuing fight against invasive predators. The scale of islands that may be treated in the future is limited only by the supporting logistics, funding and political support.

The fact that much of the aerial application expertise resides in New Zealand has more to do with the incremental development of systems, procedures and technology that has occurred here over the past 30 years. As the baiting pilot has the final control over the success of any project, it is vital that they have complete commitment to that end. Project managers should involve the likely application pilot(s) at an early stage to ensure this commitment.

Many challenges still exist, especially in tropical and subtropical regions where success rates have been lower, and there is room for continued development of equipment and systems, but the use of this method of distribution of rodenticide will continue into the foreseeable future. An increasing number of inhabited islands is now being treated and this brings a new series of challenges for project managers. Numerous issues that do not need to be considered on uninhabited islands come into play, making these operations considerably more complex.

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REFERENCES

Ambrose, M. (2006). *Operational Report for Cat, Kiore and Norway Rat Eradication on Raoul Island in the Kermadec Group*. Unpublished Report, Department of Conservation, New Zealand.

Alexander, G. and Tullett, J.S. (1967). *The Super Men*. Wellington: A.H. & A.W. Reed.

Bell, E.A.; Bell, B.D. and Merton D.V. (2016). 'The legacy of Big South Cape: rat irruption to rat eradication'. *New Zealand Journal of Ecology* 40(2): 212–218.

Black, A., Poncet, S., Wolfaardt, A., Peters, D., Hart, T., Wolfaardt, L-A., Tasker, M. and Rexer-Huber, K. (2013). *Rodent Eradication on South Georgia – Preparation and Evaluation: A Summary Report of Activities During the 2011/2012 Field Season*. United Kingdom: Overseas Territory Environment Programme.

Broome, K.G. (2009) 'Beyond Kapiti – A decade of invasive rodent eradications from New Zealand islands'. *Biodiversity* 10: 14–24.

Brown, D. (2013). *Coal Island (Te Puka-Hereka) Restoration Plan*. Unpublished Report, Department of Conservation, Southland Conservancy, New Zealand.

Buckelew, S., Byrd, V., Howald, G., MacLean, S. and Sheppard, J. (2011). 'Preliminary ecosystem response following invasive Norway rat eradication on Rat Island, Aleutian Islands, Alaska'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 275–279. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Cash, W. and Gaze, P. (2000). 'Restoration of Motuara Island – Queen Charlotte Sound'. *Ecological Management* 8: 31–36.

Chappell, R. (2008). *Operational Plan for Eradication of Ship Rats* (Rattus rattus), *Kiore* (Rattus exulans), *House Mouse* (Mus musculus) *and Rabbit* (Oryctolagus cuniculus cuniculus) *from Ohinau and Motutapere Islands*. Department of Conservation, New Zealand. DOCDM-314307.

Corson, P. and Hawkins, K. (2016). *The Eradication of Kiore, Ship Rats and Cats from Ahuahu - Great Mercury Island.* Department of Conservation internal report DOCDM-1477863.

Dubock, A.C. and Kaukeinen, D.E. (1978). 'Brodifacoum (Talon™ Rodenticide), a Novel Concept'. In: W.E. Howard and R.E. Marsh (eds.). *Proceedings of the Eight Vertebrate Pest Conference*, p. 127. Lincoln: University of Nebraska.

Griffiths, R., Buchanan, F., Broome, K., Neilson, J., Brown, D. and Weakley, M. (2015). 'Successful eradication of invasive vertebrates on Rangitoto and Motutapu islands, New Zealand'. *Biological Invasions* 17: 1355–1369.

Griffiths, R., Bell, E.A., Campbell, D.J., Cassey, P., Ewen, J.G., Green, C.J., Joyce, L., Rayner, M.J., Toy, R., Towns, D.R., Wade, L., Walle, R. and Veitch, C.R. (2019) 'Costs and benefits for biodiversity following rat and cat eradication on Te Hauturu-o-Toi/Little Barrier Island'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island Invasives: Scaling Up to Meet the Challenge,* pp. 558–567. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Golding, C. (2010). 'House mouse (*Mus musculus*) eradication by aerial bait application on Adele, Tonga and Fisherman Islands, Abel Tasman National Park, New Zealand'. *Conservation Evidence* 7: 62–88.

Gutsell, M. (2011). *Indian Island Rodent Eradication*. Department of Conservation internal report DOC-752616.

Hawkins, K. (2011). *Taranga (Hen) Restoration Project Restore, Taranga Island kiore eradication programme final report May to June 2011.* Department of Conservation internal report DOC-5455561.

Howald, G., Donlan, C.J., Galván, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Horn, S.R. and Hawkins, K. (2017). *Project Report, Antipodes Island Mouse Eradication*. Department of Conservation internal report DOC-3000055.

Innes, J. (2005). 'Norway Rat'. In: C.M. King (ed.) *The Handbook of New Zealand Mammals,* pp.174–187. Melbourne: Oxford University Press.

Jansen, W.P. (1993). 'Eradication of Norway rats and rabbits from Moutohora (Whale) Island, Bay of Plenty'. *Ecological Management* 1: 10–15.

King, C.M. (2005). *The Handbook of New Zealand Mammals*. Melbourne: Oxford University Press.

Kaudeinen, D.E. and Rampaud, M.A. (1986). 'Review of Brodifacoum Efficacy in the US and Worldwide'. In: P. Salmon, R.E. Marsh and D.E. Beadle (eds.) *Proceedings of the Twelfth Vertebrate Pest Conference,* p. 17. California: University of California, Davis.

Lee, M. (2016) *Motu Kaikoura Sanctuary Rat Management Programme – Progress Report (No.3)*. Unpublished report. Motu Kaikoura Trust.

McClelland, P.J. (2002). 'Eradication of Pacific rats from Whenua Hou'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 181–189. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

McClelland, P.J. (2011). 'Campbell Island, Pushing the Boundaries'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 204–207. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

McClelland, P.J., Coote, R., Trow, M., Hutchins, P., Nevins, H.M., Adams, J., Newman, J. and Moller, H. (2011). 'The Rakiura Tītī Islands restoration project': In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 451–454. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

McFadden, I. and Towns, D.R. (1991). 'Eradication campaigns against kiore (*Rattus exulans*) on Rurima Rocks and Korapuki Island, northern New Zealand'. *Science and Research Internal Report No. 97*. Wellington: Department of Conservation.

McFadden, I. and Greene, T. (1994). 'Using brodifacoum to eradicate kiore (*Rattus exulans*) from Burgess Island and the Knights Group of the Mokohinau Islands'. *Science and Research Series No. 70.* Wellington: Department of Conservation.

MacKay, J.W.B., Murphy, E.C., Anderson, S.H., Russell, J.C., Hauber, M.E., Wilson, D.J. and Clout, M.N. (2011). 'A successful mouse eradication explained by site-specific population data'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 198–203. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Merton, D.V., Climo, G., Laboudallon, V., Robert, S. and Mander, C. (2002). 'Alien mammal eradication and quarantine on inhabited islands in the Seychelles'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 85–91. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Merton, D.V., Powlesland, R.G. and Cockrem, J.F. (2006). 'A parrot apart: the natural history of the kakapo (Strigops habroptilus), and the context of its conservation management'. *Notornis* 53(1): 3–26.

Miskelly, R.A. and Empson, C.M. (1999). 'The risks, costs and benefits of using brodifacoum to eradicate rats from Kapiti Island, New Zealand'. *New Zealand Journal of Ecology* 23: 241–254.

Owen, K. (1997). 'Transformation of an Inland Island (Mokoia Island) Now Home to Endangered Species'. In: T. Johnson and P. Boyes (eds.) *New Zealand Conservation Management Group Annual Conference Proceedings*, pp. 74–87. Rotorua: DOC Research & Development Series 284.

Parks and Wildlife Service (2014). *Evaluation Report: Macquarie Island Pest Eradication Project*, August 2014. Hobart, Tasmania: Department of Primary Industries, Parks, Water and Environment.

Project Island Song (2009). 'Building a Wildlife Sanctuary'. <www.projectislandsong.co.nz>.

Ruscoe, W.A. and Murphy, E.C. (2005). 'House Mouse'. In: C.M. King (ed.) *The Handbook of New Zealand Mammals,* pp. 204–221. Melbourne: Oxford University Press.

Shaw, V. and Torr, N. (2011). 'Eradicating mammal pests from Pomona and Rona Islands in Lake Manapouri, New Zealand: A focus on rodents'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 356–360. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Taylor, R., Kaiser, G.W. and Drever, M.C. (2000). 'Eradication of Norway rats for recovery of seabird habitat on Langara Island, British Columbia'. *Restoration Ecology* 8: 151–160.

Tennyson A. and Martinson P. (2006). *Extinct Birds of New Zealand*. Wellington: Te Papa Press.

Thomas, B.W. and Taylor, R.H. (2002) 'A History of Ground-based Rodent Eradication Techniques Developed in New Zealand, 1959–1993'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 301–310. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Torr, N. (2002). 'Eradication of rabbits and mice from Subantarctic Enderby and Rose Islands'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 319–336. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Towns, D.R. and Stephens, T. (1997). 'Island management and commercial sponsorship: The Mercury Islands experience'. *Science and Research Series No.* 103. Wellington: Department of Conservation.

Towns, D.R. and Broome, K.G. (2003). 'From small Maria to massive Campbell: Forty years of rat eradications from New Zealand islands'. *New Zealand Journal of Zoology* 30: 377–398.

Towns, D.R. and Parrish, R (2003). 'Restoration of the Principal Marotere Islands'. Department of Conservation, New Zealand; Resource Management Unit, Ngatiwai Trust Board.

Towns, D.R., West, C.J. and Broome, K.G. (2013). 'Purposes, outcomes and challenges of eradicating invasive mammals from New Zealand islands: an historical perspective'. *Wildlife Research* 40: 94–107.

Veitch, C.R. (2002a). 'Eradication of Norway rats (*Rattus norvegicus*) and house mouse (*Mus musculus*) from Browns Island (Motukorea), Hauraki Gulf, New Zealand'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 350–352. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Veitch, C.R. (2002b). 'Eradication of Norway rats (*Rattus norvegicus*) and house mouse (*Mus musculus*) from Motuihe Island, New Zealand'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 353–356. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Veitch, C.R. (2002c). 'Eradication of Pacific rats (*Rattus exulans*) from Fanal Island, New Zealand'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 357–359. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Veitch, C.R. (2002d). 'Eradication of Pacific Rats (*Rattus exulans*) from Tiritiri Matangi Island, Hauraki Gulf, New Zealand'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 360–364. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Williams, D. and Jones, A. (2003). *Operational Report for Cat, Kiore, Norway Rat Eradication on Mayor Island (Tuhua).* Unpublished Report, Department of Conservation, New Zealand.

Successes and failures of rat eradications on tropical islands: a comparative review of eight recent projects

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Abstract Rat eradication is a highly effective tool for conserving biodiversity, but one that requires considerable planning effort, a high level of precision during implementation and carries no guarantee of success. Overall, rates of success are generally high but lower for tropical islands where most biodiversity is at risk. We completed a qualitative comparative review on four successful and four unsuccessful tropical rat eradication projects to better understand the factors influencing the success of tropical rat eradications and shed light on how the risk of future failures can be minimised. Observations of juvenile rats surviving more than four weeks after bait application on two islands validate the previously considered theoretical risk that unweaned rats can remain isolated from exposure to rodent bait for a period. Juvenile rats emerging after bait was no longer readily available may have been the cause of some or all the project failures. The elevated availability of natural resources (primarily fruiting or seeding plants) generated by rainfall prior to project implementation (documented for three of the unsuccessful projects) may also have contributed to project failure by reducing the likelihood that all rats would consume sufficient rodent bait or compounding other factors such as rodent breeding. Our analysis highlights that rat eradication can be achieved on tropical islands but suggests that events that cannot be predicted with certainty in some tropical regions can act individually or in concert to reduce the likelihood of project success. We recommend research to determine the relative importance of these factors in the fate of future tropical projects and suggest that existing practices be re-evaluated for tropical island rodent eradications.

Keywords: best practice, conservation, invasive, restoration, rodent

INTRODUCTION

Marine islands house an estimated 15–20% of terrestrial biodiversity and are home to 61% of IUCN Extinct species and 37% of IUCN Critically Endangered species (B. Tershey unpubl. data). Invasive species have been the most frequent cause of extinctions on islands and the second leading cause of Critical Endangerment (B. Tershey unpubl. data). Commensal rats (*Rattus* spp.) are considered the most damaging group of invasive species on islands because of their near global distribution and the frequency with which they cause extinctions, extirpations and ecosystem-level impacts (Towns, et al., 2006; Howald, et al., 2007; Kurle, et al., 2008). Rats can be eradicated from islands (Keitt, et al., 2011) resulting in significant species and ecosystem recovery (Bellingham, et al., 2010). Thus, rat eradication is a powerful tool with which to prevent extinctions.

Although this tool has been widely deployed, with more than 500 successful rat eradications to date (DIISE, 2017), most rat eradications have been on small, mid to high latitude islands (Howald, et al., 2007) where endemic species diversity is lower. If rat eradication is to realise its full potential to prevent extinctions, then future eradications need to be more frequently conducted where endemic species diversity is high: on larger tropical islands (Kier, et al., 2009). However, while rat eradication is being successfully conducted on increasingly large, high latitude islands, with a failure rate of less than 3% (Russell & Holmes, 2015), success on both large and small tropical islands has been more elusive, with a failure rate of 10% and very little understanding as to the underlying causes of failure (Holmes, et al., 2015; Keitt, et al., 2015).

In an attempt to better understand the mechanisms responsible for eradication failure on tropical islands and improve the rate of success of future projects, a global review of rodent eradication practice on tropical islands was instigated (Russell & Holmes, 2015). In support of the review, Holmes, et al. (2015) performed a statistical analysis on as many rat eradication attempts as possible to determine correlative factors that might pinpoint important influences on tropical rat eradication success. However, rat eradication projects are complex and multifaceted (Cromarty, et al., 2002) and, like complex projects within other disciplines, it can be challenging to determine the reason(s) for project failure. To reduce the risk that the broad-brush approach utilised by Holmes, et al. (2015) overlooked important and influential factors, we completed a second review, this time using a qualitative framework on a subset of the projects assessed by Holmes, et al. (2015).

Qualitative comparative reviews are used extensively in the social and behavioural sciences (e.g. Ragin, 1989; George & Bennett, 2005; Bennett & Elman, 2006), but also in other fields such as software engineering (Abrahamsson, et al., 2003), human resource management (e.g. Allen, et al., 1997), and political science (e.g. Bennett & Elman, 2006). A qualitative comparative review offers the opportunity to compare projects and their nuances in detail, which superficially, statistical analyses cannot do, but also allows for the possibility for making generalisations if they exist (Ragin, 1989). This approach, which we believe has greater utility in conservation biology, offered a complementary mechanism for verifying or dispelling the importance of factors identified as significant or insignificant in Holmes, et al. (2015).

We examined in depth, reported data from eight well-planned and sufficiently resourced tropical rat eradication attempts, balanced among four successful and four unsuccessful projects, to better understand: 1) the variability in factors influencing tropical rat eradication projects irrespective of outcome, 2) the factors that consistently differentiate successful from failed tropical rat eradication attempts for projects where full reported data are available, 3) what steps can be taken to improve eradication reporting and minimise the risk of failure for future tropical rat eradications.

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METHODS

Island eradication study sites

For the purposes of this study we focused on rat eradication projects that used the method shown to have the greatest chance of success, but which faced all of the challenges associated with tropical islands and described by Keitt, et al. (2015). We did not consider geographical location to be important if these conditions were met. Projects that met the following criteria were selected for our analysis:

- Rodent bait was applied by helicopter, guided by GPS. Projects that used the aerial application of bait were the focus for our study because this method has the best record of success in both temperate and tropical climates (Howald, et al., 2007).
- The project was undertaken on a tropical island or islands. Although Henderson lies just south of the tropic of Capricorn at a latitude of 24°21'S, we considered this island to be tropical in the context of rodent eradication due to the island's temperature range, vegetation and absence of pronounced seasonality (Spencer, 1995; Brooke, et al., 1996).
- The project was undertaken on an island or islands with a Precipitation Coefficient of Variance (CV) of mean monthly rainfall of less than 50% (Fig. 1). We focused our analysis not on particularly wet or dry islands, but on islands where rainfall and ecosystem productivity were more difficult to predict. We excluded projects completed on arid or semi-arid islands such as along the Pacific Coast of Mexico or North-western Australia because, for rodent eradication, these islands share the seasonality associated with temperate islands i.e. an eradication operation can be undertaken when natural food resources are scarce and breeding, within the rat population, is less likely. The island of Banco Chinchorro, Mexico was excluded from our analysis because it had a rainfall CV greater than 50%. Nevertheless, Banco Chinchorro is another well documented project and could have been a useful addition to our comparative review.
- The project was undertaken on an island or islands with land crabs. The presence of land crabs was identified as a significant influence on project success in Holmes, et al. (2015).
- Projects where reinvasion could be dismissed as an unlikely cause of failure. Projects were only included if reinvasion had been ruled out through comparative DNA analysis or were undertaken on uninhabited islands that were rarely visited and extremely remote. This excluded islands such as Denis and Curieuse in the Seychelles (Merton, et al., 2002) and the Aleipata Islands in Samoa (Butler, et al., 2011).
- Sufficiently detailed information was available to allow the project to be reviewed within the framework recommended by Keitt, et al. (2015).

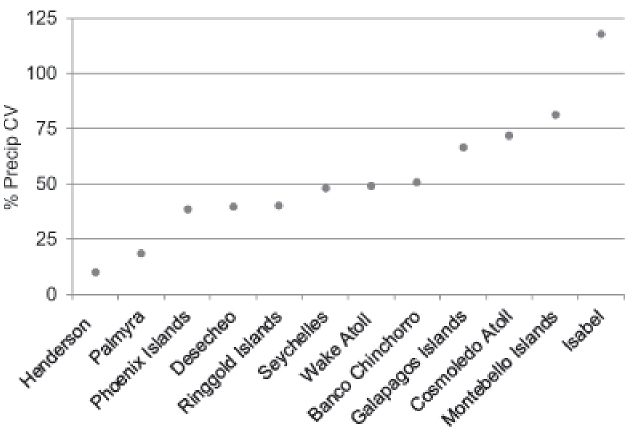

Fig. 1 Monthly Precipitation Coefficient of Variance (CV) for tropical islands where rodent eradications have been attempted using rodent bait containing a 2nd generation anticoagulant applied by helicopter.

Of the 17 discrete projects completed on tropical islands that applied rodent bait containing a second generation anticoagulant by helicopter, eight were selected for analysis. Six were completed on islands located in the tropical Pacific; Henderson (part of the United Kingdom Overseas Territory of Pitcairn), Wake (an unincorporated territory of the United States north of the Marshall Islands), Palmyra (an unincorporated territory of the United States in the Northern Line Islands), Enderbury and Birnie (part of the Phoenix Islands Group of the Republic of Kiribati) and the Ringgolds (part of Fiji). Two projects were located outside of the Pacific Region; Desecheo (Puerto Rico Archipelago) located in the Caribbean and Frégate (Seychelles) in the Indian Ocean.

Island size varied from 49 to 4,310 ha (Table 1) and all islands experienced relatively similar temperature ranges and annual rainfall (Table 1). Except for the Wake project that targeted Pacific rat (*R. exulans*) and Asian house rat (*R. tanezumi*), the eradication operations targeted the removal of just one species. *R. exulans* was targeted in four operations, ship rat (*R. rattus*) in two and Norway rat (*R. norvegicus*) in one (Table 1). Holmes, et al. (Holmes, et al., 2015) found no significant difference in eradication success between rat species for projects that applied bait aerially. Four of the islands were inhabited; Wake, Frégate, Palmyra and the Ringgolds (Table 1).

Determining success and failure

In line with best practice guidelines produced by the New Zealand Department of Conservation (Broome, et al., 2011), we considered an eradication project to be successful where the absence of rats was determined after a minimum of two breeding seasons (at least one year) after the completion of the operation, as rat populations may remain low and undetected for shorter periods. Rats were first reported as being present five months after the operation on Wake Island; eight months after on Henderson Island; 13 months after on Desecheo; and two years after on Enderbury. At the time of writing 14, six, four and three years have passed for the Frégate, Ringgolds, Birnie and Palmyra projects, respectively, and all four islands remain rat free. A failed attempt to eradicate rats from Palmyra Atoll in 2001 was hampered by both technical and implementation constraints and was not evaluated (USFWS, 2011).

Identifying potential factors that influenced success and failure.

While there are other alternate or contributing hypotheses (Table 2; Holmes, et al., 2015), the most proximate reason for the reduced rate of success for tropical rodent eradications is likely to be that not all rats consumed a lethal dose of brodifacoum, the rodenticide used in most rat eradications (Howald, et al., 2007) either because they did not have access to sufficient bait or because they did not consume bait that was available (Holmes, et al., 2015). We used the framework outlined in Keitt, et al. (2015) to review the four unsuccessful projects. To determine if some individuals within the rat population *could not* eat a lethal dose of bait, we reviewed operational design, operational procedures, GIS maps of bait coverage, baiting density,

Table 1 Characteristics of the islands where the eight projects were undertaken.

	Henderson	**Wake**	**Desecheo**	**Enderbury**	**Birnie**	**Palmyra**	**Ringgolds**	**Frégate**
Project outcome	Failed	Failed[a]	Failed	Failed	Succeeded	Succeeded	Succeeded	Succeeded
Latitude	24°21'S	19°18'N	18°23'N	3°8'S	3°35'S	5°53'S	16°30'S	4°35'S
Area (area of largest sub-unit of land) (ha)	4,310	696 (602)	116	608	49	235 (98)	266 (147)	219
Mean annual precipitation (mm)	1,623	1,780	750–1,039	750–1,300	750–1,301	4,422	2,467	2,182
Coefficient of variance for monthly precipitation (%)	10	49	40	40	38	19	39	49
Temperature Range (°C)	24–30	24–28	24–32	24–30	24–30	24–27	22–28	24–32
Maximum elevation (m)	33.5	6.4	213	6.7	4	1.8	12	125
Principal landform	Raised coral atoll	Coral atoll	Small mountainous island of volcanic origin	Coral atoll	Coral atoll	Coral atoll	Coral atoll	Granite island with two coastal plateaux
Principal vegetation types	Tangled, scrub and scrub-forest	Grass, low growing scrub, and *Casuarina* forest.	Grass and dry tropical *Bursera* forest	Grass and low growing scrub	Grass and low growing scrub	*Pisonia* and *Cocos* forest	Low scrub and *Cocos* forest	Modified forest
Target species for eradication	*R. exulans*	*R. exulans* and *R. tanezumi*	*R. rattus*	*R. exulans*	*R. exulans*	*R. rattus*	*R. exulans*	*R. norvegicus*
Permanent human population	0	100–200	0	0	0	12	2	20–30

[a]The Wake project successfully removed *Rattus tanezumi* but not *R. exulans* from Wake Atoll and successfully removed both species from part of the atoll, Peale Island (95 ha).

Table 2 Hypotheses to explain increased failure of rat eradications on tropical islands.

Proximate cause	Underlying cause	Possible response to increase success rates
Some individuals within the island's rat population could not eat a lethal dose of bait		
	Land crabs or other species consume bait	Higher bait application rates Additional bait applications Bait at a time when competitors are at lower density or less active
	Rats have small home ranges	Higher bait application rates Flexible scheduling to apply bait when food supply low
	Bait decomposes rapidly	More preservatives in bait Additional bait applications
	Lactating females or young in nest when bait available	Bait available longer (more bait, additional applications) Flexible scheduling to drop bait when breeding is reduced or non-existent.
	Rats don't leave human dwellings	Comprehensively bait entire island including within commensal areas
Some individuals within the island's rat population would not consume a lethal dose of bait		
	Bait biodegrades rapidly	More wax or preservatives in bait Additional bait applications
	Abundant natural food	Multiple bait formulations Bait available longer (more bait, additional applications) Flexible scheduling to drop bait when food supply low
	Individual foraging preferences	Multiple bait formulations Bait available longer (more bait, additional applications)
	Lactating females very neophobic	Bait available longer (more bait, additional applications)
	Different dietary preferences of lactating females	Multiple bait formulations Bait available longer (more bait, additional applications) Flexible scheduling to drop bait when food supply low
Poor quality planning and implementation		
	Lack of capability	More training & collaboration Appointment of experienced staff Adequate resourcing Peer review during the planning process
	Lax regulatory requirements	Plan & implement using internationally recognised standards
	Insufficient resourcing	Source more funding Increase collaboration
Higher rate of reinvasion		
	Warm water allows increased swimming distances	Select more isolated islands
	Human use characteristics	Better biosecurity Incorporate human use into island selection criteria

bait availability over time, timing between applications, and any operational difficulties noted. The statistical approach of Holmes, et al. (2015) could not address all of these issues because of the scarcity of well documented projects such as those we investigated. We also assessed bait toxicity and the chance that rats were resistant or tolerant to anticoagulants. Insufficient information was available to evaluate the impact of any spatial variation in land crab density across each of the islands.

To evaluate if some individuals within the rat population *would not* eat a lethal dose of bait, we looked at the operational design, the bait type, data from trials completed, the environmental conditions present at the time of the eradication and any observations made during implementation. Evidence for and against each factor was evaluated and used to form an opinion on its relative importance to the project's outcome. Evidence for the existence of a similar or different set of conditions for the successful projects was used to inform this analysis.

Not all projects monitored bait availability over time and for those projects that did, different methods were used, making it difficult to compare how long bait remained available to rats after its application. To compare between projects we used both the minimum period of time that bait was available in all plots or transects sampled and, where data were available, the lower limit of 99% CI of the T-Statistic for bait availability four days after its application as recommended by Pott, et al. (2015). For those islands where no monitoring was undertaken we used anecdotal reports to provide an estimate of the minimum period of bait availability.

Comparison among projects

We undertook a qualitative comparative review because the number of projects that formed the basis of our assessment was small, there was inconsistency between projects in the data collected and the methods by which data were obtained. A qualitative comparative review allows for generalisations to be made among cases and we considered it the best option for this study. Akin to Abrahamsson, et al. (2003), we cross-examined all projects to identify factors common to successful or unsuccessful projects. To inform this cross examination we drew from Holmes, et al. (2015) and our cumulative experience to identify a set of environmental variables and components of operational design we considered to be important to the success of rat eradication operations. These variables are listed in Tables 1–3. Information on each project was obtained from documentation prepared prior to and after project implementation and from personal communications with project team members.

RESULTS

Identifying causes of operational failure

Some individuals within the island's rat population could not eat a lethal dose of bait

The design of each of the four unsuccessful eradications, encompassing aerial application, overlapping aerial bait swaths, application rates comparatively higher than those applied in temperate regions and a minimum of two applications (Table 4), should have ensured comprehensive coverage of the islands with rodent bait. During the first bait application on Desecheo, some technical difficulties resulted in several small areas of the island (the largest being ~0.8 ha in size) receiving bait at less than the planned application rate. These issues were remedied for the second application when a more even spread of bait was achieved and, between both applications, comprehensive coverage of the island was achieved. Similarly, with the exception of areas deliberately excluded from bait application such as the sealed runway on Wake, we could not discern any biologically significant gaps in bait distribution from a review of the GIS data accumulated for any of the four unsuccessful projects. A biological gap was defined for our analysis as a gap greater than 0.015 ha in area. This was the smallest home range size reported in the literature for any of the four rat species targeted (Wirtz, 1972; King, 1990; Shiels, 2010; Low, et al. 2013).

On this basis we conclude that the operational strategy employed on Henderson, Desecheo and Enderbury likely ensured that all foraging rats encountered rodent bait. Although not identified from GIS maps of bait spread, it was more difficult to reach the same conclusion for Wake because of the more complex operational strategy (multiple methods of bait application) employed there (Griffiths, et al., 2014). The existence of interspecific competition, not a factor for the other islands, also likely limited access to bait for some individual rats. However, the successful eradication of *R. tanezumi,* formerly widespread across the atoll (Griffiths, et al., 2014), demonstrated that broad coverage across all habitats was achieved.

All four projects had factored bait consumption by non-target species such as land crabs into operational decisions on application rates (Table 4). However, bait disappeared more rapidly than anticipated from some transects monitored on Wake and Desecheo (Brown, et al., 2013; Brown & Tershy, 2013) (Table 4). Bait persisted in all transects monitored on Henderson until close to the end of the 30-day monitoring period (Brooke, et al., 2011). However, as described by Pott, et al. (2015), a different monitoring method was used and, because of the inaccessible nature of the island, monitoring was confined to a small part of the island. No monitoring of bait availability was undertaken on Enderbury but ad hoc observations suggest that rodent bait was broadly available for at least the first five days after its initial application (Pierce & Kerr, 2013).

Rat pups yet to emerge from the nest may not have had immediate access to bait. Evidence of rat breeding activity was documented on all four islands at the time of implementation (Brooke, et al., 2011; Brown, et al., 2013; Brown & Tershy, 2013; Pierce & Kerr, 2013). A rat of indeterminate age was sighted and captured on Desecheo, 23 days after the first bait application. On Wake, a juvenile *R. exulans* was found inside a bait station 18 days after bait was first applied and a second juvenile *R. exulans* was caught alive at the base of a coconut (*Cocos nucifera*) palm after 47 days. A low body weight and large head relative to body size indicated the latter individual had suffered from malnutrition likely because of having been weaned prematurely. As evidenced by liver assay, it had been exposed to brodifacoum (Griffiths, et al., 2014). No live rats were seen by project team members monitoring Henderson rails (*Porzana atra*) at the north-east end of Henderson beyond five days after the initial bait application, despite being on the island for more than three months after the operation. However, two very small, freshly dead, likely juvenile, rats were discovered 11 and 14 days after bait was applied suggesting these animals had survived for 10–13 days after the initial bait application.

Operational procedures were in some instances modified during project implementation due to environmental and physical factors encountered during the operation and/or the detection of a small number of rats after bait application. Lack of accurate geographical data led to an underestimate of island size for Henderson during project planning. As a consequence, the application rate for the second application across the island's plateau had to be reduced from 7 kg/ha to 6 kg/ha (Torr & Brown, 2012). Methods for applying bait to vegetated intertidal habitats were modified during implementation on Wake (Griffiths, et al., 2014). Bait stations were also deployed and bait was hand spread at several sites on Wake to target rats detected within five months of bait application, although such efforts were eventually abandoned after increasing numbers of rats sighted confirmed the eradication had been unsuccessful for *R. exulans* (Griffiths, et al., 2014). We do not consider the operational changes made for these three projects to have reduced the availability of bait to rats. No significant changes to the operational strategy were reported for the Enderbury project and bait application, as described by team members, followed the prescription outlined within the project's operational plan.

Based on the evidence available, we conclude that some individuals within the island's rat populations could not eat a lethal dose of bait. Unweaned rats present at the time of bait application did not have immediate access to bait and, as evidenced by individuals surviving for so long after bait application on Wake, this is also likely for some breeding female rats. However, we cannot conclude that this factor was the only cause of failure for the four failed projects.

Bait toxicity

Assays of samples of the rodent bait applied on Henderson (mean brodifacoum concentrations of 16.4 ppm), Wake (28.3 ppm) and Desecheo (29.3 ppm) confirmed that bait toxicity was within normal tolerances (Brown, et al., 2013; Brown & Tershy, 2013; RSPB, unpublished data). Inadequate bait toxicity is unlikely to have been a factor on Enderbury because the bait used there was produced at the same time as the bait used for the successful Birnie operation. Mortality associated with the operation and a rapid decline in rat numbers was also observed at all sites. All three bait types used are produced via an industrial production process with quality assurance checks in place to ensure appropriate rodenticide concentrations prior to shipping and all have been used successfully on both temperate and tropical islands. Based on the evidence available we conclude that inadequate bait toxicity was not a factor in the failure of the four unsuccessful projects reviewed.

Resistance

There were no indications to suggest rats on Henderson, Wake, Enderbury and Desecheo were resistant or tolerant to anticoagulants. Rats on Henderson, Enderbury and Desecheo had no prior exposure to anticoagulants so there was no selection pressure for pharmacodynamic resistance involving mutations in the Vkorc1 gene. For Henderson, subsequent testing of rats from the surviving population confirmed the lack of any genetic basis for resistance to brodifacoum (RSPB, unpubl. data). Although anticoagulants were used on Wake prior to the eradication (Mosher, et al., 2008) available evidence, as discussed in Griffiths, et al. (2014), did not support resistance as a factor in the project's outcome. Most importantly, although increased tolerance to brodifacoum has been documented for some rat populations, 'practical' resistance, as defined by Buckle & Prescott (2012), that might have caused the Wake project to fail, has never been encountered, even at sites where anticoagulants have been used repeatedly for long periods of time (Lund, 1984; Bailey, et al., 2005). It is unknown if any plant species present on Henderson, Wake, Desecheo and Enderbury contained elevated levels of vitamin K, but dietary-based resistance is not considered a major mechanism of resistance elsewhere (Buckle & Prescott, 2012). Based on the lack of evidence for resistance or increased tolerance to anticoagulants we conclude that this mechanism was not a factor in the recorded failures.

Some individuals within the island's rat population would not consume a lethal dose of bait

All four of the unsuccessful projects used proven bait types (Table 4) that have achieved rat eradication on other tropical islands. In addition, palatability of two of the bait types was proven by bait exposure trials undertaken on Henderson and Desecheo that showed, through use of a biomarker, 100% acceptance by trapped rats (Swinnerton & McKown, 2009; Brooke, et al., 2010). On Wake, concerns about behavioural resistance were generated after some rats in a two-choice laboratory trial undertaken on the island (Mosher, et al., 2008) were documented not eating rodent bait. Three *R. exulans* also avoided exposure during an *in situ* biomarker trial (Wegmann, et al., 2009). However, as outlined by Griffiths et al. (2014), the successful elimination of *R. tanezumi* from the atoll, the complete removal of *R. exulans* from a discrete part of the atoll (Peale Island), and the marked reduction of *R. exulans* for a period of time, are not consistent with a bait shy rat population. No pre-eradication trials to assess bait palatability were undertaken on Enderbury.

Some evidence for neophobia or rats preferring alternative foods over rodent bait was seen at the time of bait application for Enderbury and Wake. On the first night after the initial application of bait on Enderbury, rats were observed walking past rodent bait, despite it being readily available, to forage on the flowers and fruits of *Tribulus cistoides* on the island (Pierce & Kerr, 2013). Observations of rats foraging on natural foods in the presence of bait were also made on Wake (Griffiths, et al., 2014). However, it is unknown if such observations are unusual or should be considered the norm for rodent eradications, because of a lack of information.

Relative to previous site visits, signs of elevated resource availability were observed on Henderson and Enderbury islands (Cuthbert, 2012; Pierce & Kerr, 2013) at the time of project implementation. Rainfall leading up to the operations is presumed to have led to this increase (Cuthbert, 2012; Pierce & Kerr, 2013). On Henderson, three plant species, *Cyclophyllum barbatum, Myrsine hosakae* and *Eugenia reinwardtiana* were observed with more fruit than seen in previous years and the presence of a large number of recently fledged fruit doves (*Ptilinopus insularis*) indicated that a large fruiting event had occurred shortly prior to the operation (Cuthbert, 2012). On Enderbury, 10 of the 11 common plant species present were recorded as either flowering or fruiting at the time of the operation including the four dominant plants *T. cistoides, Portulaca lutea, Boerhavia albiflora* and *Sida fallax*. Higher than average rainfall prior to the unsuccessful Desecheo eradication (as evidenced by mainland weather records) may have also generated increased food availability there (Brown & Tershy, 2013). It is unknown if resources on Wake were elevated at the time of the operation, but abundant seed observed on *Casuarina* trees growing across the island at the time of the operation and high numbers of rats observed at the time of the operation correspond with this possibility.

Based on available evidence we cannot reach a definite conclusion on the role of this factor in the outcome observed in the four unsuccessful projects. However, the elevated availability of alternative resources may have compounded other factors such as rat breeding to influence project outcome.

Comparison among all eight projects

We could not separate unsuccessful projects from successful projects based on geographic location, habitat or standard climatic variables (Table 1). However, three of the unsuccessful projects were undertaken on islands significantly larger than those that were successful. Rats were also successfully removed from the smaller of the two disconnected land masses that comprise the Wake Atoll complex (Griffiths, et al., 2014). Commensal issues associated with the presence of a resident human population, a known risk factor for rodent eradications (Oppel, et al., 2011), were a significant component of the Wake project but were also present, albeit on a smaller scale, on three of the islands where rats were successfully removed suggesting these issues were not insurmountable.

Similarly, more parallels than differences were evident between successful and unsuccessful projects for the environmental variables identified by Holmes, et al. (2015) and ourselves as important to eradication success (Table 3). Elevated rainfall preceding the eradication operation differentiated three of the unsuccessful projects, Desecheo, Henderson and Enderbury. However, abundant natural food resources, as observed on Henderson, Enderbury, Desecheo and Wake at the time of project implementation, were also observed on Palmyra, the Ringgolds and Frégate where rats were successfully removed. Fruiting *Pandanus tectorius*, coconut and nesting sooty terns (*Onychoprion fuscatus*) on Palmyra, *Terminalia littoralis* fruit and coconut on the Ringgolds and coconut, multiple fruiting tree species, breeding seabirds, kitchen refuse, cultivated crops and food for livestock on Frégate all offered plentiful resources to rats. However, the level of natural food availability during project implementation relative to other times of the year for these islands is unknown. An abundance of natural resources was not documented on Birnie, where rats were successfully removed. Little flowering or fruiting by the four common plant species that are present was noted on this island at the time of the implementation (Pierce & Kerr, 2013).

Land crabs were an influential factor on all eight islands. Bait availability data provided some indication of their relative impact on each of the operations but, in the absence of crab survey data for each island, an independent assessment of relative crab population density among islands was not possible. Such data would have provided a clearer picture of the relative impact of land crabs on project success. Anecdotal observations suggest that rat numbers were high on all eight islands at the time of project implementation, but relative population densities were once again unknown. Reproduction was not investigated on the Ringgolds, but evidence indicates that rats were breeding at the time of the eradication at the other sites. On Palmyra, where rats were successfully removed, a juvenile rat was sighted and captured 28 days after the initial bait application within the island's commensal area where bait stations were being maintained. This individual was near death and an assay of its liver confirmed exposure to brodifacoum. Like the second of the two juveniles discovered on Wake after bait application, this rat also appeared malnourished. It is possible, based on observations of elevated rainfall and increased resource availability, that the intensity of rat breeding was higher on Henderson, Enderbury and possibly Desecheo than on the islands where rats were successfully removed but in the absence of data this cannot be confirmed. Two of the successful projects targeted rat populations that had previously been exposed to anticoagulants (Table 3). Rats on Palmyra, where anticoagulants had been used previously, were thought to be tolerant to brodifacoum because some individuals survived for longer than anticipated during a toxicity trial (Howald, et al., 2004), yet this project was successful.

Details for each of the eight eradication operations are presented in Table 4. All projects used a helicopter and bait spreading bucket as the principal method for bait application, utilised proven rodent bait types and applied bait with a similar swath overlap. The main difference between operations was in the amount of bait applied, which ranged between 10 and 84 kg/ha for the first application and between 6 and 79 kg/ha for the second. Difference in application rate was largely a function of decisions made by respective project teams based on an assessment of relative bait competition by land crabs for each island. While this difference was evident, there was no clear relationship between application rate and success or failure for the eight projects (Table 4). Relative to the three unsuccessful projects where monitoring of bait availability was undertaken, bait on Palmyra also disappeared rapidly but remained at higher densities beyond the seven-day observation period in coconut canopy (Berentsen, et al., 2013), a preferred habitat for rats (Wegmann, 2008). Bait persisted in all plots monitored on Frégate for 10 days after its application and bait availability would have been extended by the third application (Merton, et al., 2002) but this was not monitored. No monitoring of bait was undertaken on Birnie or the Ringgolds, but bait was reported to be widely available on both islands for the six days between the first and second applications of bait.

As with two of the failed projects, operational procedures were also modified during project implementation for two successful projects. For instance, an unplanned third application of bait was completed following the sighting of

Table 3 Environmental variables present at the time of project implementation that could have influenced the project's outcome.

	Henderson	Wake	Desecheo	Enderbury	Birnie	Palmyra	Ringgolds	Frégate
Project outcome	Failed	Failed	Failed	Failed	Succeeded	Succeeded	Succeeded	Succeeded
Hermit crabs present	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Other land crab species present	No	No	Yes	No	No	Yes	Yes	Yes
Ant species present	Yes	Yes	Yes	No	No	Yes	Yes	Yes
Permanent human population present	No	Yes	No	No	No	Yes	Yes	Yes
Rat population had been previously exposed to anticoagulants	No	Yes	No	No	No	Yes	No	Yes
Higher than anticipated rainfall preceded operation	Yes	No	Yes	Yes[a]	Unknown	No	No	No
Observations of high natural food availability immediately prior to or during project implementation	Yes	Yes	Yes	Yes	No	Yes	Yes	Yes
Seabirds nesting at time of implementation	Yes	No	No	Yes	Yes	Yes	Yes	Yes
Rat population breeding at time of project implementation	Yes	Yes	Yes	Yes	Yes	Yes	Unknown	Yes

[a]Inferred from observations of flowering and fruiting during project implementation.

a surviving rat on Frégate Island (P. Garden, pers. comm.). On Palmyra, bait was hand broadcast across a 10 ha area on Cooper Island following the discovery of the juvenile rat mentioned above (Wegmann, et al., 2012). No changes to the operational strategy were reported for the Ringgolds and Birnie projects and, as with the Enderbury project, bait application proceeded according to plan.

From our qualitative comparative analysis, we could not reach a conclusion on the role of geographic, habitat, climatic and environmental variables or operational parameters on the relative outcome of the eight projects reviewed. The two variables that best differentiated unsuccessful from successful projects were elevated rainfall preceding the operation and island size.

DISCUSSION

Reasons for project failure

Based on the robust design of the eradication operations reviewed and GIS maps of bait coverage, we conclude that bait was made available to all rats actively foraging at the time of the operation for the Henderson, Enderbury and Desecheo projects. We cannot be as confident of this for Wake, despite one rat species being successfully eradicated, because the more complex operational strategy employed there coupled with competitive exclusion may have led to functional gaps in bait availability (Griffiths, et al., 2014). Notwithstanding the greater risk on Wake, some individuals within the rat population were not actively foraging at the time of bait application on all four islands

Table 4 Key elements of operational design for the eight projects.

	Henderson	Wake	Desecheo	Enderbury	Birnie	Palmyra	Ringgolds	Frégate
Project outcome	Failed	Failed	Failed	Failed	Succeeded	Succeeded	Succeeded	Succeeded
Bait type[a]	20R	25W	25D	20R	20R	25W	20R	20R
Application rate 1st/2nd/3rd bait applications (kg/ha)[b]	10/6[c]	18/9	19/10	22/17	25/25	84/79	16/11	14/9/12
Mean total bait application rate (kg/ha)	17.4	27.7	29	38.4	50	165	27	35
Percentage swath overlap per application	50/25	50/50	50/50	50/25	50/25	50/50	50/50	50/50/50
Area of plot/transect used to sample bait availability	~270 m^2	25 m^2	25 m^2	NA	NA	2.49 m^2	NA	10 m^2
Number of days that bait remained available in all sampled plots/transects after 1st application	25+	3	2	6[d]	6[d]	1[e]	10[d]	10[f]
Number of days that bait remained available in all sampled plots/transects after 2nd application	20+	5	1	Unknown	Unknown	1[d]	Unknown	5[e]
Number of days between applications	5	9	10	5	6	6	10	5/24
Lower 99% CI of the T-statistic for bait available four days after the 1st application (kg/ha)	1.93	6.33	0.25	Unknown	Unknown	19.16	Unknown	-3.32
Areas excluded from aerial bait application	No	Yes	No	No	No	Yes	No	Yes

[a] Bait pellet types listed are 20R – Pestoff 20R rodent bait produced by Animal Control Products, Wanganui, New Zealand; 25W – Brodifacoum-25W Conservation manufactured by Bell Laboratories, Wisconsin, USA; 25D – Brodifacoum-25D Conservation manufactured by Bell Laboratories, Wisconsin, USA.
[b] Areas subject to hand broadcast were applied at the same rates as for aerial application.
[c] Rates listed here were used across the island's plateau which amounted to 95% of the island's area. Higher bait application rates were applied in the vicinity of the island's beaches where hermit crabs were most numerous.
[d] No monitoring of bait availability was undertaken and figures are inferred from ad hoc observations. The project team left the islands after the number of days listed.
[e] The figure reported is for terrestrial plots: bait persisted longer in coconut palm canopy.
[f] No monitoring was undertaken after the 3rd application which would have extended the number of days that bait was available.

where rats survived. Rats were breeding on Henderson, Wake, Desecheo and Enderbury at the time of project implementation and evidence suggests that brodifacoum is not passed on in sufficient amounts via lactation to cause mortality (Milne, et al., 2001; Gabriel, et al., 2012). Pups in the nest at the time of bait application were therefore effectively isolated for the period they were dependent on the lactating female.

Such a scenario has been previously considered by eradication practitioners as a theoretical possibility (e.g. Broome, et al., 2011), but the discovery of juvenile rats on both Palmyra and Wake after bait application validates it as a very real concern for tropical island rodent eradications, where breeding cycles cannot be predicted with certainty. Weaning times reported for *R. exulans* (Wirtz, 1972; Tobin, 1994), *R. rattus* (Cowan, 1981; Yom-Tov, 1985) and *R. norvegicus* (King, 1990) range from 21 to 28 days, much longer than the period over which bait is typically available for tropical rat eradication projects including a number of the projects reviewed here.

It has generally been accepted that breeding females, like other individuals within a rat population, would access and ingest a lethal dose of bait and die within a few days of bait application. However, there are reasons to be sceptical that this will always occur. Home ranges for female rats (e.g. *R. rattus*) can be significantly smaller than those of males (Pryde, et al., 2005) and, as has been documented for house mice (*Mus musculus*) (Krebs, et al., 1995), lactating female rats may have constricted foraging ranges. Changes in dietary requirements by rats can also occur during lactation (Leshner, et al., 1972) potentially affecting bait palatability. The maximum period of time documented for mortality following the ingestion of a lethal dose of brodifacoum is 21 days, from a trial conducted with captive *R. rattus* on Palmyra (Howald, et al., 2004). Any of these traits could increase the chance of juveniles emerging after bait is no longer readily available on an island and, with natural food abundant on many tropical islands, these individuals have an enhanced probability of survival.

The fact that bait remained available in all transects monitored on Henderson for more than 25 days challenges the premise of juvenile survival as a potential cause of failure for this project. However, as described by Pott, et al. (2015), a different method of monitoring bait availability was used for this project and monitoring was confined to one small corner of the island (Brooke, et al., 2011) so comparison with other projects is difficult. It is also possible that bait disappeared more rapidly in unmonitored parts of the island. Bait was applied at a lower rate on Henderson than in the other projects reviewed and this, coupled with the island's complicated 'makatea' or uplifted coral substrate, may have reduced the rate at which breeding female rats encountered bait.

Rats were confirmed as breeding during project implementation on Birnie, Palmyra and Frégate where rats were successfully removed. Why did these projects succeed? Some explanations can be tendered but, without additional evidence, cannot be verified. For example, the high bait application rate used on Palmyra likely ensured that breeding female rats rapidly encountered bait plus bait in the coconut palm canopy, a known nesting habitat for female rats, was accessible for a longer period. On Frégate, a third bait application extended the period of bait availability out beyond 24 days and less competition by hermit crabs and lower rat densities on Birnie may have increased bait availability there. It is also plausible that in the absence of the supplementary interventions made on Palmyra and Frégate, these projects could also have failed. Insufficient information is available to form similar conclusions for the Ringgolds project.

We were able to rule out inadequate bait toxicity and resistance as factors for the survival of rats on Henderson, Enderbury and Desecheo and the persistence of *R. exulans* on Wake. Neither has been documented for any of the 490 attempted higher latitude rat eradications and we know of no viable hypothesis that would predict a greater incidence of resistance in rats or insufficient bait toxicity for tropical rat eradication projects. For the unsuccessful projects we reviewed we reject bait toxicity as a factor based on: factory test results demonstrating that the bait used on Henderson, Wake and Desecheo contained a sufficient concentration of brodifacoum; the marked reduction in rat numbers on all three islands; and the fact that *R. tanezumi* was successfully removed from Wake. The bait applied on Enderbury was produced as part of the same consignment as that was used successfully to remove rats from Birnie.

Similarly, we found no evidence to support anticoagulant resistance as a factor in the unsuccessful outcome seen on Henderson, Wake, Desecheo and Enderbury. Rat populations on Henderson, Enderbury and Desecheo had no prior exposure to anticoagulants and the successful eradication of *R. tanezumi* from Wake, the removal of *R. exulans* from part of the atoll, and the reduction of *R. exulans* to undetectable levels elsewhere is at odds with the levels of survivorship reported for rodent populations in which practical resistance has been documented (e.g. Drummond & Rennison, 1973; Greaves, et al., 1982). Most importantly, 'practical' resistance to brodifacoum that might have caused the failure of these projects, has never been encountered, even at sites where anticoagulants have been used repeatedly for long periods of time (Buckle & Prescott, 2012). Increased tolerance to brodifacoum has been detected in some locations (Buckle & Prescott, 2012) and may have been present on the three islands where anticoagulants had been used previously. However, rats were successfully removed from two of these islands including Palmyra where a bait toxicity trial had suggested the possibility of anticoagulant tolerance.

Conflicting evidence meant we could not rule out the possibility that some rats avoided rodent bait in preference for natural foods. Certainly, for all four unsuccessful projects, natural food was readily available to rats at the time of project implementation. Observations of rats foraging on natural foods after bait application on Enderbury and Wake lend weight to this hypothesis. However, this may simply have been a function of neophobia, as described by Barnett (1956), and not necessarily active bait avoidance. We are unaware of similar observations from other projects, but this is likely a result of insufficient observational effort. The discovery of recently weaned juvenile rats on Palmyra and Wake, more than four weeks after bait application, suggests that some individuals, in this case lactating female rats, may have avoided bait for a period. Rats detected on Desecheo and Fregate after bait application also point to this possibility. Set against this evidence is the fact that natural food was also available on the islands where rats were successfully removed, and signs of malnutrition and early weaning of the juveniles found on Palmyra and Wake suggest that the females producing these pups died because they consumed bait. A necropsy verified bait consumption for the Desecheo rat and the Frégate project was ultimately successful, confirming all individuals there were eventually exposed. The successful removal of the more dominant rat species on Wake also perhaps points to bait availability rather than bait palatability as the more important influence.

In summary, it is unknown if the elevated availability of natural resources on Henderson, Enderbury, Wake and Desecheo led to bait avoidance, but the possibility cannot be discounted. Increased natural food availability may have also compounded other factors influencing project success such as the intensity of rat breeding. Given the unpredictability of resource availability within many tropical island ecosystems this will need to be an important consideration for future rat eradication projects.

Comparative analysis

We could not separate unsuccessful projects from successful projects based on habitat or standard climatic variables. However, three of the unsuccessful projects were undertaken on islands significantly larger than those that were successful and both rat species present on Wake were removed from Peale Island, one of the two land units that make up the Wake Atoll complex. This is consistent with the trend identified by Holmes, et al. (2015) of an increasing failure rate for larger islands. It is therefore possible that the outcomes observed on Henderson, Wake and Enderbury were simply a consequence of biogeographic theory. Larger populations on the bigger islands increased the chance that some individuals would avoid bait or that some breeding females would survive for long enough to wean juveniles when bait was no longer readily available. No threshold for island size has yet been identified for rodent eradications undertaken using the methodology reviewed in this paper. However, the threshold may be smaller for tropical islands because of increased availability of natural resources, higher rat population densities and the likelihood that a proportion of the population will be breeding during project implementation.

Rainfall is closely linked to ecosystem productivity on tropical islands (Murphy & Lugo, 1986) and elevated rainfall levels preceding the eradication were associated with three of the unsuccessful projects reviewed. Variability in rainfall was also found by Holmes, et al. (2015) to be correlated with failure for tropical rat eradications. However, as discussed above, we could not fully resolve whether rainfall contributed to an increased risk of failure for these projects because palatability of rodent bait was reduced in the presence of increased natural food availability or greater reproductive activity within the targeted rat populations led to juveniles surviving the eradication attempt.

In summary, although our review of eight tropical rodent eradications could not discern the relative importance of bait availability or bait palatability in the outcome of the four unsuccessful projects, it suggests that both are important to consider in the planning of future rodent eradications on tropical islands. In the absence of a more palatable bait type, we recommend greater emphasis is placed on operational design for future tropical island rodent eradications. As recommended by Keitt, et al. (2015), projects should aim to ensure that bait is readily available within all rat territories for a period of time that allows all individuals within the population to encounter bait. Even though the projects we reviewed were well documented, our analysis was limited by a lack of consistency in data collection. Until more is known about the mechanisms that promote survival during a rat eradication attempt, future monitoring of eradication projects undertaken on tropical islands should aim to document as many of the variables discussed in this paper as possible to determine the relative importance of these factors in the project's fate. Standardisation of monitoring protocols, as promoted by Keitt, et al. (2015) and Pott, et al. (2015), should also be instigated.

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REFERENCES

Abrahamsson, P., Warsta, J., Siponen, M.T. and Ronkainen, J. (2003). 'New Directions on Agile Methods: A Comparative Analysis'. In: *Proceedings of the 25th International Conference on Software Engineering*, pp. 244–254. Washington, DC, USA: IEEE Computer Society.

Allen, T.D., Poteet, M.L. and Burroughs, S.M. (1997). 'The mentor's perspective: A qualitative inquiry and future research agenda'. *Journal of Vocational Behavior* 51(1): 70–89.

Bailey, C., Fisher, P. and Eason, C.T. (2005). 'Assessing Anticoagulant Resistance in Rats and Coagulation Effects in Birds Using Small-volume Blood Samples'. *Science for Conservation* 249. Wellington, New Zealand: Department of Conservation.

Barnett, S.A. (1956). 'Behaviour components in the feeding of wild and laboratory rats'. *Behaviour* 9: 24–43.

Bellingham, P.J., Towns, D.R., Cameron, E.K., Davis, J.J., Wardle, D.A., Wilmshurst, J.M. and Mulder, C.P.H. (2010). 'New Zealand island restoration: Seabirds, predators, and the importance of history'. *New Zealand Journal of Ecology* 34(1): 115–136.

Bennett, A. and Elman, C. (2006). 'Qualitative research: Recent developments in case study methods'. *Annual Review of Political Science* 9: 455–476.

Berentsen, A.R., Pitt, W.C., Eisemann, J.D. and Engeman, R.M. (2013). 'Longevity of rodenticide bait pellets in a tropical environment following a rat eradication program'. *Environmental Science and Pollution Research* 21: 2283-2288.

Brooke, M.d.L., Jones, P.J., Vickery, J.A. and Waldren, S. (1996). 'Seasonal patterns of leaf growth and loss, flowering and fruiting on a sub-tropical central Pacific island.'. *Biotropica* 28: 164–179.

Brooke, M., Cuthbert, R., Henricson, A., Torr, N., Warren, P. and O'Keefe, S. (2010). *Towards Rat Eradication on Henderson Island: Fieldwork Report, August–September 2009*. Unpublished Report. Sandy, Bedfordshire, UK: Royal Society for the Protection of Birds.

Brooke, M.d.L., Harrison, G. and Cuthbert, R.J. (2011). *The Capture, Husbandry and Captive Breeding of Henderson Rails During the Henderson Island Rat Eradication and the Impact of the Project on Native Landbirds*. Unpublished Report. Sandy, Bedfordshire, UK: Royal Society for the Protection of Birds.

Broome, K.G., Brown, D., Cox, A., Cromarty, P., McClelland, P., Golding, C., Griffiths, R. and Bell, P. (2011). *Current Agreed Best Practice for Rodent Eradication – Aerial Broadcasting Poison Bait (Version 2.2)*. Unpublished Report. Wellington, New Zealand: Department of Conservation.

Brown, D., Pitt, W.C. and Tershy, B.R. (2013). *A Review of the Wake Island Rat Eradication Project.* Unpublished Report. Honolulu, Hawaii: US Fish and Wildlife Service.

Brown, D. and Tershy, B.R. (2013). *A Review of the Desecheo Island Rat Eradication Project.* Unpublished Report. Cabo Rojo, Puerto Rico: US Fish and Wildlife Service.

Buckle, A. and Prescott, C. (2012). *The Current Status of Anticoagulant Resistance in Rats and Mice in the UK*. Unpublished Report. Reading, Berkshire, UK: Rodenticide Resistance Action Group of the United Kingdom, Vertebrate Pests Unit, University of Reading.

Butler, D., Tye, A., Wylie, M. and Tipama'a, F. (2011). 'Eradicating Pacific rats (*Rattus exulans*) from Nu'utele and Nu'ulua Islands, Samoa–Some of the Challenges of Operating in the Tropical Pacific'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 407–412. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Cowan, P. (1981). 'Early growth and development of roof rats, *Rattus rattus* L.'. *Mammalia* 45(2): 239–250.

Cromarty, P.L., Broome, K.G., Cox, A., Empson, R.A., Hutchinson, W.M. and McFadden, I. (2002). 'Eradication planning for invasive alien animal species on islands – the approach developed by the New Zealand Department of Conservation'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 85–91. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Cuthbert, R. (2012). *Notes on Flowering and Evidence of Breeding Birds on Henderson July–August 2011*. Unpublished Report. Sandy, Bedfordshire, UK: Royal Society for the Protection of Birds.

DIISE. (2017). 'The Database of Island Invasive Species Eradications, Developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group. University of Auckland and Landcare Research, New Zealand.' <http://diise.islandconservation.org>. Accessed 30 September 2017.

Drummond, D. and Rennison, B.D. (1973). 'The detection of rodent resistance to anticoagulants'. *Bulletin of The World Health Organization* 48(2): 239–242.

Gabriel, M.W., Woods, L.W., Poppenga, R., Sweitzer, R.A., Thompson, C., Matthews, S.M., Higley, J.M., Keller, S.M., Purcell, K., Barrett, R.H., Wengert, G.M., Sacks, B.N. and Clifford, D.L. (2012). 'Anticoagulant rodenticides on our public and community lands: Spatial distribution of exposure and poisoning of a rare forest carnivore.'. *PLOS ONE* 7: e40163.

George, A.L. and Bennett, A. (2005). *Case Studies and Theory Development in the Social Sciences*. Cambridge: Cambridge University Press.

Greaves, J., Shepherd, D. and Gill, J. (1982). 'An investigation of difenacoum resistance in Norway rat populations in Hampshire'. *Annals of Applied Biology* 100(3): 581–587.

Griffiths, R., Brown, D., Tershy, B., Pitt, W., Wegmann, A., Hanson, C., Moran, M., Rex, K., White, S., Flint, B., Keitt, B., Holmes, N., Howald, G. and Torr, N. (2014). 'The Wake Island Rodent Eradication – Part Success, Part Failure, but Wholly Instructive'. In: R.M. Timm (ed.) *Proceedings of the 26th Vertebrate Pest Conference*, pp. 101–111. Davis, CA: University of California.

Holmes, N.D., Griffiths, R., Pott, M., Alifano, A., Will, D., Wegmann, A.S. and Russell, J.C. (2015). 'Factors associated with rodent eradication failure'. *Biological Conservation* 185: 8–16.

Howald, G., Samaniego, A., Buckelew, S., McClelland, P., Keitt, B., Wegmann, A., Pitt, W.C., Vice, D.S., Campbell, E., Swift, K. and Barclay, S. (2004). *Palmyra Atoll Rat Eradication Assessment Trip Report, August 2004*. Unpublished Report. Santa Cruz, California, USA: Island Conservation.

Howald, G., Donlan, C.J., Galván, J.P., Russell, J., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21(5): 1258–1268.

Keitt, B., Campbell, K., Saunders, A., Clout, M., Wang, Y., Heinz, R., Newton, K. and Tershy, B. (2011). 'The global islands invasive vertebrate eradication database: a tool to improve and facilitate restoration of island ecosystems'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 74–77. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

Kier, G., Kreft, H., Lee, T.M., Jetz, W., Ibisch, P.L., Nowicki, C., Mutke, J. and Barthlott, W. (2009). 'A global assessment of endemism and species richness across island and mainland regions'. *Proceedings of the National Academy of Sciences* 106(23): 9322–9327.

King, C.M. (ed.) (1990). *The Handbook of New Zealand Mammals.* Oxford: Oxford University Press.

Krebs, C.J., Kenney, A.J. and Singleton, G.R. (1995). 'Movements of feral house mice in agricultural landscapes'. *Australian Journal of Zoology* 43(3): 293–302.

Kurle, C.M., Croll, D.A. and Tershy, B.R. (2008). 'Introduced rats indirectly change marine rocky intertidal communities from algae- to invertebrate-dominated'. *Proceedings of the National Academy of Sciences* 105(10): 3800–3804.

Leshner, A.I., Siegel, H.I. and Collier, G. (1972). 'Dietary self-selection by pregnant and lactating rats'. *Physiology & Behavior* 8(1): 151–154.

Lund, M. (1984). 'Resistance to the Second-generation Anticoagulant Rodenticides'. In: D.O. Clark (ed.) *Proceedings of the 11th Vertebrate Pest Conference*, pp. 89–94. Davis, CA: University of California.

Merton, D., Climo, G., Laboudallon, V., Robert, S. and Mander, C. (2002). 'Alien mammal eradication and quarantine on inhabited islands in the Seychelles'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species.* pp. 182–198. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Milne, L., Wright, G., Eason, C. and O'Connor, C. (2001). 'Human Exposure to Poisons Through Contaminated Food'. In: Vertebrate Pest Committee (ed.) *Proceedings of the 12th Australasian Vertebrate Pest Conference*, pp. 141–145. Melbourne, Australia: Department of Natural Resources and Environment.

Mosher, S., Hebshi, A., Swift, K., Dunlevy, P., Vice, D., Wegmann, A., Jacobs, B., Thomas, B. and Gilardi, J. (2008). *Rat Eradication Feasibility Study: 29 September–27 October 2007*. Unpublished Report. Hickam Air Force Base: Deptartment of Defense.

Murphy, P.G. and Lugo, A.E. (1986). 'Ecology of tropical dry forest'. *Annual Review of Ecology and Systematics* 17(1): 67–88.

Oppel, S., Beaven, B.M., Bolton, M., Vickery, J. and Bodey, T.W. (2011). 'Eradication of invasive mammals on islands inhabited by humans and domestic animals'. *Conservation Biology* 25(2): 232–240.

Pierce, R.J. and Kerr, V. (2013). *Preliminary Report on a Biota Survey of the Phoenix Islands Atolls, Kiribati, May 2013*. Unpublished report. Speewah, QLD, Australia: Eco Oceania Pty.

Pott, M., Wegmann, A., Griffiths, R., Samaniego-Herrera, A., Cuthbert, R., Brooke, M., Pitt, W., Berentsen, A., Holmes, N., Howald, G. and Russell, J.C. (2015). 'Stacking the deck: Assessing bait availability before rodent eradications to inform bait application rates'. *Biological Conservation* 185: 27–35.

Pryde, M., Dilks, P. and Fraser, I. (2005). 'The home range of ship rats (*Rattus rattus*) in beech forest in the Eglinton Valley, Fiordland, New Zealand: A pilot study'. *New Zealand Journal of Zoology* 32(3): 139–142.

Ragin, C.C. (1989). *The Comparative Method: Moving Beyond Qualitative and Quantitative Strategies*. Berkeley, CA: University of California Press.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Shiels, A.B. (2010). 'Ecology and Impacts of Introduced Rodents (*Rattus* spp. and *Mus musculus*) in the Hawaiian Islands'. PhD thesis. Honolulu, HI: The University of Hawaii.

Spencer, T. (1995). 'The Pitcairn Islands, South Pacific Ocean: Plate tectonic and climatic contexts'. *Biological Journal of the Linnean Society* 56: 13–42.

Swinnerton, K. and McKown, M. (2009). *Scoping Trip to Desecheo Island, Puerto Rico, to Evaluate Feasibility of Rodent Eradication, and Implement Baseline Biological Surveys: February 12–26, 2009*. Unpublished Report. Santa Cruz, California, USA: Island Conservation.

Tobin, M.E. (1994). 'Polynesian Rats'. In: S.E. Hygnstrom, R.M. Timm and G.E. Larson (eds.) *Prevention and Control of Wildlife Damage*, pp. 121–124. Lincoln, NE: University of Nebraska.

Torr, N. and Brown, D. (2012). *Henderson Island Restoration Project Post-Operational Report*. Unpublished Report. Sandy, Bedfordshire, UK: Royal Society for the Protection of Birds.

Towns, D., Atkinson, I. and Daugherty, C. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?'. *Biological Invasions* 8: 863–891.

USFWS. (2011). *Palmyra Atoll National Wildlife Refuge Rat Eradication Project Final Environmental Impact Statement*. Honolulu, HI: US Fish & Wildlife Service.

Wegmann, A. (2008). 'Home Range and Habitat Use Observations of Introduced Rats (*Rattus rattus*) and Native Land Crabs (Coenobitidae and Gecarcinidae) at Palmyra Atoll'. PhD thesis. Honolulu, HI: University of Hawaii.

Wegmann, A., Helm, J., Torr, N., Golding, C. and Howald, G. (2009). *Wake Atoll Rat Eradcation Strategic Plan: Biomarker Study Trip Report (27 February to 27 March, 2009).* Unpublished Report. Kelowna, Canada: Island Conservation.

Wegmann, A., Flint, E., White, S., Fox, M., Howald, G., McClelland, P., Alifano, A. and Griffiths, R. (2012). 'Pushing the Envelope in Paradise: A Novel Approach to Rat Eradication at Palmyra Atoll'. In: R.M. Timm (ed.) *Proceedings of the 25th Vertebrate Pest Conference*, pp. 48–53. Davis, CA: University of California.

Wirtz, W.O.I. (1972). 'Population ecology of the Polynesian rat, *Rattus exulans*, on Kure Atoll, Hawaii'. *Pacific Science* 26: 433–461.

Yom-Tov, Y. (1985). 'The reproductive rates of Australian rodents'. *Oecologia* 66(2): 250–255.

Eradication of mice from Antipodes Island, New Zealand

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Abstract In winter 2016, the New Zealand Department of Conservation (DOC) eradicated mice (*Mus musculus*) from the Antipodes Islands located at 49°S 178°E, 760 km south-east of New Zealand's South Island. Mice were the only mammalian pest species present. They have extensively impacted the abundance and survival of invertebrates, with likely secondary impacts on endemic terrestrial birds and nesting seabird fauna. Public-private partnerships with DOC instigated the project and provided essential financial support. Baseline scientific data for operational planning and outcome monitoring were collected by a research expedition in July 2013 and project planning began in 2014. At the time of writing, this is the largest eradication of mice undertaken where mice are the sole mammalian pest species. Logistical challenges were complicated by a broad range of regulatory obligations. The expedition-style project used a ship to deliver a team and equipment to Antipodes Island where they established camp and remained until the completion of baiting. Bait spread was completed incrementally as weather allowed, comprehensively covering the islands in two separate treatments between 18 June 2016 and 12 July 2016. The last sign of mice was detected 20 days after the first application of bait and the eradication of mice was confirmed by monitoring in late summer 2018. Public engagement was achieved with regular operational updates across multiple platforms and positive media coverage. Non-toxic bait trials accurately predicted some by-kill of pipit (*Anthus novaeseelandiae steindachneri*) but did not anticipate poisoning of some Antipodes parakeet (*Cyanoramphus unicolor*) and Reischek's parakeet (*Cyanoramphus hochstetteri*). Known pest-free islands were not baited, providing refuge for land birds to mitigate the risk. Fledging success of Antipodean albatross *(Diomedea antipodensis antipodensis)* chicks was not impacted by the operation and those species that were affected had recovered by summer 2018.

Keywords: brodifacoum, house mouse, Million Dollar Mouse, *Mus musculus*, non-target impacts, subantarctic

INTRODUCTION

The Antipodes Islands group (2,100 ha) is in New Zealand's Subantarctic Islands region and was gazetted as a Nature Reserve in 1978 and a World Heritage site in 1998. The group comprises six islands and one islet located in the Southern Ocean, at 49°41'S, 178°48'E, 760 km from New Zealand's South Island (Fig. 1). The islands are uninhabited and administered by New Zealand's Department of Conservation (DOC). House mouse (*Mus musculus*) was the only mammalian pest species present and known only on the main island, Antipodes Island (2,012 ha).

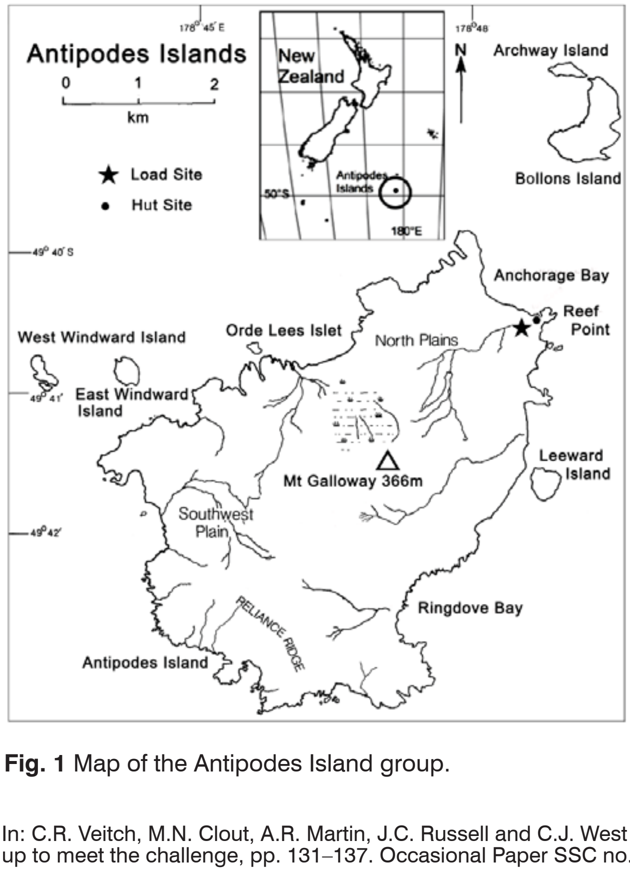

Fig. 1 Map of the Antipodes Island group.

The Antipodes were discovered in 1800 and sealers arrived by 1804 (Taylor, 2006). A small shelter (castaway depot) was built in 1886 to support shipwreck survivors. It was resupplied periodically until 1927 (Taylor, 2006). Mice were first recorded on Antipodes Island in 1907 but probably arrived earlier (McIntosh, 2001) with sealers or as the result of a foreign shipwreck (*Spirit of the Dawn*) in 1896 (Taylor, 2006). DNA studies of the mouse population identified a mtDNA haplotype also found in Spain but not elsewhere in New Zealand (Searle, et al., 2009.).

Mice were abundant; their density has been recorded as high as 147/ha in the coastal zone (Russell, 2012). They have had a significant detrimental impact on the endemic, rare and threatened animal species. Invertebrates have been severely depleted. Mice are responsible for the general absence of large beetles and the extirpation of at least two taxa: *Loxomerus* n.sp. and *Tormissus guanicola* (Marris, 2000); and several large ground dwelling species are severely restricted in distribution (Marris, 2000; Russell, 2012). Mice also compete with the four endemic land birds and have suppressed at least two species of burrowing seabirds: black-bellied storm petrels (*Fregetta tropica*) and subantarctic little shearwater (*Puffinus elegans)* (Imber, et al., 2005).

The aim of the project was to eradicate mice from the archipelago to halt the degradation of biodiversity and allow native species to recover and flourish. Eradicating mice would also protect potentially vulnerable species, for example the nationally critical Antipodean albatross *(Diomedea antipodensis antipodensis)*, from potential attacks as recorded on Gough Island and Marion Island (Davies, et al., 2015; Dilley, et al., 2016).

The site has good ongoing biosecurity integrity. The islands are remote and isolated, landing requires a permit and the coastline is generally inaccessible, with no harbour. In 2012, DOC partnered with the Morgan Foundation to initiate the project. The Morgan Foundation fronted a highly publicised fundraising campaign "Million Dollar Mouse" (MDM), and matched public donations dollar for dollar. Additional funding came from DOC and other partners, WWF New Zealand and Island Conservation.

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MATERIALS AND METHODS

Planning

DOC planned and managed the operation from its Murihiku Office in Invercargill. Planning started in February 2014, with the employment of a full-time project manager, and took two and half years with a core team of two increasing to four in the last six months. A much larger DOC team supported pre-departure preparations. The Department's Island Eradication Advisory Group (IEAG) was engaged from the start, providing technical oversight. Eradication design was based on agreed best practice (Broome, et al., 2019). DOC's Animal Pest Framework and elements of DOC's Project Management Framework (PMF) provided the tools to manage the project.

Procurement

Helicopter and shipping services were sourced using government processes. In early 2016, DOC contracted the services of the M.V. *Norfolk Guardian*, a coastal freighter flagged in Kingdom of Tonga and a yacht, S.V. *Evohe*, to supplement passenger transport.

An experienced eradication pilot was engaged as a consultant to progress planning while a helicopter supplier was being sought. Following consultation with potential suppliers, a temporary hangar (16 m × 12 m × 5. 6 m high) and a large wooden platform (29 m × 13.8 m) incorporating a helipad were added to the planned infrastructure to help protect helicopters and other sensitive equipment from the elements. The hangar was fastened to the wooden platform and the whole structure anchored with 38 t of water ballast positioned around the base of the hangar frames in palletised 1,000 l cage tanks (Intermediate Bulk Containers). The anchoring system was designed for easy installation and extraction and to withstand winds of up to 190 km/hr.

A specialist company "Island Aerial Solutions Ltd" (IASL) was contracted to supply helicopter services and a helicopter engineer. Three helicopters were taken to the island, two AS350 Squirrels (1 × B2 and 1 × FX2) and one Robinson R44. The R44 provided contingency for marine search and rescue, enabling baiting to continue using one AS350 if the other became inoperative.

Preparations

The hangar construction was trialled in a large warehouse prior to departure. The International Chamber of Shipping Guide to Helicopter/Ship Operations (2008) was used in the development of protocols for managing shipborne helicopter operations. Ship preparations included establishing a helipad and upgrading emergency response capabilities onboard. Two months before departure, interaction trials allowed pilots to practice shipborne helicopter operations and familiarise the ship's crew. Two methods were also trialled for loading helicopters onto the ship and baiting systems were tested during the same period. Bucket calibration was done by sowing non-toxic bait across a line of marked quadrants (5 m × 10 m) extending 65 m perpendicularly from each side of a flight line over tarmac. Baits were counted in every quadrat to determine "usable swath width" – the distance to which bait is reliably spread at or above the desired rate.

An experienced operational team was selected, with additional skills and experience including engineering and mechanical repairs, a recovery doctor with extensive patient extraction and remote emergency medicine experience, biodiversity monitoring, bait bucket mechanics, technical eradication knowledge, remote construction, digger driving and rigging and receiving external helicopter loads.

Biosecurity was a significant part of preparations, and actions were coordinated with a biosecurity plan. A dedicated DOC team quarantined equipment and supplies arriving from all around New Zealand. Quarantined items were generally wrapped in plastic or sealed in plywood boxes (pods). Pest detection and prevention devices, including inked tracking cards in tunnels, insect traps, and rodent bait stations, were in place at the ports of departure and facilities where equipment and supplies were stored. The cargo ship's holds were fumigated for insects. Transport vessels required a certified clean hull to travel to the island. A dive inspection of the *Norfolk Guardian* discovered biofouling on its hull and the invasive organism Mediterranean fan worm (*Sabella spallanzanii*) in the seachests. A hull clean and treatments of the seachests were completed and inspected before each voyage to the island.

Animal Control Products (ACP now trading as Orillion) based in Whanganui, New Zealand, produced 65.5 t of Pestoff® 20R Rodent Bait containing 20 ppm brodifacoum between 21 April 2016 and 3 May 2016. ACP analysed samples from each 500 kg batch of bait, measuring toxicity using Liquid Chromatography Mass Spectrometry with a detection limit of 1×10^{-5}% (0.1ppm). The agreed acceptable range was 16 ppm to 24 ppm brodifacoum by weight.

The bait was packed in four-walled paper bags each containing 25 kg of bait and transported and stored on Antipodes Island in large plywood boxes (pods) portable by forklift and helicopter. The maximum safe load capability of the helicopters determined the size of the pods (each contained 28 bags of bait and weighed a total of 805 kg). The weatherproof pods included a large plastic liner to protect bait against water ingress.

On 23 May 2016, the *Evohe* departed Dunedin, New Zealand, for Antipodes Island with 12 of the project team onboard. On 25 May 2016, the *Norfolk Guardian* departed Timaru, New Zealand with seven project team members, three helicopters, bait in 94 pods, 30 t of jet fuel and 20 t of sundry equipment and supplies. Two 1.6 tonne diggers were taken to the island to prepare a level site for the helicopter hangar. A satellite dish was installed, providing a fast internet connection. The *Evohe* remained at the island while the cargo ship was present, transferring personnel between ship and shore, and ready to respond in case of an incident over water during helicopter unloading of the ship.

Poison baiting

Bait uptake trials were conducted on Antipodes Island in winter 2013 to assess the palatability of the proposed bait to mice and the potential risks to non-target species. The trial used a non-toxic version of Pestoff® 20R Rodent Bait with the biotracer pyranine added. Baits were spread by hand over 6 ha at 16 kg/ha. Subsequently, mice were captured in a grid of Longworth live capture traps and land birds were captured with hand nets. Captured individuals were inspected for signs of bait consumption using a UV light. Observations of birds interacting with baits were also recorded. Bird faeces were collected opportunistically along a transect and inspected under UV light. Faecal samples were assigned to a species by visual inspection or by DNA analysis for a subset of samples that tested positive for pyranine (Elliott, et al., 2015).

A boundary flight recorded the treatment area as 2,114 ha before baiting commenced. The boundary was flown again more tightly before treatment two, recording the area as 2,075 ha. An advisory team (technical advisor, chief pilot and assistant project manager) assisted the project manager with finalising the load site location and layout, and daily assessment of conditions for baiting. AS350 helicopters, directed by Tracmap GPS systems, spread

65.5 t of 2 g Pestoff® 20R Rodent Bait from underslung bait buckets to complete two comprehensive treatments. The nominal application rate was 16 kg/ha for treatment one and 8 kg/ha for treatment two. A minimum interval of 14 days between treatments was preferred, to increase the likelihood of bait availability for emergent young if mice were breeding. Parallel flight lines were set at 45 m apart for a usable swath width of 90 m, giving 50% overlap of baiting swaths to minimise the risk of gaps. During each treatment, additional bait was applied to the coastline, steep slopes (50° to 70°), cliffs (greater than 70°) and other areas of concern to the pilots or identified by geospatial analysis as having potentially insufficient coverage. An observer in the back of the helicopter monitored distribution of bait on cliff baiting flights, which were undertaken at about 40 metre vertical increments.

Bait was made available inside storage containers and the interior and sub floor spaces of buildings by hand spreading or placing baits in bait stations. A bait station comprised a numbered shallow clear petri dish with ten Pestoff® 20R Rodent Bait pellets. These were placed in each compartment or room of a structure and checked daily. A total of 72 bait stations were placed in structures on 18 June. Baits were thrown by hand to achieve coverage of approximately four bait pellets per square metre under the hut and Castaway Depot and in the open wastewater drain. Toilet pits were checked daily and a handful of baits were scattered down each pit as required to maintain availability to mice. Holes were drilled in the floor of the helipad and hangar to access the subfloor space, and baits dropped through. Mouse activity was monitored around the accommodation area using inked tracking cards secured in tunnels (tracking tunnels) and baited with Pestoff® 20R Rodent Baits; and three trail cameras focused on bait stations under the hut and Castaway Depot. Approximately 4 kg of bait was used for structure baiting.

West Windward Island (7.0 ha) and East Windward Island (8.5 ha) were not baited during the first treatment as it was unknown if mice were present. These islands were monitored for mice between treatments using ten inked tracking cards baited with peanut butter and placed in tunnels (tracking tunnels) for 12 nights. Bollons Island (52.6 ha) was believed to be mouse-free prior to the operation but six tracking tunnels were installed between bait treatments for 12 nights and baited with peanut butter to provide further confidence in its status.

Monitoring to determine if mice had been eradicated occurred in late summer 2018, approximately 18 months after the baiting operation. By this time, a surviving mouse population should have recovered to detectable levels. Late summer was chosen as any breeding would have peaked and juveniles would have been present. Monitoring for mice was undertaken using 280 inked tracking cards in tunnels baited with peanut butter and distributed along 28 transect lines. Each transect comprised 10 tracking tunnels spaced 200 m apart. The transects were distributed extensively across Antipodes Island. They were placed in all habitat types, particularly in areas where mice had previously been in high abundance (e.g. near penguin colonies) and adjacent to inaccessible terrain. Tracking cards were checked and replaced approximately every five days for a period of three weeks. Supplementing this, two rodent detection dogs and their handlers searched the island for mice between 21 February and 15 March 2018. The dogs searched in accessible areas across the plateau and southern coast.

Non-target species

A non-target species technical advisory group recommended a strategy for managing risks to native species that did not include captive management but relied on natural populations outside of the treatment area. This strategy became part of the application to DOC, as administrators of the site, for consent to spread bait. Three of the four endemic land bird taxa were considered at risk from either primary or secondary poisoning. Bollons Island (52.6 ha) and Archway Island (6.2 ha) were excluded from the treatment area during planning because evidence from historic studies of invertebrates (Marris, 2000; McIntosh, 2001; Russell, 2012) and limited monitoring for mice on Bollons Island in 2014 (B. Rance pers. comm. 2014) gave sufficient confidence that mice were not present. These islands provided a natural refuge of 58.8 ha, 1.5 km north of Antipodes Island, where species would not be exposed to bait.

Baseline monitoring of endemic land bird taxa was conducted on Antipodes Island between 2013 and 2016 including immediately prior to bait application in winter 2016. Post-eradication monitoring occurred in the weeks after bait application in July 2016, and in the summers of 2017 and 2018, to record any population impacts of the operation. Distance sampling (Buckland, et al., 2001) was used to estimate the density and abundance of the endemic Antipodes parakeet (*Cyanoramphus unicolor*), Reischek's parakeet (*Cyanoramphus hochstetteri*), and the endemic subspecies of the New Zealand pipit (*Anthus novaeseelandiae steindachneri*). The perpendicular distance to individuals or groups of birds was measured from transect lines of variable length to the nearest metre using a laser range-finder. Transects were distributed throughout the island and repeated as often as practicable. The aim was a sample of 60 to 80 encounters of each species for robust modelling of the detection probability and resultant population density. The technique relies on sightings of birds, so sampling was generally avoided when the weather was wet and cold as birds are less conspicuous. The computer software 'Distance 6.2' (Thomas, et al., 2010) was used to analyse the data and compute population estimates. As the number of detections recorded was low for many of the survey periods, data were pooled and a global detection function was computed, from which survey specific estimates of density were calculated (Buckland et al. 2001). Visual comparison of point estimates and their 95% confidence intervals were reinforced using a comparison of Poisson rates (`poisson.test`; R Core Team, 2013) for three paired pre- and post-toxin application survey dates and departures from a hypothesis of no change in density tested.

Antipodes snipe (*Coenocorypha aucklandica meinertzhagenae*) were monitored by recording the number of snipe seen per hour by observers traversing the island on foot, to give an encounter rate. The change in encounter rate between years was assessed using a generalised linear model with negative binomial errors.

To determine if the breeding success of Antipodean albatross was impacted by the operation, the fledging success of Antipodean albatross chicks within 50 m of the load site was recorded in summer 2017 by visiting the nests prior to chicks fledging. The results were compared with fledging success of chicks, alive at the time of bait application, in two study areas on Antipodes Island.

No formal searching for potentially poisoned animals was done but carcasses found opportunistically were examined. The gut cavity was opened and inspected for haemorrhaging and or the presence of green bait in the stomach or intestines indicating poisoning by brodifacoum. Liver samples were collected from the carcasses of pipits and snipe and stored frozen. Samples were sent to Landcare Research and analysed using High Performance Liquid Chromatography with a detection limit of 1×10^{-6}% (0.001ppm).

Project communication

Public engagement was measured by recording the number of media articles about the project (on television, radio, print) and visits to the project's website www.milliondollarmouse.org.nz) and Facebook page (www.facebook.com/milliondollarmouse) during the operational phase.

RESULTS

The baiting operation was implemented and completed in winter 2016. Insufficient resourcing in the first year of planning and competition with other organisational priorities put pressure on the project team and risked delaying implementation. The development of project knowledge and a wealth of experience enabled quality advice from DOC's IEAG. Their strong support maintained focus on objectives and influenced the prioritisation of resources in the preparation phase. Procuring helicopter services and a cargo ship were the crux of logistics planning but proved difficult due to a small pool of suitable suppliers and complex processes. Over a year and a half was spent investigating options and developing trust with potential suppliers to prove the viability of the work and find capable operators who were willing to commit.

Calibration of bait buckets gave a usable swath width of 90 m for standard buckets (360° spread) and 40 m for the deflector bucket (180° spread). Pre-departure trials identified important improvements in systems and componentry including changes to the pneumatic feed from helicopters to the bait bucket, replacement of incorrectly sized bracing elements on the hangar and refinement of the system for its construction. Trials identified that lifting helicopters by the rotor head was the best technique to manoeuvre them in and out of the ship's hold.

The toxicity of all 131 batches of bait supplied met the contract standards. The average toxicity was 19.8 ppm of brodifacoum and the range was 16.5 ppm to 23.9 ppm ± 7%. The operational team arrived at Antipodes Island on 27 May 2016. It took approximately 90 minutes to extract each helicopter from the ship's hold and ready them for flying. Ship unloading was completed with 250 loads flown ashore over 12 days with suitable weather for helicopter operations occurring periodically on five of those days. Helicopter long-line operations to unload and load the ship were challenging and required precision from the pilots and a strong communicator on the deck of the ship to inform the pilot of the position of the hook and help direct the work. The construction team of six people established the field camp, completed complex site preparations and safely installed temporary infrastructure within 11 days before departing with the transport vessels on 7 June 2016. An emergency response exercise was conducted on 8 June to practice helicopter recovery of a person from the water with a rescue scoop net and a rescuer in a human sling on a long-line.

Readiness for baiting was achieved by 9 June 2016 but poor weather delayed baiting until 18 June 2016 when a brief respite in conditions allowed baiting of a small area (54 ha). This gave the opportunity for an initial test of personnel, loading systems and equipment ahead of better weather windows. The baited area incorporated the field camp and load site, enabling structure baiting to be completed to make bait available early in the programme around the accommodation area where there was the highest risk of alternative food sources for mice. Aerial baiting continued incrementally as the weather allowed until coverage was complete. Suitable weather windows for baiting operations were generally short, and conditions were changeable and generally windy. The longest continuous period of bait application achieved was 3.5 hours. Each day's baiting built on previous work using a "rolling front" approach, with the aim of minimising the area needing rebaiting if work was interrupted for too long.

Treatment one was completed on 29 June 2016 with bait application occurring on 18, 21, 22, 27, 28 and 29 June. The interruption after baiting on 22 June was greater than three days, so the last two bait swaths sown that day were sown again on 27 June with 50% overlap. A total application of 45.6 t of bait was applied during treatment one at an average rate of 21.6 kg/ha. No mouse sign was detected on either of the Windward islands so neither were baited, increasing the area where land birds would not be exposed to bait to 75.3 ha.

Treatment two commenced on 8 July, continued to 10 July and was completed on 12 July 2016. A total of 19.9 t was spread at an average application rate of 9.6 kg/ha. The average sowing rate for both treatments combined was 31.2 kg/ha, including application of all the contingency bait. Contingency bait was additional bait (20% of the planned total) taken to mitigate the risk of loss or damage during transport and storage, or of the treatment area being larger than expected. The rate of bait spreading averaged 1.79 t/hr for the first treatment and 0.93 t/hr for the second, giving an overall average of 1.44 t/hr. The interval between treatments was at least 16 days for 97% of the area, and between ten and twelve days for the remainder. Few technical issues with bait spread were encountered and none limited operations.

Rainfall data were collected daily, and some form of precipitation fell most days. A total of 7.9 mm fell in the 48 hours following application of 15.6 t of bait on 22 June in treatment one. Bait degradation was not formally monitored. However, visual inspection showed baits were weathered but generally intact at the start of treatment two, 20 days after application.

Analysis of GPS flight records for aerial bait spread showed that comprehensive bait coverage was achieved with no apparent gaps. The total maximum amount of bait taken from all bait stations set up for structure baiting was 240 g of the 4 kg available. Most of the bait take occurred in the first three nights and 73% of consumption occurred by night six. Imagery from a trail camera showed mice picking up and carrying away the 2 g bait pellets. Two mice were last recorded taking bait on 7 July, 20 days after application. Dissection of a mouse trapped nearby on the same day showed the stomach and intestines were green and full of bait.

Table 1 Incidental dead bird finds on Antipodes Island following bait application.

Species	Autopsy	Brodifacoum (µg/g) ± 6%
Antipodes parakeet *Cyanoramphus unicolor*	1 poisoned	Unknown
Reischek's parakeet *Cyanoramphus hochstetteri*	1 poisoned	Unknown
Pipit *Anthus novaeseelandiae steindachneri*	3 poisoned	0.028; 0.034; 0.01
Snipe *Coenocorypha aucklandica meinertzhagenae*	2 no sign	0.015; 0.031
Mallard duck *Anas platyrhynchos*	1 poisoned	Unknown

Table 2 Comparison of Poisson rates at two time points pre- and post-application of toxin on Reischek's parakeet, Antipodes parakeet and Antipodes pipit. Rate ratios, their 95% CI's and tests of departure from a hypothesis of no change in density between surveys are reported. Rate ratios <1.0 indicate population decline and those >1.0 population increase between surveys.

	Comparison of Poisson rates between surveys		
	Pre-drop 2016 & Post-drop 2016	Pre-drop 2016 & Jan/Feb 2018	Post-drop 2016 & Jan/Feb 2018
Reischek's parakeet	0.17 (0.13–0.23)**	0.85 (0.63–1.17)#	4.97 (3.92–6.30)**
Antipodes parakeet	0.57 (0.36–0.95)*	2.91 (1.81–4.88)**	5.09 (3.81–6.77)**
Antipodes pipit	0 (0.05–0.10)**	1.38 (1.08–1.76)*	19.44 (13.84–27.94)**

** *P* <0.001; * *P* <0.05; # not significant

No mouse sign was detected from 7,170 tracking tunnel nights and searching with dogs during mouse monitoring in summer 2018. The search effort and the evidence were reviewed by DOC's Island Eradication Advisory Group and the eradication of mice from Antipodes Island was declared successful in March 2018.

Non-target species impacts

Bait trials in 2013 demonstrated 100% uptake of the bait by mice and suggested a risk of primary poisoning for pipits but not for parakeets or snipe (Elliott, et al., 2015). During the eradication operation itself, eight dead birds of five species were found incidentally and all had been poisoned (Table 1). The associated search effort was at least 103 hours of extensive field work for monitoring land birds. Additionally, staff walked an 800 m route between Reef Point and the load site (Fig 1) almost daily for the six weeks between initial bait application in the area and departure. During the operation, some pipits were observed occasionally pecking at baits and some baits were found to have been chewed by parakeets, but most baits were untouched.

Despite the use of a global detection function, low numbers of observations led to large confidence intervals about density estimates derived from distance sampling (Figs 2, 3 and 4). Prior to 2016, only the sampling of Reischek's parakeets in October 2014 (61 encounters) reached the desired sample size of 60 to 80 encounters. In 2016, pre-baiting sampling for Antipodes parakeets (22 encounters) and post-baiting sampling for pipits (40 encounters) failed to reach this target. Overall, more sampling was done immediately post-baiting in 2016 (329 encounters) than before (186 encounters) due to time constraints. Poor weather also often constrained the method. The results (Table 2; Figs 2, 3, and 4) suggest that a significant number of pipits and parakeets probably succumbed to brodifacoum poisoning immediately following the application of bait. However, the populations of pipits and both parakeet species were able to persist and have increased greatly each year, recovering to densities that are similar to or higher than pre-eradication estimates by summer 2018 (Table 2; Figs 2, 3 and 4). Pipits have responded particularly strongly with very large year on year increases in density estimates since 2016. Anecdotal observations in summer 2018 were consistent with the reported increase. On most occasions when monitoring team members sat down in the field, pipits would immediately appear and walked around and on them, finding food items such as caterpillars within minutes (F. Cox, pers. comm. 2018).

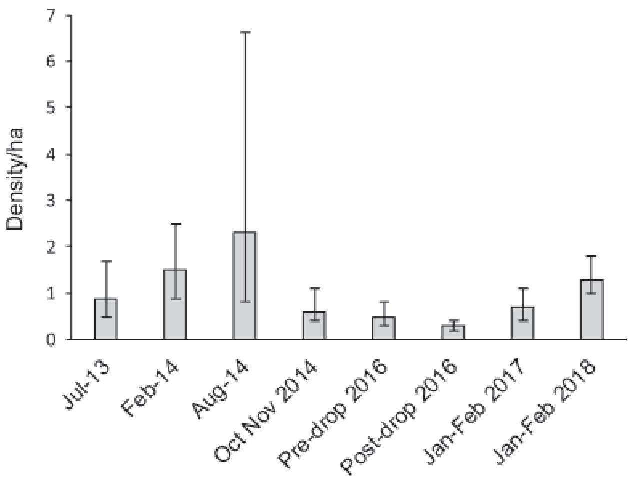

Fig. 3 Distance sampling results for Antipodes Island parakeets, Antipodes Island.

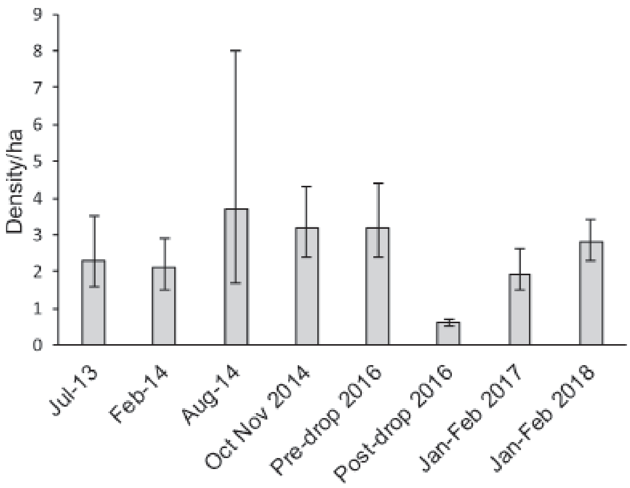

Fig. 2 Distance sampling results for Reischek's parakeets, Antipodes Island.

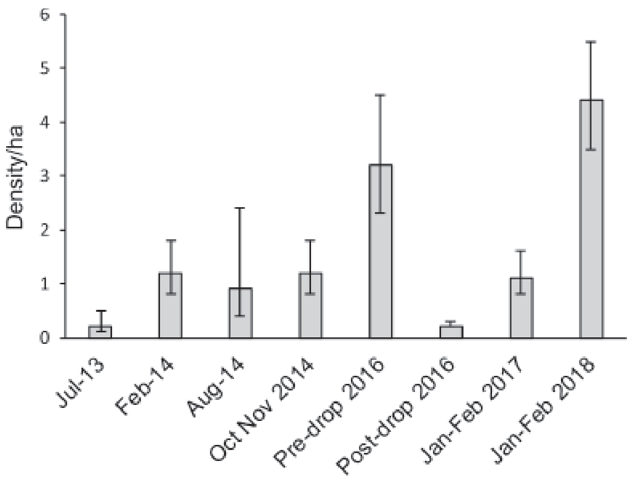

Fig. 4 Distance sampling results for pipits, Antipodes Island.

Table 3 Results of snipe encounter rate surveys recorded on Antipodes Island between 2013 and 2017.

Year	Person hours	Snipe seen	Snipe seen per hour	Change between years (%)	p
2013	341	38	0.1079		
2014	206.75	26	0.1322	123	0.4938
2015	140.5	17	0.1279	97	0.9267
2016	178	6	0.0330	26	0.0085**[a]
2017	224	8	0.0345	105	0.9373
2018	783	132	0.1640	475	0.0001***[b]

[a] Note significant difference in encounter rate between 2015 and 2016 prior to the eradication operation.
[b] Note significant difference in encounter rate between 2018 and 2017.

Snipe have been monitored each summer between 2013 and 2018. Snipe were more abundant in 2018 than ever before, but there has been considerable inter-annual variation in snipe abundance and the difference between 2018 and all the other years is not significant (Table 3). The between-year change in snipe abundance is probably more informative. Significant changes in snipe abundance occurred in 2015–2016 (a decline) and 2017–2018 (an increase). The large decline (72%) in the snipe encounter rate between 2015 and 2016 occurred before the mouse eradication so was not a result of the poison operation. The reason for this is unknown. There was a small, non-significant increase in the snipe encounter rate between 2016 and 2017 (Table 3), suggesting little or no by-kill of snipe during the mouse eradication. In contrast a dramatic increase (475%) occurred in snipe encounters between 2017 and 2018.

Helicopter activity did not have a detrimental effect on nearby Antipodean albatross chicks. All seven chicks within 50 m of the bait loading site were alive at the completion of operations and six out of the seven of them (86%) fledged successfully in early 2017, comparable with 90% outside the load site.

Scientists visiting Antipodes Island in summer 2017 and summer 2018 also noted a greater abundance of moths and the endemic fly (*Xenocalliphora antipodea*) than before the eradication of mice, observing them on flowers of the native groundsel (*Senecio radiolatus*) and Macquarie Island cabbage (*Stilbocarpa polaris*). This endemic fly was also abundant inside the Antipodes Hut for the first summer in over 20 years of visitation. A gathering of hundreds of large noctuid moths, suspected to be *Graphania ustistriga*, was also observed for the first time in 2018 despite 10 previous month-long summer visits to Antipodes Island between 1996 and 2017 (K. Walker, pers. comm. 2018). Large caterpillars, suspected to be larvae of the same noctuid moth species were regularly seen and observed being preyed on by pipits (K. Walker, pers. comm. 2018).

Project communication

Media coverage of the operation included seven prime-time television news stories and several radio interviews, print and online stories. Social media engagement peaked in June 2016 with 23,906 views of the MDM website and 71,967 on the MDM Facebook page. DOC social media also peaked at 77,710 views for the month. Outreach was amplified through the communications networks of project partners, the Morgan Foundation, WWF-New Zealand and Island Conservation.

DISCUSSION

A robust plan was formulated and delivered despite initial difficulties sourcing shipping and helicopter services. Complex projects require good resourcing in the planning phase and organisational prioritisation with significant scale up in resourcing for the preparation phase. Key factors for the delivery of the project were a) quality technical advice, b) single point accountability for overseeing the work and a team approach during preparations and operational phases, c) use of experienced personnel in key roles, d) a proven bait product, e) dependable and tested equipment, f) extensive contingency planning, g) a partnership approach with suppliers and e) the financial and moral support of private and public partners.

The brevity and inconsistency of weather opportunities in this environment showed the importance of being prepared and effectively using every opportunity to complete baiting. Additional skills and operational experience improved team performance and self-sufficiency. Equipment could generally be maintained on site and situational decision-making benefitted from the advice of senior team members. High speed internet access and video production capabilities enabled the team to communicate the project directly and engage an audience. Pilots' long-lining capabilities for ship operations could be considered a separate skill from baiting and, if necessary, pilots with specific skills should be engaged for the task. Similarly, coastal baiting with the deflector bucket requires specific attention and experience.

Non-target impacts

Monitoring evidence suggests the adverse effects of the operation on land birds were short lived. These impacts are expected to be outweighed by the long-term benefits to native species from the permanent removal of competition with mice. The risk to non-target species was effectively limited by relying on natural populations on Bollons and Archway Islands where they weren't exposed to bait. Prior to the mouse eradication, both parakeets and the pipit had rarely been observed making flights of more than 100 m on Antipodes Island, so while they are capable of crossing the 1.5 km strait between Bollons Island and the main Antipodes Island, it must have been a rare event. The risk of parakeets and pipits, resident on Bollons and Archway Islands, being killed by poison when they commuted across the strait was judged low. This reasoning eliminated the need to catch and maintain a captive population. During the bait uptake trial neither parakeet species was detected eating bait, yet both species were killed by the poison. Parakeets may have become habituated to the bait during the operation because of the longer exposure (more than 35 days) and changing palatability of baits as they weathered relative to the non-toxic trial (14 days). The large variability in population density estimates derived from distance sampling were largely driven by the relatively low encounter rates for all three species monitored using this method and should be treated as indicative only. More data would have improved the robustness of the results as would an improved sampling design to account for only recently discovered shifts in winter distribution for both parakeet and pipits. This, however, is difficult to achieve for such a remote and expensive site to visit and for one that frequently experiences less than ideal survey conditions in generally time-constrained survey periods.

It is unlikely that recruitment alone could account for the apparent rapid recovery of pipits and parakeets by summer 2017 (Figs 2, 3, and 4), suggesting the distance sampling results overestimated the losses and/or recovery. For both parakeet species, the large increases in population density, relative to post-baiting lows, were observed before most chicks had fledged (G. Elliott pers. comm. 2017). Pipits are unlikely to have raised more than one clutch by January 2017 which doesn't account for the nearly 500% increase in the population density estimate in summer 2017 since their post-baiting low. The similarly large increase in the estimated density of pipits between 2017 and 2018 (Fig. 4) is more likely to be real considering the observations of field staff.

The very large increases in the encounter rate of snipe and the density estimate of pipits in summer 2018 are presumed to be the result of large increases in the abundance of invertebrates following the eradication of mice and the resultant increases in reproductive output and survival.

Effective distance sampling for pipits within dense coastal vegetation, a habitat favoured by pipits in winter, was problematic. The short time-frames available during the operation for monitoring immediately before and after baiting meant distance sampling occurred in variable conditions and with variable effort across different habitat types, which may have exaggerated the estimated population declines following bait application. The extraordinarily large estimate of pipit population density pre-baiting in 2016 (Fig. 4) is possibly biased by proportionally greater sampling effort of abandoned penguin colonies (where pipits and parakeets are now known to congregate in winter) relative to that within the island interior (and where most of the 2013 counts were done). This reinforces the uncertainty of results.

The seasonal timing of distance sampling for land birds before and after baiting was also inconsistent (Figs 2, 3 and 4). The observed changes in seasonal distribution of these species therefore makes the use of a global detection function (which assumes constant detectability across surveys) problematic and dilutes direct comparability of the density results. Changes in detectability caused by movements to and from the coast may be biasing the results and at least partly account for the relatively low population density estimates so soon after the bait spread. It is recommended that results from surveys done at the same time be pooled if sufficient data are available.

The eradication of mice from Antipodes Island is a huge achievement for conservation in New Zealand. Hundreds of years of ecological devastation by mice has been halted and indigenous wildlife has started to recover. The importance of the result is reflected by the national and international protection of the site, recognising its special natural heritage values. The result provides momentum to New Zealand's Predator Free 2050 initiative and is a step closer to the vision of a New Zealand Subantarctic Islands region free of mammalian pests. Of the five island groups in the region, only Auckland Island now has mammalian pests: pigs (*Sus scrofa*), cats (*Felis catus*) and mice (*Mus musculus*). Over time it is expected that the invertebrate fauna on Antipodes Island will recover to reflect the abundance and species diversity recorded on Bollons Island and Archway Island, where no mice were present. It is hoped that species of larger-bodied ground invertebrates (for example tenebrionids), reduced to low abundance, will recover and others which became extinct on Antipodes Island through predation by mice (for example the unidentified weta and *Loxomerus* sp.), can be successfully reintroduced from the offshore islands where they may survive. The population densities of land bird species are expected to further increase and stabilise with the recovery of food sources and lack of competition with mice. Absent burrowing seabirds, for example black-bellied storm petrel, are also expected to recommence breeding on Antipodes Island. Further monitoring for land birds will occur opportunistically on an annual basis in conjunction with albatross research. Broader outcome monitoring will be repeated in approximately five to ten years' time and will include a repeat of invertebrate sampling, sampling of the seabird species breeding on Antipodes Island and measurement of change in vegetation monitoring plots.

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REFERENCES

Broome, K., Brown, D., Brown, K., Murphy, E., Birmingham, C., Golding, C., Corson, P. and Cox, A. (2019). 'House mice on islands: management and lessons from New Zealand'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 100–107. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L. and Thomas, L. (2001). *Introduction to Distance Sampling: Estimating Abundance of Biological Populations*. UK: Oxford University Press.

Davies, D., Dilley, B.J., Bond, A.L., Cuthbert, R.J. and Ryan, P.G. (2015). 'Trends and tactics of mouse predation on Tristan albatross *Diomedea dabbenena* chicks at Gough Island, South Atlantic Ocean'. *Avian Conservation and Ecology* 10(1): 5.

Dilley, B., Schoombie, S., Schoombie, J. and Ryan, P. (2016). 'Scalping' of albatross fledglings by introduced mice spreads rapidly at Marion Island'. *Antarctic Science* 28(2):73–80.

Elliott, G.P., Greene, T.C., Nathan, H.W. and Russell, J.C. (2015). 'Winter bait uptake trials and related field work on Antipodes Island in preparation for mouse *(Mus musculus)* eradication'. *DOC Research and Development Series* 345. Wellington, New Zealand: Department of Conservation.

Imber, M.J., Bell, B.D. and Bell, E.A. (2005). 'Antipodes Island birds in autumn 2001'. *Nortornis* 52(3): 125–132.

International Chamber of Shipping (2008). *Guide to Helicopter/ship Operations; Fourth Edition*. 12 Carthusian Street, London: Marisec Publications.

Marris, J.W.M. (2000). 'The beetle (Coleoptera) fauna of the Antipodes Islands, with comments on the impact of mice and an annotated checklist of the insect and arachnid fauna'. *Journal of the Royal Society of New Zealand,* 30(2): 169–195.

McIntosh, A.R. (2001). 'The Impact of Mice on the Antipodes Islands'. In: P. McClelland (ed.) *Antipodes Island Expedition, October–November 1995*, pp. 52–57. Invercargill, New Zealand: Department of Conservation.

R Core Team (2013). *R: A Language and Environment for Statistical Computing*. Vienna, Austria: R Foundation for Statistical Computing.

Russell, J.C. (2012). 'Spatio-temporal patterns of introduced mice and invertebrates on Antipodes Island'. *Polar Biology* 35: 1187–1195.

Searle, J.B., Jamieson, P.M., Gunduz, I., Stevens, M.I., Jones, E.P., Gemmill, C.E.C. and King, C.M. (2009). 'The diverse origins of New Zealand house mice'. *Proceedings of the Royal Society Bulletin* 276: 209–217.

Taylor, R. (2006). *Straight through from London. The Antipodes and Bounty Islands, New Zealand*. Christchurch, New Zealand: Heritage Expeditions.

Thomas, L., Buckland, S.T., Rexstad, E.A., Laake, J.L., Strindberg, S., Hedley, S.L., Bishop, J.R.B., Marques, T.A. and Burnham, K.P. 2010. 'Distance software: design and analysis of distance sampling surveys for estimating population size'. *Journal of Applied Ecology* 47: 5–14.

Scaling down (cliffs) to meet the challenge: the Shiants' black rat eradication

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Abstract A successful ground-based eradication of black rats (*Rattus rattus)* was undertaken on the remote, uninhabited Shiant Isles of north-west Scotland over winter (14 October–28 March) 2015–16. The rat eradication was carried out as part of the Shiants Seabird Recovery Project, which aims to secure long-term breeding habitat for protected seabirds and to attract European storm petrels and Manx shearwaters to nest on the Shiants. Throughout the eradication operation, teams were stationed on two of the three main Shiant islands (Eilean an Tighe, Eilean Mhuire), with access to the third (Garbh Eilean) via a boulder causeway from Eilean an Tighe. Bait (Contrac® blocks containing the anticoagulant bromadiolone 0.005% w/w), was deployed in a grid of 1,183 bait stations covering all areas of the islands and sea stacks. Bait stations were set 50 m apart, with intervals reduced to 25 m in coastal areas of predicted high rat density. Difficult areas were accessed by boat and cliffs of ~120 m in height were accessed by abseiling down ropes made safe using either bolted anchors or ground stakes. The team of staff and volunteers worked through difficult conditions, deploying bait and monitoring intensively for any surviving rats using a combination of tools. The islands were declared rat free in March 2018. This ambitious and challenging project has greatly enhanced UK capacity in rodent eradications for the purposes of conservation.

Keywords: biosecurity, conservation priority, eradicate, incursion, invasive alien vertebrate, island restoration, reinvasion

INTRODUCTION

The Shiant Isles is a group of small, uninhabited islands that lie in the Minch (57.9° N, 6.4° W), ca. 6 km east of the island of Lewis and Harris, north-west Scotland. Of the Shiants' three main islands, the largest two: Garbh Eilean (GE, 88 ha) and Eilean an Taighe (ET, 54 ha) are connected by a boulder causeway, and ~500 m to the east of GE lies Eilean Mhuire (EM, 31 ha). A chain of sea stacks, the Galtachan, lie to the west of GE (Fig. 1).

Archaeological evidence documents previous inhabitation of these islands by humans dating back perhaps to the Iron Age (Foster, 2000) but since the 18th century the Shiants have remained uninhabited, and the last remaining building (the 'bothy' on ET, close to the boulder causeway) is occupied only during visits by the islands' owners, or by tourists.

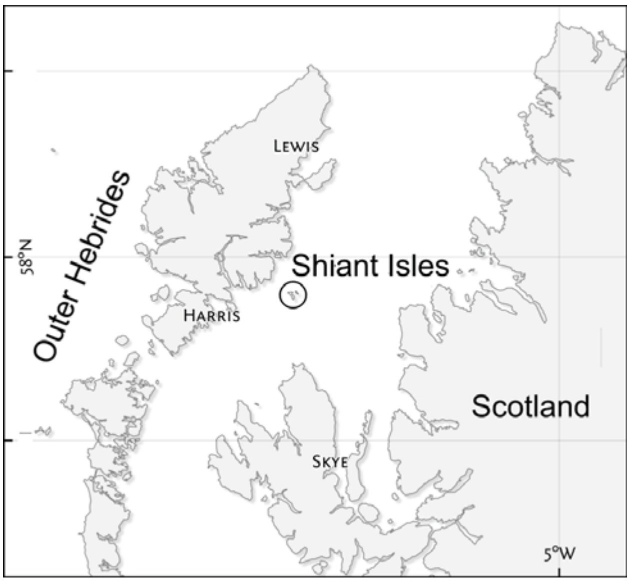

Fig. 1 Location of the islands in the wider area of north-west Scotland.

The Shiants consist mainly of dolerite sills, formed by intrusion of igneous rock between overlying sedimentary rock strata. These sills were then exposed to reveal impressive, columnar structures that now rise steeply to a height of ~ 150 m at their highest point on GE and have been eroded to form extensive boulder scree areas, particularly on the east side of GE (Walker, 1930). The smallest of the three main islands, EM, has cliffs rising to around 80 m, and more conglomerate substrate than ET and GE (Walker, 1930; Gibb & Henderson, 1996).

Habitats present on the islands range from blanket bog and wet heath across the interior of GE and ET, to fertile, species rich grasslands along the coasts of GE and ET and across ME. The maritime environment has a strong influence on the composition of the islands' vegetation and soils have been enriched by guano from centuries of seabird occupation and from past human cultivation. The three main islands have all been historically grazed by sheep (*Ovis aries*) (counts of sheep performed year-round gave estimates of 50 to 80 per island). A colony of grey seals (*Halichoerus grypus*) breeds on the islands, and both common seals (*Phoca vitulina*) and otters (*Lutra lutra*) are frequent visitors. Other than the sheep and an introduced population of black rats (*Rattus rattus*) there are no other known resident populations of terrestrial mammals.

The remoteness of the Shiants, their large amount of suitable habitat and proximity to feeding grounds makes the islands ideal breeding sites for various seabirds. Their importance is internationally recognised through designation as a Site of Special Scientific Interest (SSSI; site code 8575) and as a Special Protection Area (SPA; EU code UK9001041) for breeding populations of puffins (*Fratercula arctica*) (approximately 10% of the UK breeding population, Mitchell, et al., 2004), razorbills (*Alca*

In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) (2019). Island invasives: scaling up to meet the challenge, pp. 138–146. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

torda), common guillemots (*Uria aalgae*), European shags (*Phalacrocorax aristotelis*), black-legged kittiwakes (*Rissa tridactyla*), and northern fulmar (*Fulmarus glacialis*), and wintering barnacle geese (*Branta leucopsis*). The seabird assemblage also includes great skua (*Stercorarius skua*), black guillemots (*Cepphus grylle*), herring gulls (*Larus argentatus*), common gulls (*L. canus*), great black-backed gulls (*L. marinus*) and lesser black-backed gulls (*L. fuscus*). White-tailed eagles (*Haliaeetus albicilla*) returned to breed on the islands in 2014 after an absence of over 100 years, following the re-introduction of the species to Scotland (Love, 1983). Other seabirds such as European storm-petrel (*Hydrobates pelagicus*) and Manx shearwater (*Puffinus puffinus*) have not been recorded as breeding at the Shiants, despite the large amount of suitable habitat for these birds. Both of these species are of international conservation concern. At the last major census, European storm petrels breeding on the isles of the UK and Ireland were estimated to number 125,000 pairs, representing 3–11% of the global population. In the same census Manx shearwaters were estimated at 332,000 pairs breeding in the UK and Ireland, with the majority found on the islands of Rum, north-west Scotland (120,000 pairs), Skomer, Wales (102,000 pairs) and Skokholm, Wales (46,000 pairs, Mitchell, et al., 2004). A further survey of Manx shearwaters on Rum has also estimated 70,000 breeding pairs at the Rum colony (Murray & Shewry, 2002).

Rats (*Rattus* spp.) are among the highest risk invasive species, having had devastating effects on native wildlife on island groups such as New Zealand (Towns, et al., 2006) and worldwide through predation, and both competition for and modification of habitat (Jones, et al., 2008). Rats have been recorded on more than 80% of the world's island groups (Atkinson, 1985), but their successful removal from islands ranging in size from less than 1 ha to 12,875 ha has been pioneered in New Zealand and is being applied across the globe. In the UK, rats have been successfully eradicated from islands ranging in size from just one hectare (e.g. Inchgarvie, Firth of Forth, Scotland) to 1,300 ha (Canna & Sanday, Scotland) (Ratcliffe, et al., 2009; Thomas, et al., 2017a; Bell, 2019). Of the successful UK island rat eradications, all were of brown (Norway) rat (*Rattus norvegicus*) except in the case of Lundy, which included populations of both brown rat and black rat (Thomas, et al., 2017a; Bell, 2019). The removal of rats is essential where predation either limits productivity or threatens to lead to the complete loss of important seabird colonies.

Black rats were introduced to the Shiant Isles (accidentally, it is assumed) by humans, either through stock movements by previous island inhabitants or by shipwreck (e.g. Haswell Smith, 2004), though no evidence has established how the rats arrived. The rats are thought to have had negative impacts on the seabirds at these islands as follows. Diet analysis at the Shiants has indicated that rats consumed a range of material of marine origin (Stapp, 2002) as well as vegetation and invertebrates present at the Shiants (Stapp, 2002; Bell, 2013). The stable isotope ratios of carbon and nitrogen, extracted from rat tissues of individuals caught at seabird colonies were closer to those from tissues of seabird origin than those of rats caught from areas away from seabird colonies (Stapp, 2002). This indicated that in the seabird breeding season, coastal colonies of rats were likely to have fed upon on seabird eggs and chicks.

Following a detailed assessment of UK islands with invasive, non-native species the Shiant Isles were identified as being a priority site for rat eradication because of their abundance of potential petrel and storm-petrel breeding habitat (Ratcliffe, et al., 2009). A successful rat eradication at these islands would additionally benefit the existing colonies of protected seabirds. Since the islands lie approximately 6 km offshore and are uninhabited by humans, the risk of natural invasion by brown rats from the nearest islands of the Outer Hebrides is considered to be low. A feasibility study commissioned by the Royal Society for the Protection of Birds (RSPB), and undertaken by Wildlife Management International Ltd (WMIL) in April 2012, found that eradication of the black rat population at the Shiants was feasible (Bell, 2013).

Subsequently, the Shiant Isles Seabird Recovery Project (SSRP) was established as a four-year partnership between the islands' owners (the Nicolson family), RSPB and Scottish Natural Heritage (SNH). The four core aims of the project were: i) to eradicate the invasive black rat population; ii) actively encourage petrels (European storm petrel and Manx shearwater) to nest at the islands; iii) audit island biosecurity at UK SPAs and iv) increase UK capacity in island restoration. Funding for the project was provided by the EU LIFE fund (LIFE13/NAT/UK/000209 LIFE+ SHIANTS), SNH, and RSPB.

The eradication component of the SSRP was undertaken over the period 2015–2016. An open tender process was used to invite operators to bid for a contract to undertake eradication work at the Shiants. This resulted in the selection of WMIL to carry out the eradication operations. The eradication set up, methods and technical operations will be reported on here.

METHODS

Pre- and post-eradication monitoring

Monitoring of the two main islands' (ET and GE) existing seabirds, land birds, vegetation and invertebrates was carried out for one year before the eradication and for the subsequent three years post-eradication. The aims of this ecosystem monitoring were to detect changes, if possible, and hence assess the benefits of the eradication. Full methodology and results for this will be presented elsewhere. A population census of all seabirds, carried out by RSPB and SNH, was undertaken at the Shiants during June 2015, as part of SNH's programme of Common Standards Monitoring of protected areas (SSSIs and SPAs) (Taylor, et al., 2018). A pre-eradication assessment site visit was undertaken during July 2015 to finalise plans, logistics, and health and safety requirements.

Permits and authorisations

A Habitats Regulations Appraisal (HRA) was carried out by SNH to assess the likelihood of any adverse impact of the rat eradication on the qualifying features of the SPA. This required a full Appropriate Assessment under the Conservation (Natural Habitats, &c.) Regulations 1994 (as amended). In addition, a full assessment of the Operations Requiring Consent (ORC) was also undertaken for the Shiant Isles SSSI. Justification of the chosen rodenticide (bromadiolone) formulation, estimated quantity needed, and method of application was presented in the Appropriate Assessment and ORC application, detailing how the operation would be undertaken across all islands and sea stacks of the Shiants. A licence under the Wildlife & Countryside Act (1981) was granted to cover possible disturbance to breeding golden eagles (*Aquila chrysaetos*) and white-tailed eagles which are specially protected by Schedule 1 of the Act. Planning permission was obtained from the Comhairle na Eilean Siar (Western Isles Council) to allow the temporary installation of portable cabins on the island to store rodenticide bait and provide shelter for winter eradication teams. For the installation of two temporary moorings, a five-year marine license (issued by Marine Scotland) was granted for which an annual fee was paid to the Crown Estate. Assessments of archaeological sensitivity were carried out in person by experts from the Comhairle nan Eilean Siar and also by RSPB. Maps of archaeologically sensitive sites were used to ensure that these features were not disturbed by placement of cabins, bait stations or by the passage of workers around the islands during the eradication.

A detailed health and safety plan was written in collaboration by RSPB and WMIL. This outlined living and working protocols and the establishment of emergency procedures. As part of health and safety requirements, the islands were zoned to indicate areas considered too dangerous to access without the use of support ropes. Rope access was hence deemed necessary to place bait and to check bait stations on steep vegetated slopes or ground ending abruptly at steep cliffs.

Contracts with two local boat operators (Sea Harris Ltd and Engebret Ltd) were established in order to provide boat access to and around the Shiants through the winter operations. An invitation to quote was issued, with the subsequent selection of contractors based on project needs, cost, and suitability of boat service provision.

Rodent anticoagulant resistance tests

An assessment of potential resistance of the rats on the Shiants to bromadiolone rodenticide was carried out by Reading University (Vertebrate Pests Unit), using protocols developed to extract and sequence DNA for the identification of anticoagulant resistance mutations in brown rats (Pelz, et al., 2005; Prescott, et al., 2010). A similar protocol developed specifically for black rat rodenticide resistance testing was not available. However, the approach used represented the best option available because of the lack of rodenticide resistance work that had been undertaken on black rats at the time of the eradication. Rat DNA samples were collected by project personnel from ET in July 2015. Snap traps placed inside tunnels were set overnight and baited with peanut butter. These were visited early the next morning and any rat specimens caught were collected and dissected. A portion of the tail was placed in 100% ethanol for subsequent rodenticide resistance testing. Morphometric measurements (body length, tail length, hind foot length, ear length) were recorded. Stomach contents, sex and reproductive status were also assessed for all of these trapped rats (Bell & Boyle, 2015). All DNA samples were archived for reference in case of resistance or reinvasion by rats at the islands.

Equipment preparation

Off-island preparation of equipment included the construction of 0.75 m long bait stations (Fig. 2) from lengths of 10 cm diameter plastic drain coil. Help was sought from local community volunteers from the Isle of Harris and construction of approximately 700 bait stations was carried out over two days at the Harris Volunteer Centre in Tarbert. The remaining bait stations were constructed by project personnel off and on the Shiants. Bait stations and other equipment were airlifted to the Shiants over two days as part of the set-up phase of the eradiation.

Access to challenging terrain

Camps were established on ET and EM for the winter teams. Portable cabins were installed for safe storage of rodenticide baits and shelter for winter eradication teams. The flat-packed cabins were airlifted to the islands by helicopter in October 2015 and constructed on-site. The existing bothy on ET was re-roofed during the summer of 2015 and was also used as a base camp during the winter operation. Two moorings were installed close to ET, to improve safety for boat access. Boats were used to land on less accessible areas such as the Galtachan sea stacks, and a large rock to the east of EM.

Rope access training was undertaken by seven WMIL and RSPB personnel. Bolts were set in rock at the top of twelve rope access routes (eleven on ET and one on EM). A further eight routes (one on ET, two on EM and five on GE) were accessed using ropes secured by anchors manually set up using a series of three lashed metal stakes.

Non-target mitigation

Measures to prevent secondary poisoning of eagles were provided by the establishment of diversionary feeding protocols. Dead rabbits (*Oryctolagus cuniculus*, collected by manual trapping at a nearby site on the Isle of Harris) were attached to two tables located on GE, with motion activated cameras set up to monitor activity. The diversionary feeding was made available to the eagles from October 30, 2015 until March 17, 2016. However, no fresh food was attached to the table after 12 November 2015 because bait take had reduced to such low levels that the risk of secondary poisoning was deemed negligible. It was also noted that eagles were only intermittently present at the islands, and there was no evidence, e.g. from motion activated cameras, to suggest that diversionary food was utilised by any eagle. Wire clips were fitted to all bait stations after a raven was seen to open one and access the bait – no further instances of non-target vertebrate species accessing bait were observed.

Bait quantity

An estimate of the quantity of rodenticide needed for the eradication was calculated as follows during the planning phase. An application rate was assumed of 0.28 kg rodenticide per bait station (i.e. 10 blocks per station in 684 bait stations on a 50 × 50 m grid; 1.12 kg/ha) for the first four weeks, then 0.14 kg per bait station (0.56 kg/ha) for the subsequent four weeks and 0.056 kg per bait station (0.224 kg/ha) for the remainder of the operation, with consumed bait replenished at each check. Each application, or "round" of bait station checks was expected to take one day using a team of 10 people. It was expected to require at least 30 complete rounds (with replacement of bait) of each station to ensure the eradication of all the rats. At this rate, up to four tonnes of bromadiolone (LD50, oral ingestion 1.125 mg/kg, Meehan, 1978) were estimated to be required to cover the combined island area (171 ha) over approximately five months. Note, that although the stated LD_{50} for bromadiolone as given by the manufacturer is 0.525 mg/kg (Bell Laboratories Material Safety Data Sheet) this is based on laboratory-bred brown rats. Wild populations of black rats may be more tolerant to bromadiolone (Sridhara & Krishnamurthy, 1992). Individual and sex-specific variations in toxicity of bromadiolone to black rats have also been reported (Garg

Fig. 2 Bait station shown open, with rodenticide blocks wired in place.

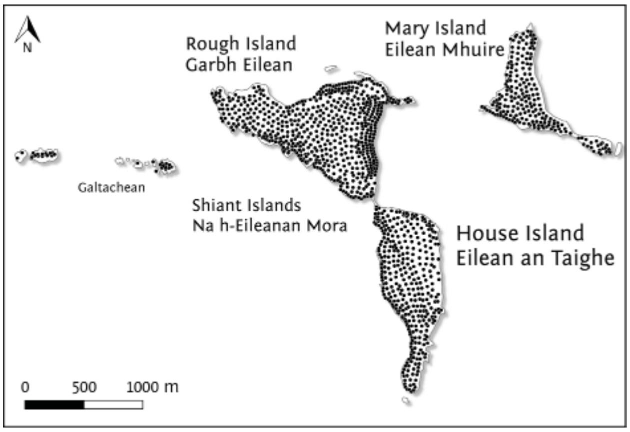

Fig. 3 Map of the bait station grid.

& Singla, 2014). Therefore, a higher LD_{50} was assumed here (Meehan, 1978), to account for these differences.

The actual application rates, bait station grid and number of days to complete a full bait station check during the eradication were different to those calculated in the planning phase due to operational requirements, predicted high rat density areas (i.e. coastal areas) and adaptation to both bait consumption and rat behaviour.

Bait application

Island-wide baiting grids with a total of 1,183 bait stations (Fig. 2) were established across all islands and sea stacks (347 on ET, 594 on GE, 207 on EM, 28 on Galtachan, four on Arch Island and three on Seann Chaisteal) during October 2015. Bait stations were spaced at 50 m intervals across the islands, and 25 m along coasts and through areas of boulder scree e.g. the large area of boulders known as Carnach Mhòr on GE (Fig. 3). This grid spacing has become the current UK best practice protocol for rat eradications (Thomas, et al., 2017b). Rodenticide bait and rat monitoring tools were placed on the Galtachan sea stacks on 17 October 2015.

A team of between two and four people was stationed on ET throughout the operation with a break of one week over the Christmas period. A second team of between two and three people was stationed on EM on a week on/ week off schedule. Rotation of people within these teams occurred on a weekly basis, where weather permitted safe access for boat transfers. Baiting commenced at the islands on 4th November (Table 1). Cereal-based wax blocks (28 g Contrac® All-weather Blox™ (Cas No. 28772-56-7, EU 528/2012), containing the anticoagulant rodenticide bromadiolone at 0.005% w/w) were initially placed loose within bait stations. Regular checks were then made. Stations that were accessible on foot were checked between 16 and 23 times with an average of 6–8 days between checks (Tables 1 and 2). Rope access routes were checked between 3–13 times with an average of 10–15 days between checks (Tables 3 and 4). Frequency of checks at rope access routes was limited by the availability of trained staff on each island and the length of time required to check relatively few bait stations on the routes. The rodenticide blocks were wired in to bait stations starting from the seventh round of baiting on GE, round eight on EM and round nine on ET, with all bait wired in by 8 January 2016 (Tables 1 and 3). Records of bait application at each station were kept in waterproof notebooks and transferred to an electronic database each night. Bait was replaced if damaged by weather or slugs, or at the sign of rat incisor marks. An alternative bait, a soft block (100 g Romax® Rat CP™ (Cas No. 5836-29-3, UK UK-2016-1003), containing the anticoagulant coumatetralyl at 0.0375% w/w), was wired into bait stations (one block alongside the Contrac® blocks) during January and February 2016 (depending on the island) (Tables 1 and 3). This provided an alternative bait for rats not consuming the Contrac® blocks.

Monitoring

During baiting operations, personnel regularly searched for carcasses, including any dead or dying rats present at the surface. Systematic monitoring for surviving rats commenced at the islands on November 28, 2015 and continued for 14 weeks, in tandem with baiting. Monitoring stations were set up at every bait station and at intervals half way between bait stations. Monitoring tools employed included: non-toxic flavoured paraffin wax blocks (chocolate, peanut butter, peanut essence, aniseed); soap; tracking tunnels; snap traps and motion-activated cameras. After 14 weeks, intensive monitoring was reduced and permanent monitoring stations were established at key locations (Fig. 4, Fig. 5) on the three main islands where early detection of any surviving black rats, or an invasion of brown rats, would be likely. These comprised 44 commercially available Protecta™ boxes and ten wooden rodent motels baited with non-toxic chocolate wax blocks (Fig. 4). Monitoring stations were checked in winter of October 2016, January 2017, March 2017 and November 2017, with replacement of old blocks each time. Regular checks were also carried out in the summer (April–August 2016–2018) during island monitoring as part of the Shiants Seabird Recovery Project.

Following the placement of rodenticide and monitoring tools at the Galtachan sea stacks, subsequent checks on 4 November 2015 and 27 February 2016 revealed no rat sign or bait take. The sea stacks were assumed to be rat free and were not treated with further rodenticide during the eradication phase.

A database of all baiting, monitoring, and other activities was maintained throughout the eradication. Observations of potential non-target species, carcasses, and other relevant information were documented throughout the operation.

RESULTS

Bait consumption

Consumption of bait was higher around the coasts of all islands and was the highest around areas of known seabird colonies (Fig. 6). Rats consumed approximately 270 kg (or 9666 blocks) of Contrac® bait in total, mainly during the phase when blocks were loose in bait stations and available for rats to remove whole and cache (Fig. 7). Consumption of Romax® blocks by rats was zero.

An estimate of the numbers of rats present at the time of the eradication was made as follows. Assumptions were made that: each block removed by rats was consumed in its entirety by a single rat; all rats consumed between three and 24 times the lethal dose of bromadiolone (where a lethal dose is delivered by consuming 9.5 g of bait corresponding to approximately one third of a block, assuming an LD_{50} of 1.125 mg/kg for black rats). Hence, it is estimated that there were between 1,208 and 9,666 rats present on the Shiants at the start of the eradication (assuming between one and eight blocks were taken by each rat before death). The mean (± SE) of 4.6 ± 0.1 blocks consumed per bait station overall leads to the estimation that there were 2,099 ± 97 rats on the Shiants. This is lower than previous estimates for black rat on the Shiants; 22–85 rats/ha (3,762–14,535 rats) in 1998 (Key, et al., 1998) and 14–27 rats/ha (2,394–4,617) in 2012 (Bell, 2013). Methods used and timings of each of these population estimates vary.

Table 1 Bait application for stations accessible by foot – bait check, dates for each bait check, quantities deployed and schedule. All rodenticide used was Contrac® All-weather BloxTM unless indicated.

Baiting No	Eilean an Tighe	Garbh Eilean	Eilean Mhuire
1	4/11/15–5/11/15 (8 blocks)	6/11/15–7/11/15 (8 blocks)	7/11/15 (8 blocks)
2	9/11/15 (8 blocks)	11/11/15–15/11/15 (8 blocks)	9/11/15 (8 blocks)
3	16/11/15–17/11/15 (8 blocks)	18/11/15–19/11/15 (8 blocks)	11/11/15–12/11/15 (8 blocks)
4	20/11/15 (8 blocks)	22/11/15–27/11/15 (8 blocks)	22/11/15–27/11/15 (8 blocks)
5	22/11/15 (8 blocks)	30/11/15–2/12/15 (8 blocks)	28/11/15–29/11/15 (8 blocks)
6	28/11/15–29/11/15 (8 blocks)	10/12/15–15/12/15 (8 blocks)	30/11/15 (8 blocks)
7	3/12/15–4/12/15 (8 blocks)	1/1/16–4/1/16 (2 blocks wired in place)	12/12/15–13/12/15 (8 blocks)
8	5/12/15–6/12/15 (2 blocks wired in place)	8/1/16–11/1/16 (2 blocks wired in place)	8/1/16–11/1/16 (2 blocks wired in place)
9	8/12/15–10/12/15 (2 blocks wired in place)	14/1/16–16/1/16 (2 blocks wired in place)	22/1/16 (2 blocks wired in place)
10	17/12/15–6/1/16 (2 blocks wired in place)	19/1/16–21/1/16 (2 blocks wired in place)	24/1/16–25/1/16 (2 blocks wired in place)
11	12/1/16 (2 blocks wired in place)	28/1/16–5/2/16 (1 block wired in place)*	26/1/16 (2 blocks wired in place)
12	18/1/16 (2 blocks wired in place)	7/2/16–9/2/16 (1 block wired in place)*	3/2/16–6/2/16 (1 block wired in place)*
13	24/1/16 (2 blocks wired in place)	12/2/16–14/2/16 (1 block wired in place)*	7/2/16–9/2/16 (1 block wired in place)*
14	30/1/16–2/2/16 (1 block wired in place)*	17/2/16–21/2/16 (1 block wired in place)*	20/2/16–21/2/16 (1 block wired in place)*
15	6/2/16 (1 block wired in place)*	22/2/16–23/2/16 (1 block wired in place)*	9/3/16–11/3/16 (1 block wired in place)
16	11/2/16 (1 block wired in place)*	25/2/16–26/2/16 (1 block wired in place)	14/3/16 (1 block wired in place)
17	14/2/16–5/2/16 (1 block wired in place)*	29/2/16–1/3/16 (1 block wired in place)	
18	21/2/16 (1 block wired in place)*	5/3/16–7/3/16 (1 block wired in place)	
19	26/2/16–27/2/16 (1 block wired in place)	9/3/16–10/3/16 (1 block wired in place)	
20	4/3/16 (1 block wired in place)	16/3/16–20/3/16 (1 block wired in place)	
21	8/3/16 (1 block wired in place)		
22	12/3/16 (1 block wired in place)		
23	22/3/16–23/3/16 (1 block wired in place)		

* Romax® Rat CP™

Interference with bait stations by non-target species was low. Invertebrates, particularly slugs, consumed some bait (1.78 kg). Sheep were estimated to consume 19.6 kg of bait released by kicking up the stations. Ravens and crows were observed to take bait (10.9 kg) by pulling out wires and removing bait station lids, until a more secure wire fastening system was established. No evidence was found of non-target species being affected by the rodenticide.

No carcasses showing signs of anticoagulant ingestion were collected. An adult golden eagle carcass discovered on 15th November was autopsied and showed no signs of anticoagulant poisoning, and the state of decomposition suggested it had almost certainly died before the start of the baiting operation. Diversionary food provided throughout the operation was not removed by any species.

Table 2 Frequency of replenishment of bait stations accessible by foot. Mean number of days (± SE) between the first day of each bait station check; range in number of days to complete check and total number of checks given in parentheses.

Island	Number of days between the first day of each bait station check
Eilean an Tighe	6.2 ± 1.0 days (2–27 days; 23 checks)
Garbh Eilean	6.7 ± 1.0 days (3–22 days; 20 checks)
Eilean Mhuire	8.1 ± 1.8 days (2–27 days; 16 checks)
Total (all islands combined)	6.9 ± 0.7 days (2–27 days; 16–23 checks)

Table 3 Frequency of replenishment of bait stations accessed by rope. Mean number of days (± SE) between the first day of each bait station check; range in number of days to complete each check and total number of checks given in parentheses.

Island	Number of days between the first day of each bait station check
Eilean an Tighe	10.8 ± 1.7 days (4–24 days, 13 checks)
Garbh Eilean	13.5 ± 2.3 days (2–23 days,11 checks)
Eilean Mhuire	14.4 ± 3.2 days (2–29 days, 9 checks)
Total (all islands combined)	12.7 ± 1.3 days (2–29 days; 9–13 checks)

Table 4 Bait application for stations accessed by rope – bait check, dates for each bait check, quantities deployed and schedule. All rodenticide used was Contrac® All-weather Blox™ unless indicated.

Bait No	Eilean an Tighe	Garbh Eilean	Eilean Mhuire
1	17/11/15–29/11/15 (8 blocks)	19/11/15–3/12/15 (8 blocks)	11/11/15 (8 blocks)
2	5/12/15 (2 blocks wired in place)	6/12/15–7/12/15 (8 blocks)	23/11/15 (8 blocks)
3	12/12/15–15/12/15 (2 blocks wired in place)	17/12/15 (8 blocks)	14/12/15–15/12/15 (8 blocks)
4	5/1/16–6/1/16 (2 blocks wired in place)	2/1/16–4/1/16 (2 blocks wired in place)	8/1/16–9/1/16 (2 blocks wired in place)
5	15/1/16 (2 blocks wired in place)	14/1/16 (2 blocks wired in place)	6/2/16 (1 block wired in place)*
6	23/1/16–2/2/16 (1 block wired in place)*	16/1/16–21/1/16 (2 blocks wired in place)	8/2/16–9/2/16 (1 block wired in place)*
7	11/2/16–13/2/16 (1 block wired in place)*	8/2/16–14/2/16 (1 block wired in place)*	22/2/16 (1 block wired in place)*
8	15/2/16 (1 block wired in place)*	21/2/16 (1 block wired in place)*	11/3/16 (1 block wired in place)
9	25/2/16 (1 block wired in place)*	29/2/16 (1 block wired in place)	14/3/16 (1 block wired in place)
10	4/3/16–6/3/16 (1 block wired in place)	7/3/16 (1 block wired in place)	
11	8/3/16 (1 block wired in place)	17/3/16 (1 block wired in place)	
12	16/3/16 (1 block wired in place)		
13	22/3/16–24/3/16 (1 block wired in place)		

* Romax® Rat CP™

Bait stations near to coasts were affected by weather, with 60 washed away during large storms (resulting in a loss of 9.5 kg of bait). A total of 84 kg of bait (from 583 stations) was removed by hand because of damage by mould or dampness that could have rendered it unpalatable to rats.

Rat sign and monitoring

Four rats were found dead at the surface on GE (one fresh carcass on 7 November 2015, four days after baiting commenced); two fresh carcasses on 11 November 2015 (seven days after baiting commenced) and one desiccated carcass on 4 February 2016 (90 days after baiting commenced). Rat sign was recorded on flavoured wax at three monitoring points following the start of the initial baiting phase. These were at three different cliff stations on ET (on 13 December, on 14 December and on 26 February). Monitoring at these cliff stations through March 2016 did not yield any further rat sign. The detection of incisor marks on the flavoured wax block in February 2016, when other food sources were beginning to become available, was treated as a possible survivor from the eradication. To establish whether any rats had survived, in October 2016 rope access teams re-established 470 monitoring stations

Fig. 4 Rodent motel (lower) and ProtectaTM box (upper) monitoring stations.

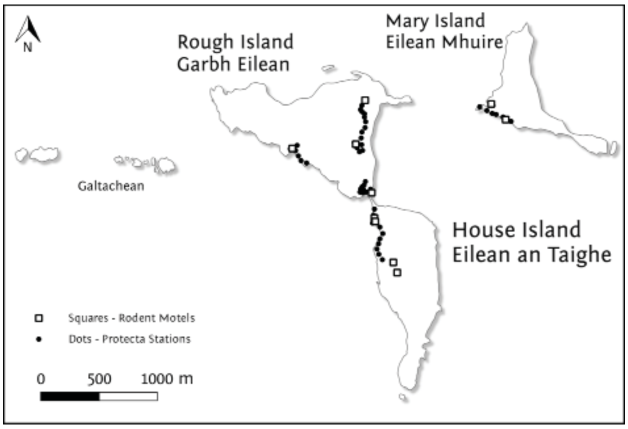

Fig. 5 Locations of permanent monitoring stations.

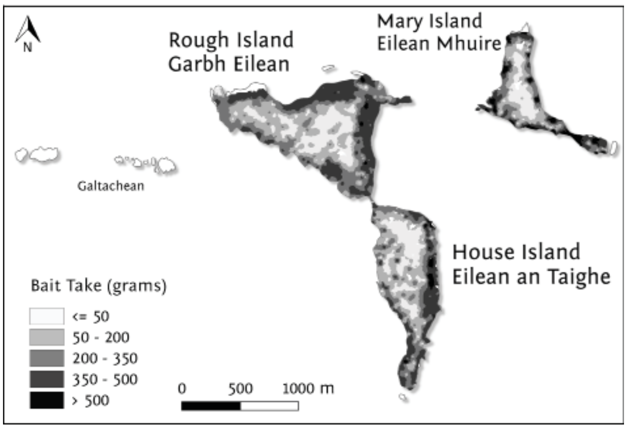

Fig. 6 Quantities of bait taken by rats during the eradication.

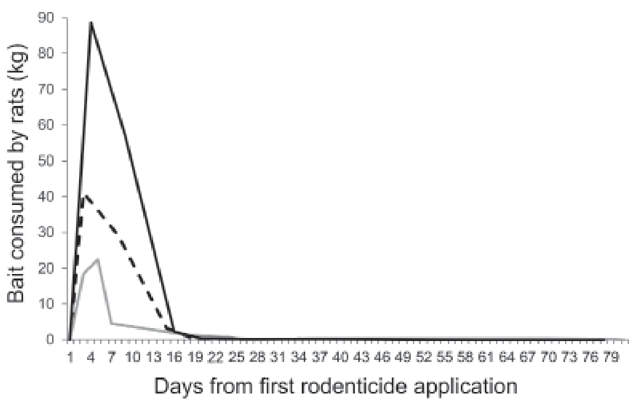

Fig. 7 Quantity (kg) of bait consumed by rats following each baiting application on each island. Solid black line = Garbh Eilean; Dashed line = Eilean an Tighe; Grey line = Eilean Mhuire.

across the three islands with a focus on the cliff stations on ET. Four checks were completed on EM and ET and three checks on GE, finding no sign of rat presence. Permanent monitoring stations on ET, GE and EM were checked monthly in the summer (April–August) and every three months outside the summer season until February 2018. A month-long intensive monitoring check was carried out on all islands and sea stacks in February 2018, with the declaration of rat-free status made on 2 March 2018.

Rodenticide resistance testing

Although some mutations were present within the section of genome sequenced, these mutations were not the same as brown rat mutations known to confer genetic resistance to bromadiolone.

DISCUSSION

Eradication of black rats at the Shiant Isles was one of the four core aims of the Shiants Seabird Recovery Project, now successfully achieved.

The Shiants black rat eradication was an ambitious undertaking on a remote, uninhabited island group, with no existing infrastructure or facilities except the bothy on ET. The operation required the establishment of safe working environments and the provision of shelter, for example, on EM where there are no existing structures on an island of 80 m height. The lack of sanitation, water supply and electricity necessitated basic living conditions and robust and appropriate waste management procedures. Weather influenced operations throughout, sometimes becoming so severe as to prevent access to various parts of the island group. The terrain of the Shiant Isles is steep across many sections of each island, hence the need for rope access at various points. The use of extensive rope access, in particular, allowed a neat solution to the problems of challenging access and a ground-based operation. Other methods, such as bait stations deployed at the end of lines, may have been more difficult to monitor and would also have necessitated the close approach to cliff edges (also requiring support ropes in order to comply with health and safety requirements). As well as the challenging access on foot, the need to address separate islands at the same time by boat presented further logistical and personnel considerations.

New challenges had arisen since the feasibility study (2012) including changes in the profile of the boulder causeway connecting ET and GE resulting in reduced access on foot to GE from the main camp on ET. White tailed eagles had established a breeding site at the islands and mitigation actions were required to address potential secondary poisoning of these predators, which had not been necessary at the time of the feasibility study. This highlights the need to review feasibility studies as an ongoing process within the planning phase.

A number of valuable lessons were learnt during the course of the eradication. In light of the challenges faced by working in difficult conditions, the operation was delivered to a high standard by an effective team. Rope access elements worked well as a result of the thorough training, and of equipment and safety considerations which were appropriate to the operation. Once established, procedures concerning training, preparation, boating and accommodation all worked well because conditions had been considered thoroughly as part of a detailed health and safety plan. Boat access arrangements allowed sufficient access to the separate islands to achieve baiting and monitoring throughout. The whole operation provided positive input into the nearby economy of Lewis and Harris over the winter months. At the end of the eradication phase, the establishment of permanent monitoring stations provided early detection capability, which is necessary as part of delivering long term biosecurity at the islands.

Periods of heavy workload for personnel involved in eradication preparations resulted from time and resource pressures during the preparation phase. There was a need for careful planning of logistics, the satisfying of legal obligations, the need to train local personnel and set up health and safety. As a result, UK-based capacity for undertaking eradications for conservation purposes has been greatly enhanced, and the need for detailed planning from early on has been highlighted. A dedicated logistics coordinator would have been a useful additional staff member to have had in place.

Technical rope access training required a further investment of time and resources and, although the number of trained personnel was sufficient to be able to carry out checks, this did limit the total number of checks possible. Successful communication between team leaders and volunteers took place regularly throughout the operation but has been noted as an area in which continued focus is important in complex operations. Lessons learnt from this eradication will form part of a full project review planned by RSPB.

A lack of work on the genetic resistance of black rats to bromadiolone posed a potential problem, since it was not possible from the start to confidently predict whether alternative bait types might be needed. However, the consistent lack of rat sign across the islands following baiting with bromadiolone, and zero take of the alternative rodenticide, indicates that there was no genetic resistance of rodents to bromadiolone at the Shiant Isles.

The project has contributed to building UK capacity for delivering rat eradications, biosecurity and incursion response through its training of staff. Local community members at the Western Isles were involved in bait station assemblage, service provision (e.g. boats) and volunteer work during eradication operations. Providing safe breeding habitat and maintaining rodent-free status at important island sites will be an important part of the long-term legacy of protection for UK seabirds.

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REFERENCES

Atkinson, I.A.E. (1985). 'The Spread of Commensal Species of *Rattus* to Oceanic Islands and their Effects on Island Avifaunas'. In: P.J. Moors (ed.) *Conservation of Island Birds*, pp. 35–81. ICBP Technical Publication 3. Cambridge, UK: BirdLife International.

Bell, E. (2013). *Shiant Isles Project: Feasibility Study and Operational Plan for the Eradication of Black Rats (*Rattus rattus*) from the Shiant Isles, Hebrides, Scotland*. Unpublished report. Blenheim, New Zealand: Wildlife Management International Ltd.

Bell, E. and Boyle, D. (2015). *Black Rat Eradication, Shiants Seabird Recovery Project, Shiant Isles, Scotland. Pre-assessment Visit 23 June–9 July 2015*. Unpublished report. Blenheim, New Zealand: Wildlife Management International Ltd.

Bell, E.A. (2019). 'It's not all up in the air: the development and use of ground-based rat eradication techniques in the UK'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 79–87. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Foster, P (2000). 'Shiant Islands Project.' In: P. Milburn (ed.) *Discovery and Excavation in Scotland NS Vol. 1 Council for Scottish Archaeology*, pp. 95–96. Edinburgh, UK: Archaeology Scotland.

Garg, N. and Singla, N. (2014). 'Toxicity of second generation anticoagulant bromadiolone against *Rattus rattus*: individual and sex specific variations'. *Cibtech Journal of Zoology* 3(2): 43–48.

Gibb, F.G.F. and Henderson, C.M.B. (1996). 'The Shiant Isles main sill structure and mineral fractionation trends'. *Mineralogical Magazine* 60: 67–97.

Jones, H.P., Tershy, B.R., Zavaleta, E.S., Croll, D.A., Keitt, B.S., Finkelstein, M.E. and Howald, G.R. (2008). 'Severity of the effects of invasive rats on seabirds: A global review'. *Conservation Biology* 22(1):16–26.

Key, G. Fielding, A.H. Goulding, M.J. Holm, R.S. and Stevens-Woods, B. (1998). 'Ship rats *Rattus rattus* on the Shiant Islands, Hebrides, Scotland'. *Journal of Zoology* 245: 228–233.

Love, J.A. (1983). *The Return of the Sea Eagle*. Cambridge, UK: Cambridge University Press.

Meehan, A.P. (1978) 'Rodenticidal activity of bromadiolone – a new anticoagulant'. In: W.E Howard and R.E. Marsh (eds.) *Proceedings of the Eighth Vertebrate Pest Conference 1978*, pp. 122–126. Davis, CA: University of California.

Mitchell, P.I., Newton, S., Ratcliffe, N. and Dunn, T.E. (eds.) (2004). *Seabird Populations of Britain and Ireland: results of the Seabird 2000 census (1998–2002)*. London, UK: T and A.D. Poyser.

Murray, S. and Shewry, M. (2002). *A Baseline Census of Manx Shearwaters on Rum National Nature Reserve in 2000 and 2001.* Commissioned Report to Scottish Natural Heritage, Inverness, UK: Scottish Natural Heritage.

Pelz, H-J., Rost, S., Hünerberg, M., Fregin, A., Heiberg, A-C., Baert, K., MacNicoll, A.D., Prescott, C.V., Walker, A-S., Oldenburg, J. and Müller, C.R. (2005). 'The genetic basis of resistance to anticoagulants in rodents'. *Genetics* 170: 1839–47.

Prescott, C.V., Buckle, A.P., Gibbings, J.G., Allan, E.N.W., and Stuart, A.M. (2010). 'Anticoagulant resistance in Norway rats (*Rattus norvegicus* Berk.) in Kent – a VKORC1 single nucleotide polymorphism, tyrosine139phenylalanine, new to the UK'. *International Journal of Pest Management* 57(1): 61–65.

Ratcliffe, N., Mitchell, I., Varnham, K., Verboven, N. and Higson, P. (2009). 'How to prioritise islands for the benefit of petrels: A case study of the UK, Channel Islands and Isle of Man'. *Ibis* 151: 699–708.

Sridhara, S. and Krishnamurthy, T.R. (1992). 'Potentiation of anticoagulant toxicity to *Rattus rattus* by two non-steroid anti-inflammatory drugs'. In: J.E. Borrecco and R.E. Marsh (eds.) *Proceedings of the Fifteenth Vertebrate Pest Conference 1992*, pp. 212–217. Davis, CA: University of California.

Stapp, P (2002). 'Stable isotopes reveal evidence of predation by ship rats on sea birds on the Shiant Islands, Scotland'. *Journal of Applied Ecology* 39: 831–840.

Taylor, P., MacMinn, R., Sophie Marsh, S., Brooke, M.de L., Macdonald, M., Hughes, P., Donald, F., Docherty, H., Over, A., Beaton, J., Scridel, D., Robertson, L. and Currie, N. (2018). Seabird population trends on the Shiant Isles, Outer Hebrides, 2000–2015. *Scottish Birds* 38(1): 3–14.

Thomas, S., Brown, A., Bullock, D., Lock, L., Luxmoore, R., Roy, S., Stanbury, A. and Varnham, K. (2017a). 'Island restoration in the UK – past present and future'. *British Wildlife* 28: 231–242.

Thomas, S., Varnham, K., and Havery, S. (2017b). *Current Recommended Procedures for UK (Bait Station) Rodent Eradication Projects (Version 4.0).* Sandy, Bedfordshire: Royal Society for the Protection of Birds.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats been exaggerated?' *Biological Invasions* 8: 863–91.

Walker, M.A. (1930). 'The geology of the Shiant Isles'. *Quarterly Journal of the Geological Society* 36: 355–398.

The effect of Norway rats on coastal waterbirds of the Falkland Islands: a preliminary analysis

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Abstract The Falkland Islands have been affected by anthropogenic-induced habitat modification including introduction of invasive species and grazing by livestock. Introduced Norway rats are known to have a large effect on native Falklands passerines but their effect on other native birds has not been explored. We investigated the effects of several environmental variables, including the presence of Norway rats and chronic grazing by livestock, on an assemblage of 22 species of coastal waterbirds by comparing species richness and relative abundance of birds among 65 rat-infested islands, 29 rat-eradicated islands and 76 historically rat-free islands. Bird count data from 299 km of coastline were used to estimate relative bird abundance, expressed as the number of individuals per kilometre of coastline for each species. Our study provided three key results. First, coastal waterbird abundance on islands historically without rats was twice as high as that on islands where rats were present. Second, bird abundance on rat-eradicated islands was intermediate between that of historically rat-free and rat-infested islands. Third, habitat modification by grazing appeared to reduce bird abundance in both rat-free and rat-infested habitats. From a conservation perspective, this study suggests that rat eradication programmes in the Falkland Islands are effective at restoring coastal waterbird abundance and would be even more so if carried out in conjunction with restoration of native coastal plant communities.

Keywords: cat, ecotype, eradication, grazing, marine, mouse, Norway rat, *Poa flabellata*, relative abundance, tussac, warrah

INTRODUCTION

The Falkland Islands are an archipelago of 477 islands (Falkland Islands Government, 2014) that differ in size, habitat modification and presence of introduced species. This creates unique opportunities to examine the effect of anthropogenic factors and stochastic events on the distribution and abundance of native species (Hall, et al., 2002; St. Clair, et al., 2011).

Human colonisation of the islands in the late 1700s led to considerable changes in native flora and fauna. Grazing by livestock caused a reduction of almost 80% in the coastal grasslands of tussac (*Poa flabellata)* (Strange, et al., 1988), which greatly affected bird species and populations (Strange, 1992; Woods, 1984; Woods & Woods, 2006). Major changes to bird populations are also attributed to the introduction of Norway rats (*Rattus norvegicus*)*,* black rats (*R. rattus*), house mice (*Mus domesticus*), cats (*Felis catus*) and Patagonian foxes (*Lycalopex griseus*) (Woods & Woods, 2006; Falklands Conservation, 2006). Norway rats are now present on about half of the archipelago's islands (Tabak, et al., 2015a) and are known to have a large effect on the abundance and diversity of Falklands' native passerine species (Hall, et al., 2002; Tabak, et al., 2015b), while the black rat has been recorded on one island only. Cats are known to prey on thin-billed prions (*Pachyptila belcheri*) (Matias & Catry, 2008), although nothing is known of their impact on other Falkland bird species. Mice impact small burrowing petrels and some passerine species (Rexer-Huber, et al., 2013) and Patagonian foxes reduce the breeding success of coastal waterbirds (Poncet, 1998). Prior to the arrival of these invasive predators, the only terrestrial mammal was the endemic warrah or Falklands wolf (*Dusicyon australis*)*.* Restricted to the two largest islands (East Falkland and West Falkland), this native canid may have been present for at least 70,000 years before being exterminated in 1876; craniodental evidence and first-hand accounts indicate that the warrah was an efficient predator, subsisting on penguins, geese and seals (Slater, et al., 2009). Beyond all doubt, it must have had a major impact on the distribution and abundance of all wildlife species on East Falkland and West Falkland.

The effects of introduced predators and their removal on passerine and seabird species have been relatively well documented for many islands world-wide (Ebbert & Byrd, 2002; Courchamp, et al., 2003; Rauzon, 2007; Kurle, et al., 2008; Towns, 2011; Veitch, et al., 2011), including the Falkland Islands (Woods, 1970; Strange, 1992; Woods & Woods 1997; Hilton & Cuthbert, 2010; Poncet, et al., 2011; Tabak, et al., 2014) where the eradication of Norway rats from 80 islands since 2001 provides a large-scale experiment for evaluating wildlife response to the removal of rats. Studies have shown that successful eradications result in a higher species richness of passerines (Hall, et al., 2002) and an increase in abundance of both passerines (Tabak, et al., 2015b) and invertebrates (St. Clair, et al., 2011). However, nothing is known of the impact of introduced predators on Falkland's shags, gulls, wildfowl, waders and birds of prey (referred to hereafter as coastal waterbirds) or of the response of these birds to rat eradication.

The evaluation of the subsequent recovery of native species and ecosystems following alien species eradication is an essential part of the process of determining the success of an operation (Courchamp, et al., 2011). In this study, we examined the potential effect of several environmental variables on the distribution and abundance of a coastal waterbird assemblage consisting of 22 species of ground-nesting, coastline-foraging birds. Using estimates of relative bird abundance (individuals per unit of coastline transect length) and species richness as the response variables, we compared bird populations on tussac islands that were historically rat-free, with those that were rat-infested and those where rats had been eradicated, and we examined bird response to habitat modification by grazing. We hypothesise that (1) the presence of rats would reduce bird abundance and species richness; (2) bird abundance on islands where rats had been eradicated would be higher than on rat-infested islands; and (3) long-term grazing of native habitats would reduce bird abundance and species richness.

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METHODS

Study area

The Falkland Islands archipelago (12,200 km^2) is situated approximately 500 km east of continental South America between latitudes 51°S and 53°S in the South Atlantic Ocean. It consists of the two large island masses East Falkland (6,480 km^2) and West Falkland (4,450 km^2), 475 smaller vegetated offshore islands (an island being defined as any vegetated land surrounded by water at low tide) and numerous associated rocks and stacks (Falkland Islands Government, 2014), totalling over 6,000 km of coastline. Surveys of all bird species were conducted on 168 offshore islands, and on East Falkland and West Falkland. The majority of islands included in this study were remote and uninhabited, requiring access by boat.

Coastlines surveyed were typical of most rocky offshore tussac islands with upper littoral and intertidal zones of gently to moderately sloping rock, shingle or boulder, occasional small sand beaches and short stretches of sheer cliff on exposed coasts (Strange, 1992). Coastline vegetation on ungrazed islands was dominated by the native grass tussac (*Poa flabellata*), which grows up to 3 m tall and forms dense canopies. On islands grazed by livestock, plant communities were dominated by short swards of grasses and herbs.

Data collection

Surveys were carried during the breeding season (September to May) between 2008 and 2014. All surveys were conducted in favourable weather conditions and by the same two observers (S. Poncet and K. Passfield). Environmental characteristics for each transect and each island, and the identity of each bird species and the number of birds detected were recorded following a standardised data collection protocol (Tabak, et al., 2015b). Surveyors walked along the coastline at a slow and consistent pace, noting birds that moved ahead or accompanied the surveyor to avoid counting the same bird multiple times. Counts were of adults and subadults; breeding status and social structure were also recorded. The geographical location of individuals, pairs and groups of birds was recorded using a hand-held global positioning system (GPS) receiver (Garminmap 62). Introduced predators were detected visually or by field signs typical for each species.

The sampling unit (transect) on each island consisted of a 100 m-wide swathe of coastline extending from approximately 20 m inland of the high tide mark out to approximately 80 m offshore. The inland distance of 20 m was determined on the basis that this is the maximum distance at which most birds would be visible or heard by an observer walking along the shoreline. Transect length was obtained from the surveyor's GPS track. Surveys involved walking at least 1 km of coastline.

For each transect we recorded the date, local time at the start and end of survey, transect length, observer name, wind speed and direction, cloud cover, temperature, precipitation, tide state, geographic region, grazing intensity and dominant vegetation.

For each island we recorded rat eradication status, island surface area and percent of island covered in tussac using data sourced from the Falkland Islands Biodiversity Database (Falkland Islands Government, 2014). Island coastline perimeter was obtained either from the observer's GPS track data or by using mapping software to measure coastlines on map sheets 1–29 (Directorate of Overseas Surveys, 1962).

Study species

All bird species (native, non-native, resident, vagrant and migratory) encountered on transects were recorded. The number of individuals of most species was also counted. Most shoreline species are of high to very high detectability and occur in habitats that are generally open to view from long distances (Woods & Woods, 1997), there being no trees or woodlands on the islands. Species that were not easily detected (burrowing petrels which nest underground) were not counted, and nor were penguins and black-browed albatross for which the coastline survey methodology was not suitable due to the amount of time required to count large concentrations of colonial seabirds.

In this study, 22 shoreline species were grouped to form an assemblage called coastal waterbirds. These were defined as any non-passerine species that relies on the littoral and sub-littoral zones for breeding, wintering and/or foraging, and they include waterfowl, waders, a bird of prey and some seabirds (Table 1). The majority of species are common and widespread around Falklands' coasts and forage predominantly on shoreline and inshore coastal habitats (Woods & Woods, 1997). Species detected on surveys that did not conform to this definition were passerines, southern giant petrels (*Macronectes giganteus*), upland geese (*Chloephaga picta*) and ruddy-headed geese (*C. rubidiceps*). The five introduced mammalian species recorded were feral cats, black rats, Norway rats, house mice and Patagonian foxes.

Data analysis

Relative abundance and species richness

Relative abundance of each coastal waterbird species for all 170 islands surveyed was estimated as the number of individuals counted divided by the total length of transects (in kilometres) walked on that island. Species richness of an island was defined as the total number of coastal waterbird species detected in transects on each island.

Ecotypes

The 170 islands were grouped into six different 'ecotype' categories based on several specific environmental variables (Table 2). Variables were the presence or absence of Norway rats (distinguishing between historically rat-free and rat-free following successful rat eradication), the presence or absence of heavy grazing, and the presence or absence of tussac-dominant vegetation along an island's coastline. The relative abundance and species richness for the coastal waterbirds was calculated for each of these ecotypes.

Effect of environmental variables on bird abundance and species richness

We modelled the relationship between response variables (total relative abundance and species richness of the coastal waterbird assemblage) and a number of potential driving environmental variables for a subset of 139 islands using Generalised Linear Models (GLMs). The predictor variables included in these models were the presence or absence of rats (excluding islands where rats had been eradicated), the presence or absence of heavy grazing, the percent of an island covered in tussac grass ("tussac cover"), island coastline perimeter and weather and tide at the time of survey. Data from the two largest islands East Falkland and West Falkland were excluded from these GLM analyses because the difference in coastline perimeter

Table 1 Summary of the abundance of all 42 passerine and non-passerine bird species recorded on coastal transects on 170 islands in the Falkland Islands.

Species	No. birds counted	No. of islands	% of islands
Falkland Island steamer duck* *Tachyeres bracypterus*	5,432	170	100
Kelp goose* *Chloephaga hybrida*	2,400	144	85
Blackish oystercatcher* *Haematopus ater*	752	140	82
Rock shag* *Phalacrocorax magellanicus*	5,668	131	77
Crested duck* *Anas specularioides*	2,143	133	75
Kelp gull* *Larus dominicanus*	1,726	125	74
Turkey vulture* *Cathartes aura*	687	120	71
Magellanic oystercatcher* *Haematopus leucopodus*	1,380	105	62
Tussacbird *Cinclodes antarcticus*	2,378	95	56
Upland goose *Chloephaga picta*	1,295	95	56
Black-crowned night-heron* *Nycticorax nycticorax*	670	93	55
Austral thrush *Turdus falcklandii falcklandii*	378	84	49
Black-chinned siskin *Carduelis barbata*	541	80	47
Dark-faced ground-tyrant *Muscisaxicola maclovianus*	364	70	41
Grass wren *Cistothorus platensis*	242	68	40
Dolphin gull* *Leucophaeus scoresbii*	762	60	35
Black-throated finch *Melanodera melanodera*	302	66	33
Striated caracara* *Phalcoboenus australis*	262	55	32
Cobb's wren *Troglodytes cobbi*	645	52	31
Snowy sheathbill* *Chionis albus*	796	51	30
King cormorant* *Phalacrocorax albiventer*	3,969	43	25
Long-tailed meadowlark *Sturnella loyca*	168	37	22
Ruddy-headed goose *Chloephaga rubidiceps*	145	35	21
Brown-hooded gull* *Larus maculipennis*	404	34	20
Southern giant petrel *Macronectes giganteus*	812	33	19
South American tern* *Sterna hirundinacea*	687	29	17
Falkland skua* *Catharacta antarctica*	97	27	16
Southern caracara* *Caracara plancus*	35	24	14
Two-banded plover* *Charadrius falklandicus*	332	21	12
Magellanic snipe* *Gallinago magellanica*	31	17	10
Speckled teal* *Anas flavirostris*	68	16	9
White-rumped sandpiper* *Calidris fuscicollis*	292	11	6
Rufous-chested dotterel* *Charadrius modestus*	71	8	5
Correndera pipit *Anthus correndera grayi*	14	6	<5
Red-backed hawk *Buteo polyosoma*	7	6	<5
White-tufted grebe* *Rollandia rolland*	5	4	<5
Peregrine falcon *Falco peregrinus*	3	3	<5
Silver teal *Anas versicolor fretensis*	10	2	<5
Chiloe wigeon *Anas sibilatrix*	4	2	<5
House sparrow *Passer domesticus*	68	1	<5
Domestic goose *Anser anser*	2	1	<5
Cattle egret *Bubulcus ibis*	1	1	<5
Total	36,048	170	

Species ranked by frequency of occurrence on the 170 islands surveyed.
* Species in the coastal waterbird assemblage analysed in this study.

Table 2 A comparison of relative abundance (mean ± s.e.) of the coastal waterbird assemblage counted using standardised surveys on 170 islands and six ecotypes during the period 2008–2014.

Island ecotype	No. birds counted	Birds/km	No. islands	Km of coastline	No. transects
I 'mainland'[1]	1,387	31 ± 2.6	2	44.98	18
II rat-infested tussac[2]	6,051	74 ± 4.8	57	82.27	64
III rat-free tussac[3]	11,775	156 ± 14.2	70	75.49	81
IV rat-eradicated tussac[4]	5,521	138 ± 9.8	29	40	32
V rat-infested non-tussac[5]	2,642	60 ± 4	6	43.97	23
VI rat-free non-tussac[6]	1,257	101 ± 7.3	6	12.41	10
Total	28,633		170	299.12	228

[1] Mainland East and West Falkland: grazed and/or massively modified by past grazing; Norway rats, mice and cats present; no Patagonian foxes.
[2] Tussac islands with Norway rats: tussac dominant along the coastline; not permanently grazed and/or not massively modified by past grazing; no cats, no ship rats, no Patagonian foxes, no mice.
[3] Tussac islands without Norway rats: tussac dominant along the coastline; not permanently grazed and/or not massively modified by past grazing; no cats, no ship rats, no Patagonian foxes, no mice.
[4] Tussac islands where Norway rats have been eradicated: tussac dominant along the coastline; not permanently grazed and/or not massively modified by past grazing; no cats, no ship rats, no Patagonian foxes, no mice.
[5] Non-tussac islands with Norway rats: little or no tussac; permanently grazed and/or massively modified by past grazing; no cats, no ship rats, no Patagonian foxes, no mice.
6 Non-tussac islands without Norway rats: little or no tussac; permanently grazed and/or massively modified by past grazing; no cats, no ship rats, no Patagonian foxes, no mice.

between these islands and the smaller islands resulted in a disproportionate influence on model output. The full model contained all covariates of interest, and an intercept term and was described as where the response variable was either relative abundance or species richness of the coastal waterbird assemblage. A Gaussian distribution with an identity link function was used to describe the relative abundance response variable while a Poisson distribution with a log link function was used to describe the species richness variable. We also ran all possible reduced models, containing all possible combinations using these covariates, and calculated Akaike Information Criterion corrected for small sample size (AIC_C) for each model. For each set of models, we conducted model averaging on the best models (those with $DAIC_C < 7$), to obtain a single ensemble model for relative abundance and species richness (Burnham & Anderson, 2002). All calculations and modelling were conducted using R v.3.2.2 (R Core Team, 2015).

The individual effects of each of the above covariates on the response variable were also analysed. Discrete covariates (rats and grazing) were analysed using Welch's two sample t-test; continuous covariates (percent tussac cover and coastline perimeter) were analysed using Analysis of Variance (ANOVA).

The effect of Norway rat status on bird abundance and species richness

We also analysed the relationship between response variables and the potential driving environmental variable of Norway rat status (rat-infested, rat-free and rat-eradicated) for a subset of 155 tussac islands of less than 10 km perimeter and less than 200 ha. Bird data for ecotypes II, III and IV (Table 2) were analysed using a Kruskal-Wallis test with a Bonferroni post-hoc test to assess the effect of rat eradications on bird abundance. A one-way ANOVA with Tukey HSD post-hoc test was used to assess ecotype effect on species richness (Table 3).

RESULTS

Data overview

A total of 42 bird species (of which 22 were classed as coastal waterbirds) and 36,000 individual birds were recorded along 299.12 km of coastal transects on East Falkland, West Falkland and 168 offshore islands (Table 1; Fig. 1). The majority (156) of these offshore islands were predominantly tussac-covered, uninhabited and ungrazed; the other 12 islands had little or no tussac cover and were

Table 3 Results of the Kruskal-Wallis test with Bonferroni post-hoc tests of effects of rat status on relative abundance and of the one-way ANOVA post-hoc tests of effects on species richness of coastal waterbirds, on 155 tussac islands of which 70 were historically rat-free, 57 were rat-infested at the time of survey and 28 had been successfully cleared of rats.

Coastal waterbird	Test used	Kruskal-Wallis and ANOVA results	Post hoc results (p-value; difference in mean)	
			Rat-infested vs rat-free	*Rat-infested vs rat-eradicated*
Relative abundance	Kruskal-Wallis with Bonferroni	$\chi^2 = 166.339$ df = 153.2 p = 0.000	0.000; -3.866	0.008; -2.794
Species richness	ANOVA with Tukey HSD	F (153.2) = 2.715 p = 0.069	0.485; 0.540	0.056; 1.417

Table 4 Model-averaged estimates for each parameter on relative abundance and species richness of the coastal waterbird assemblage on 139 islands for 214.15 km of coastline surveyed. Low magnitude of covariate values (i.e. values that are close to zero) indicate that the variable has a weak or insignificant effect on the response, while high absolute values indicate that the variable is important in predicting the response.

	Model-averaged estimates for coastal waterbirds				
	Intercept	rats (present)	heavy grazing (present)	coastline perimeter (km)	percent tussac cover
Relative abundance	140.07	-69.33	-4.46	-0.13	0.28
Species richness	8.07	0.04	-0.01	1.34	0.00

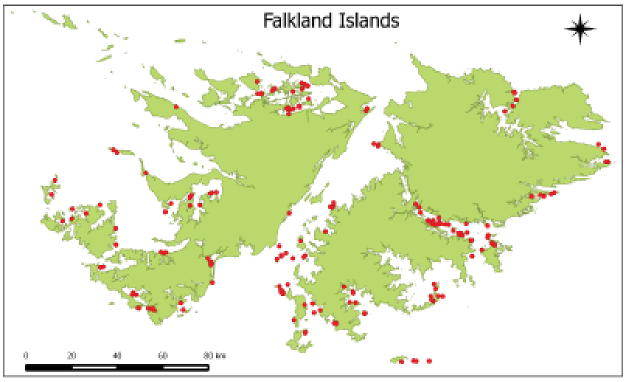

Fig. 1 The Falkland Islands showing islands where coastal bird surveys were undertaken.

grazed year-round by sheep (Table 2). Offshore islands ranged in size from 0.1 to 5,600 ha. The average length of coastline surveyed was 1.3 km; the entire coastline perimeter was surveyed on 58 of the smallest islands of less than 125 ha. Seventy six islands had one predator only (Norway rat); 63 had never had any introduced predators; 29 had been formerly occupied by one predator only (Norway rat) which had been subsequently eradicated and was absent at the time of bird survey. East Falkland and West Falkland, (collectively referred to as 'mainland') had cats, mice, Norway rats, permanent human settlements and year-round grazing by sheep and cattle. None of the islands had black rats or Patagonian foxes.

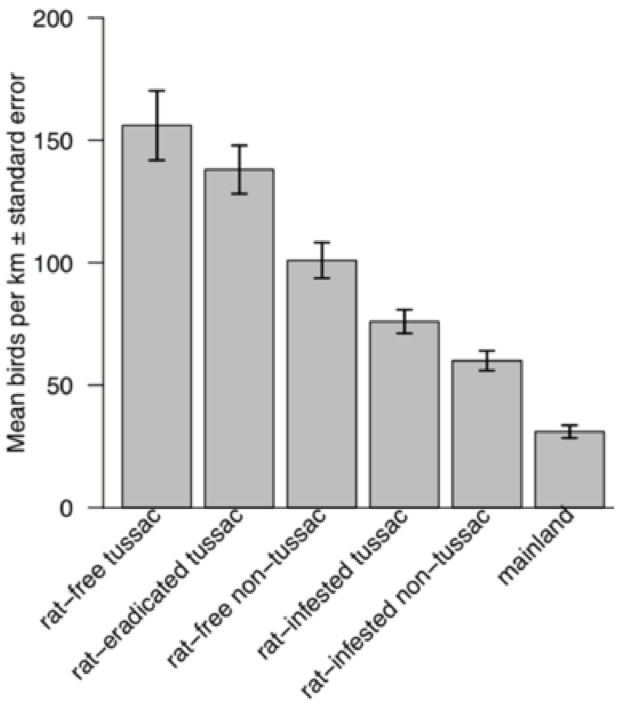

Fig. 2 The relative abundance (birds per kilometre of coastline surveyed) of the Falkland Islands coastal waterbird assemblage in six different ecotypes.

Coastal waterbirds accounted for 80% of all individual birds detected; species with the highest number of birds detected were the rock shag (*Phalacrocorax magellanicus*) and Falkland steamer duck (*Tachyeres brachypterus*), the latter being present on all islands (Table 1). Coastal waterbird abundance varied considerably between ecotypes. It was highest in predator-free tussac habitats, followed closely by rat-eradicated tussac habitat and lowest on the grazed (non-tussac) mainland coastlines where rats, mice and cats were present. It was twice as high in rat-free tussac habitat than in rat-infested tussac. A similar pattern emerged for non-tussac (i.e. grazed) islands where abundance in rat-free habitat was also nearly twice as high as that of rat-infested habitat. Overall, bird abundance in rat-free habitats, regardless of their tussac and grazing status, was higher than that in all rat-infested habitats. However, grazing also appeared to exert some effect, in that abundance was systematically lower in non-tussac habitat compared to tussac (Table 2, Fig. 2).

Effect of environmental variables on bird abundance and species richness

GLM analyses of the coastal waterbird relative abundance data from the subset of 139 islands (214.15 km of coastline) indicated that the model best supported by the data included the presence of rats (Table 4). Heavy grazing appeared to have a slightly negative effect on relative abundance; coastline perimeter of an island and the percent of an island covered in tussac had a negligible effect. Rats, heavy grazing and percent tussac cover had no effect on species richness. Coastline perimeter, however, may affect species richness, with more species being present on larger islands. The covariates depicting weather at the time of

Table 5 Results of Welch's two sample t-tests and ANOVA on the effect of rat presence, heavy grazing, percent of an island covered in tussac and coastline perimeter on the relative abundance of the coastal waterbird assemblage on 139 islands for 214.15 km of coastline surveyed.

	Welch's t-test		ANOVA
	t-value	p-value	F-value
Rats (present)	-4.63	0.0000	
Heavy grazing (present)	-2.85	0.0069	
Percent tussac cover			13.37
Coastline perimeter			6

Table 6 Results of Welch's two sample t-test and ANOVA on the effect of rats, heavy grazing, percent of an island covered in tussac and coastline perimeter on the species richness of the coastal waterbird assemblage on 139 islands for 214.15 km of coastline surveyed.

	Welch's t-test		ANOVA
	t-value	p-value	F-value
Rats (present)	0.47	0.6405	
Heavy grazing (present)	2.05	0.0513	
Percent tussac cover			0.77
Coastline perimeter			0.30

survey (i.e., wind speed, precipitation, cloud cover, and tide) did not appear in any of the best models (those with DAIC$_C$ < 7), so we concluded that they did not affect bird abundance or species richness. The effects of individual covariates analysed using Welch's two sample t-test results indicate a highly significant negative effect of both rats and heavy grazing on the relative abundance of coastal waterbirds. ANOVA results show that relative abundance increased significantly with percent tussac grass cover on an island and decreased with coastline perimeter (Table 5). Welch's two sample t-test and ANOVA results indicate that species richness is not affected by rats, heavy grazing, percent tussac cover on an island or island coastline perimeter (Table 6).

Effect of Norway rat status on bird abundance and species richness

Kruskal-Wallis tests with Bonferroni post-hoc tests showed that there was a significant effect of rat status on coastal waterbird abundance (Table 3). Bird abundance differed significantly between rat-infested and historically rat-free islands and between rat-infested and rat-eradicated islands, indicating that rat eradication resulted in an increase in coastal waterbird abundance. There was no significant difference in abundance between historically rat-free and rat-eradicated islands, which may possibly indicate that bird populations had nearly fully recovered following eradications. Results from the one-way ANOVA showed no effect of rat status on species richness.

DISCUSSION

The presence of Norway rats was the most important factor in predicting the relative abundance of the coastal waterbird assemblage (Table 4). Rat presence had a strong and significant negative effect on bird abundance (Tables 3, 4 and 5). This negative effect and the significant recovery benefits of rat eradication are like those observed for passerines (Hall, et al., 2002; Tabak, et al., 2015b). In contrast, rats did not affect species richness of the coastal waterbird assemblage.

Heavy grazing and the percent of tussac cover on an island also had significant negative effects on coastal waterbird abundance (Tables 2 and 3). Previous work has shown that grazing has a negative effect on bird abundance (Batáry, et al., 2007). In the Falklands, grazing led to the disappearance of the majority of the coastline's original vegetation of tall native grasses and shrubs which, in a landscape devoid of trees, provided optimal breeding habitat for the majority of the islands' bird populations. The impact of grazing is reflected in the higher bird abundance of rat-free tussac coastlines (ecotype III) compared to rat-free non-tussac (grazed) coastlines (ecotype V; Table 2).

An indication of the effectiveness of rat eradications in restoring coastal waterbird populations is shown by the large difference in bird relative abundance between rat-eradicated and rat-infested islands (Table 2). Abundance levels on the former are nearly twice as high as on the latter and approximate those of historically rat-free islands (Table 3) indicating that significant increases in coastal waterbird populations are likely when rats are eradicated. The importance of rat-free tussac islands (ecotype II) for Falkland bird populations is clearly demonstrated by the statistically significant difference in relative abundance of coastal waterbirds on these islands compared with rat-infested tussac islands (ecotype III) where relative abundance is 50% less. However, it is the coastlines of East Falkland and West Falkland (ecotype I where Norway rats, cats and mice are present and the impacts of grazing and destruction of native habitats are widespread), that show the largest response with a five-fold reduction in bird abundance. The impact that mice and cats exert on coastal waterbird species of the Falklands is largely unknown (Matias & Catry, 2008; Rexer-Huber, et al., 2013), although negative impacts have been assumed (Johnson & Stattersfield, 1990). Suggestions that the likely impact of the endemic warrah (*Dusicyon antarcticus*) on bird abundance was continued by later anthropogenic mammal introductions (notably cats) (Hall, et al., 2002) is an important consideration in any assessment of the potential of East Falkland and West Falkland for future vertebrate pest eradications.

CONCLUSION

Our study of the effect of environmental variables, and notably predators and over-grazing of native vegetation, on coastal waterbirds has identified the potential for differences in relative abundance of these species to serve as indicators of ecosystem recovery following rat eradications and habitat restoration activities. It is a first step in understanding the range of environmental factors that influence the distribution and abundance of Falkland coastal waterbirds. However, caution is required when interpreting differences as they may be caused not only by the balance of predation effects of Norway rats upon birds or by grazing impacts and habitat alteration but also by other indirect effects such as annual oceanographic and climate variations, biogeographical factors, island size and mesopredator release of birds (Watari, et al., 2011).

Future work will aim to determine the impact of these factors on individual species. Additionally, conducting repeated visits on islands will allow for future models to incorporate estimates of each species' detection probability. An improved understanding of how coastal bird distribution and abundance is affected by ecosystem processes is essential for informing future eradications in the Falkland Islands and for monitoring other large-scale landscape-level ecological changes to the Falklands coastline and its inshore marine environment.

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REFERENCES

Batáry, P., Báldi, A. and Erdős, S. (2007). 'Grassland versus non-grassland bird abundance and diversity in managed grasslands: Local, landscape and regional scale effects'. *Biodiversity and Conservation* 16: 871–881.

Burnham, K.P. and Anderson, D.R. (2002). *Model Selection and Multimodel Inference: A Practical Information-theoretic Approach*. New York, NY: Springer.

Courchamp, F., Chapuis, J.L. and Pascal, M. (2003). 'Mammal invaders on islands: Impact, control and control impact'. *Biological Reviews* 78: 347–383.

Courchamp, F., Caut, S., Bonnaud, E., Bourgeois, K, Angulo, E. and Watari, Y. (2011). 'Eradication of alien invasive species: surprise effects and conservation successes'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 285–289. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Directorate of Overseas Surveys. (1962). *The Directorate of Overseas Surveys 453 (Series H791).* UK: Directorate of Overseas Surveys.

Ebbert, S.E. and Byrd, G.V. (2002). 'Eradications of invasive species to restore natural biodiversity on Alaska Maritime National Wildlife Refuge'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 102–109. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Falklands Conservation. (2006). *Important Bird Areas of the Falklands*. UK: Falklands Conservation.

Falkland Islands Government. (2014). *The Falkland Islands Biodiversity Database*. Stanley.

Hall, J.R., Woods, R.W., Brooke, M.de L. and Hilton, G.M. (2002). 'Factors affecting the distribution of landbirds on the Falkland Islands'. *Bird Conservation International* 12: 151–167.

Hilton, G.M. and Cuthbert, R.J. (2010). 'The catastrophic impact of invasive mammalian predators on birds of the UK Overseas Territories: A review and synthesis'. *Ibis* 152: 443–458.

Johnson, T.H. and Stattersfield, A.J. (1990). A' global review of island endemic birds'. *Ibis* 132: 167–180.

Kurle, C.M., Croll, D.A. and Tershy, B.R. (2008). 'Introduced rats indirectly change marine rocky intertidal communities from algae- to invertebrate-dominated'. *Proceedings of the National Academy of Sciences* 105(10): 3801–3804.

Matias, R. and Catry, P. (2008). The diet of feral cats at New Island, Falkland Islands, and impact on breeding seabirds. *Polar Biology* 31: 609–616.

Poncet, S. (1998). 'Eradicating Foxes from Beaver Island'. In: J. Fisher-Smith, & L. Wallace (eds). *The Wool Press* 100, pp. 15–16. Stanley: Department of Agriculture.

Poncet, S., Poncet, L., Poncet, D., Christie, D., Dockrill, C. and Brown, D. (2011). 'Introduced mammal eradications in the Falkland Islands and South Georgia'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management. Proceedings of the International Conference on Island Invasives,* pp. 332–336. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

R Core Team. (2015). *R: A Language and Environment for Statistical Computing*. Vienna, Austria: R Foundation for Statistical Computing.

Rauzon, M.J. (2007). 'Island restoration: Exploring the past, anticipating the future'. *Marine Ornithology* 35: 97–107.

Rexer-Huber, K., Parker, G.C., Reeves, M., Stanworth, A.J. and Cuthbert, R.J. (2013). 'Winter ecology of house mice and the prospects for their eradication from Steeple Jason (Falkland Islands)'. *Polar Biology* 36(12): 1791–1797.

St Clair, J.J.H, Poncet, S., Sheehan, D.K., Sze´kely, T. and Hilton, G.M. (2011). 'Responses of an island endemic invertebrate to rodent invasion and eradication'. *Animal Conservation* 14: 66–73.

Slater, G.J., Thalmann, O., Leonard, J.A., Schweizer, R.M., Koepfil, K-P., Pollinger, J.P., Rawlence, N.J., Austin, J.J, Cooper, A. and Wayne, R.K. (2009). 'Evolutionary history of the Falklands wolf'. *Current Biology* 19(20): R937–R938.

Strange, I.J., Parry, C.J., Parry, M.C. and Woods, R.W. (1988). *Tussac Grass in the Falklands.* Port Stanley: Falkland Island Foundation Project Report.

Strange, I. J. (1992). *Wildlife of the Falkland Islands and South Georgia.* London: HarperCollins.

Tabak, M.A., Poncet S., Passfield K. and Martinez del Rio, C. (2014). 'Invasive species and land bird diversity on remote South Atlantic islands'. *Biological Invasions* 16: 341–352.

Tabak, M.A., Poncet S., Passfield K., and Martinez del Rio, C. (2015a). 'Modeling the distribution of Norway rats (*Rattus norvegicus*) on offshore islands in the Falkland Islands'. *NeoBiota* 24: 33–48.

Tabak, M.A., Poncet S., Passfield K., Goheen, J.R. and Martinez del Rio, C. (2015b). 'Rat eradication and the resistance and resilience of passerine bird assemblages in the Falkland Islands'. *Journal of Animal Ecology* 84: 755–764.

Towns, D.R. (2011). 'Eradications of vertebrate pests from islands around New Zealand: What Have we Delivered and What Have we Learned?'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 364–371. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Veitch, C.R., Gaskin, C., Baird, K. and Ismar S.M.H. (2011). 'Changes in bird numbers on Raoul Island, Kermadec Islands, New Zealand, following the eradication of goats, rats and cats'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 373–376. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Watari, Y., Caut, S., Bonnaud, E., Bourgeois, K. and Courchamp, F. (2011). 'Recovery of both a mesopredator and prey in an insular ecosystem after the eradication of rodents: a preliminary study'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 377–383. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Woods, R. W. 1970. The avian ecology of a tussac island in the Falklands Islands. *Ibis* 112: 15–24.

Woods, R. W. 1984. A census of breeding Falkland Islands passerine birds in tussac grass. UK: Unpublished report to the Falkland Islands Foundation.

Woods, R.W. and Woods, A. 1997. *Atlas of Breeding Birds of the Falkland Islands.* Oswestry, Shropshire, U.K.: Anthony Nelson.

Woods, R.W., and Woods, A. 2006. *Birds and Mammals of the Falkland Islands*. Hampshire: WILDGuides.

Applying lessons learnt from tropical rodent eradications: a second attempt to remove invasive rats from Desecheo National Wildlife Refuge, Puerto Rico

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Abstract The introduction of invasive rats, goats, and rhesus macaques to Desecheo National Wildlife Refuge, Puerto Rico led to the extirpation of regionally significant seabird colonies and negatively impacted plant and endemic reptile species. In 2012, following the successful removal of goats and macaques from Desecheo, an attempt to remove black rats using aerially broadcast rodenticide and bait stations was unsuccessful. A review of the operation suggested that the most likely contributors to the failure were: unusually high availability of alternative foods resulting from higher than average rainfall, and insufficient bait availability. In 2016, a second, successful attempt to remove rats was conducted that incorporated best practice guidelines developed during a workshop that focused on addressing the higher failure rate observed when removing rats from tropical islands. Project partners developed a decision-making process to assess the risks to success posed by environmental conditions and established go/no-go decision points leading up to implementation. Observed environmental conditions appeared suitable, and the operation was completed using aerial broadcast of bait in two applications with a target sowing rate of 34 kg/ha separated by 22 days. Application rates achieved on the ground were stratified such that anticipated high risk areas in the cliffs and valleys received additional bait. We consider the following to be key to the success of the second attempt: 1) monitoring environmental conditions prior to the operation, and proceeding only if conditions were conducive to success, 2) reinterpretation of bait availability data using the lower 99% confidence interval to inform application rates and ensure sufficient coverage across the entire island, 3) treating the two applications as independent, 4) increasing the interval between applications, 5) seeking regulatory approval to give the operational team sufficient flexibility to ensure a minimum application rate at every point on the island, and 6) being responsive to operational monitoring and making any necessary adjustments.

Keywords: bait availability, environmental conditions, operational monitoring, regulatory approval

INTRODUCTION

Tropical islands are rich in biodiversity but are susceptible to invasive species. Invasive species are the leading threat to island biodiversity (Tershy, et al., 2015), with invasive rodents known to be particularly harmful (Towns, et al., 2006). Eradications have been successful in removing invasive species (Veitch, et al., 2011), allowing island species to recover (Jones, et al., 2016), however there has been a greater record of success on temperate islands than on tropical islands (Russell & Holmes, 2015).

The two rodent eradication attempts (failed, then subsequently successful) on Desecheo Island, Puerto Rico offer an opportunity to explore the challenges of tropical rodent eradications. Here, we highlight the key changes that were made to the operational strategy during the second attempt, the role of the recently developed recommended best practices for tropical rodent eradications from Keitt, et al. (2015), and chronicle the recently confirmed successful project.

Study area

Desecheo is a small (117.1 ha) hilly island (18° 23’ N, 67° 29’ W) situated in the Mona Passage about 17 km offshore of the west coast of Puerto Rico (Fig. 1). Desecheo is composed of a peak of volcanic calcareous rock with a mosaic of grassy patches, shrublands, woodlands with candelabra cacti, and semideciduous forests dominated by *Bursera simaruba* (Woodbury, et al., 1971). The highest point is nearly 200 m with steep slopes ranging from 20 to 35 degrees

Historically, Desecheo was a major seabird rookery and in the early 1900s tens of thousands of seabirds nested on the island (Wetmore, 1918; Meier, et al., 1989) and it is home to three single-island endemic and two native reptile species (Evans, et al., 1991) and a US Endangered Species Act listed threatened cactus, higo chumbo (*Harrisia portoricensis*). Desecheo was originally set aside as a wildlife preserve in 1912, but the introduction of invasive goats (*Capra hircus*), rhesus macaques (*Macaca mulatta*), feral cats (*Felis catus*) and black rats (*Rattus rattus*), and human uses of the island, had a substantial impact on the island’s habitat, contributing to the collapse of the large seabird populations (Evans, 1989; Meier, et al., 1989).

In 1976, the island was transferred to the US Fish and Wildlife Service (USFWS) who currently manage it as

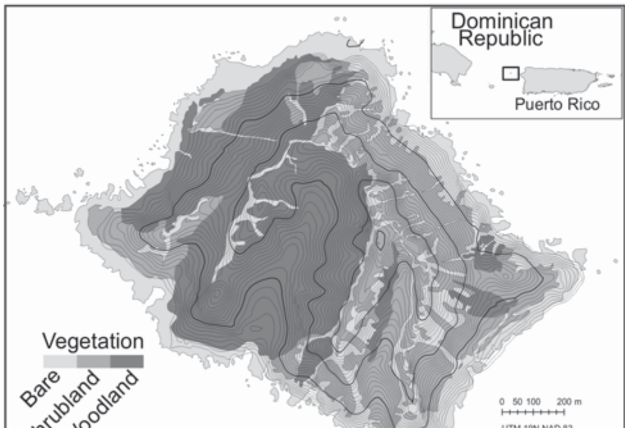

Fig. 1 Desecheo National Wildlife Refuge, located 17 km west of Puerto Rico in the Mona Passage.

In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) (2019). Island invasives: scaling up to meet the challenge, pp. 154–161. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Desecheo National Wildlife Refuge. To restore the island, the USFWS and collaborators removed feral cats in 1987 (Evans, 1989), feral goats in 2003 and the rhesus macaque population was reduced to being functionally extinct (i.e. reproduction ceased with only one wild macaque known to remain on the island) between 2009 and 2015 (Hanson, et al., 2019). In the absence of herbivory from goats and macaques, island species showed evidence of recovery, the higo chumbo resurged from the suppression caused by herbivory (Figuerola-Hernández, et al., 2017) and researchers detected seabirds prospecting for suitable habitat and attempting to nest on the main island in small numbers. However, recovery of the island ecosystem would not be possible until rats were removed.

MATERIALS AND METHODS

Project planning

Planning for the removal of black rats on Desecheo began in 2007 through the National Environmental Policy Act (NEPA) review process. The Finding of No Significant Impact (FONSI) identified aerial application of cereal bait pellets containing brodifacoum as the preferred alternative.

The ‘dry’ season from January to April was considered the ideal period for baiting because food for rats would be limited due to the dry environmental conditions and there was a higher likelihood of suitable weather for conducting an aerial application. Field trials were conducted in February and March 2009 and 2010 to evaluate rodent breeding status, presence of naturally occurring foods, abundance of non-target bait competitors, bait application rates, and detection capabilities of rodent surveillance devices.

Rats trapped during the 2009 (n = 33) and 2010 (n = 70) field trials indicated that rat reproduction appeared to be low during the dry season with no juvenile rats caught and no captured females showing signs of lactation or foetal development. The mean hermit crab density surveyed in 2010 was 696 crabs/ha but densities were higher in woodland sites (833 crabs/ha) than in shrubland sites (61 crabs/ha). Tomahawk live traps proved to be an effective surveillance device for rats with a 25% capture rate in 2009 and 55% in 2010.

In 2009, bait availability trials using a placebo biomarker in woodland habitat showed that bait applied at 18 kg/ha remained available in most plots for at least three days. The second trial in 2010 showed similar results for the same habitat. However, plots located on ridges in shrubland habitat exhibited a much faster rate of bait disappearance. Bait consumption by ants, considered to be in higher numbers on the island’s exposed ridgelines, was suspected to be one of the key factors driving this result. Monitoring during the trials also demonstrated, through non-toxic biomarker bait, that native and endemic reptiles could be exposed. Additionally, all surveyed hermit crabs in woodland sites tested positive for the presence of biomarker; this, together with high densities of crabs, indicated that hermit crabs would be a significant consumer of rodent bait.

Based on trial data, a bait application strategy was designed to achieve a bait density on the ground of 18 kg/ha during the first application followed approximately 10 days later by a second application targeting 9 kg/ha. Desecheo has a planar area (2-dimensional) of 117.1 ha including the offshore islets, and a topographical surface area (3-dimensional) of 134 ha; the surface area is 13% higher than the planar area. To account for the island’s steep topography, bait was sown at a rate of 20 kg/ha followed by 10 kg/ha to achieve the bait density required on the ground.

2012 eradication attempt

Brodifacoum Conservation 25-D (Bell Labs, Madison WI) 2 g pellets were applied aerially by helicopter in March 2012 using a spreader bucket slung below the helicopter. To minimise the risk of bait entering the marine environment, bait was applied along the coastal zone with a directional half swath bucket (deflector) and in the interior with a full swath starting and stopping inside of the coast. A full coastal swath was flown inland of the coast at the interface of the coastal and interior zones to provide sufficient overlap or ‘safety buffering’ and reduce the risk of bait gaps and areas of lower than target bait density (Fig. 2).

Additionally, to offset suspected ant consumption and supplement aerial broadcast in high risk areas bait stations were established at an interval of 25 m along two parallel transects on the ridgelines. Ant stations armed with Amdro®Pro fire ant bait (0.73% hydramethylnon) were placed within 1.5 m of each bait station. Stations were checked at least weekly and bait was replaced as needed for six weeks.

Bait availability transects were established across two of the same habitats as the trials (woodland and shrubland) measuring 1 × 25 m. The number of pellets in each transect was standardised and plots were sampled for seven consecutive days after each aerial broadcast or until all pellets had disappeared. At each visit, the number of pellets remaining was counted.

A captive programme was undertaken to hold representative samples of two endemic reptiles as a preventive action to reduce the risk of population-level impact from the application of rodenticide. A reptile mark-recapture monitoring study was done between February and April 2012 to confirm that the use of brodifacoum did not cause any observed population-level impacts in wild reptile populations on Desecheo (Herrera Giraldo, et al., 2019).

A live rat was found and captured 12 days after the 2nd application at the field camp and a buffer of bait stations was deployed in trees surrounding the field camp. No bait take was observed, and no additional rats were seen during the next week staff were on island.

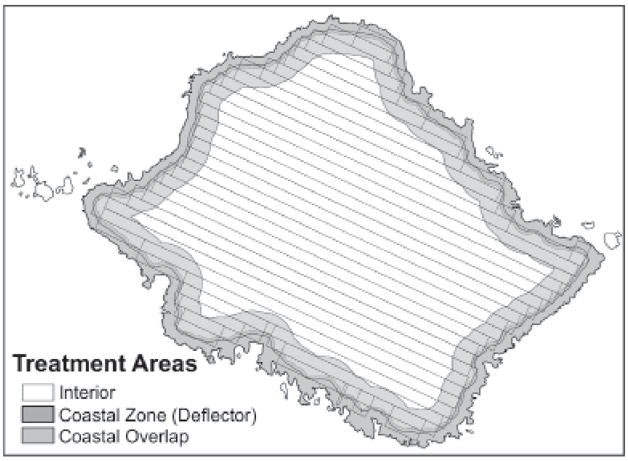

Fig. 2 Bait application strategy showing flight plan used to minimise bait into the marine environment with interior flight lines starting and stopping inside the coastal edge, a coastal half swath (deflector) along the coastal edge, and a full swath coastal overlap at the interface of the interior and coastal zones.

Eradication failure

Rats were not detected during fieldwork in October 2012 (six months post-operation), but in March 2013 (one-year post-operation) rats were observed and captured. Subsequent analysis of remote cameras deployed in 2012 showed the first rat detection in November 2012. Genetic testing indicated the eradication operation was not successful and the presence of rats was not the result of a reintroduction (i.e. operational failure).

To determine reasons why the operation may have failed, a review of the project investigated if rats *could not* eat a lethal dose because of gaps in bait coverage, insufficient bait availability, resistance to the toxin in the bait, or that the bait was not toxic enough; or if they *would not* eat a lethal dose of bait because of the palatability of the bait, availability of natural food resources, or breeding behavioural changes (i.e. pregnant females or emerging pups). Resistance to the toxin in the bait, bait toxicity, and bait palatability were not considered likely because the bait product had a proven record of success and rats captured during the biomarker trials showed a high level of acceptance. Despite implementing in the 'dry season', rainfall leading up to the operation was significantly higher than it was prior to either of the placebo bait trials, which may have resulted in a subsequent increase in the availability of natural food resources for rats and probable rat breeding. Bait disappeared quickly in several of the woodland plots with all bait disappearing within two to three nights of each application, likely the result of the significant crab densities in the woodland habitat. Finally, while there were few true gaps in bait coverage some areas during the first bait application received bait at less than half the prescribed rate. Thus, insufficient bait availability due to localised low bait densities during the first bait application and invertebrate bait competition, and an increase in the availability of natural food resources and rodent breeding due to above average rainfall, were identified as factors that could have individually or collectively contributed to the failure.

Tropical rodent eradication failures

About the same time that the 2012 attempt failed there were several other high-profile rat eradication failures on tropical islands, including Wake Atoll, western tropical pacific; Enderbury, Phoenix archipelago; and Henderson Island, Pitcairn group (Keitt, et al., 2015). A subsequent analysis of historical data showed that tropical rat eradications fail more than twice as often as temperate eradications (Russell & Holmes, 2015), resulting in a workshop attended by global experts to evaluate the possible reasons for this higher risk of failure and recommend solutions. The result of this workshop was a paper that provided recommended guidelines for rat eradications on tropical islands using aerial broadcast of brodifacoum (Keitt, et al., 2015).

Revised project approach

Starting in 2014, a steering committee of project partners (USFWS, USDA, and Island Conservation) was established to evaluate how to conduct a second attempt, the available strategy options, and how to manage ongoing project risk. A revised operational strategy was developed based on information from the review of the 2012 attempt and the recommended guidelines produced during the workshop on tropical rodent eradications (Keitt, et al., 2015). The following highlights the key changes:

1) Monitoring environmental conditions prior to the operation and proceeding only if conditions were conducive to success

A comprehensive review of factors influencing environmental conditions on Desecheo was conducted showing that rainfall and soil moisture content were key drivers of resource availability, typical of Puerto Rican subtropical dry forests. Inter-annual variability was evaluated using monthly rainfall totals and vegetation greenness, as a proxy for resource availability, between 2000 and 2013. Vegetation greenness was derived from remote sensing analyses using 30 m resolution 16-day MODIS Enhanced Vegetation Index (EVI) data. EVI data were smoothed using the HANTS algorithm (Roerink, et al., 2000) and mean monthly EVI were extracted from pixels that intersected the island using R (R Core Team, 2016).

Four assessments were conducted between three months and one week prior to implementation to evaluate the risk that short-term climatic changes could trigger higher biological productivity on the island prior to an irretrievable commitment of resources. Increased greenness represented more food availability via plants and invertebrates, and thus, increased opportunities for rodent breeding and increased bait competition due to invertebrate abundance. Each assessment included a review of regional climatic summaries, regional forecast products, and local weather conditions. Additionally, four island site visits were conducted to measure local rainfall, plant fruiting and flowering productivity, canopy cover and rodent breeding. To assist in data collection an automated logging rain gauge (WeatherShop, California, U.S.), three time-lapse cameras taking two photos per day (Day6Outdoors, Georgia, U.S.), and eight standardised photo point locations, were established on island.

A summary of conditions following each assessment was provided to the project steering committee for review. These summaries provided a subjective evaluation based on the team's knowledge of the island and the recommendations were used as part of a holistic evaluation of risk factors facing project implementation to make an operational go/no-go decision.

2) Reinterpretation of bait availability data

Using recent guidelines from Pott, et al. (2015) and data from the 2012 eradication attempt, bait availability was recalculated based on the lower-limit for a 99% t-based confidence interval. The linear rate at which bait disappeared was estimated by calculating the slope from four days of bait availability: 5.97 kg/ha per day in the woodland plots during the 2012 eradication attempt. This daily disappearance rate was used to calculate a conservative target bait density on the ground of 30 kg/ha to ensure that bait was available to rats for approximately five consecutive days after each application.

3) Treat the two applications as independent events

Following the guidelines outlined in Keitt, et al. (2015), the second attempt targeted the same application rate for each application and the target interval between application was increased to approximately 24 days. Two critical habitats, the valleys and steep cliffs identified in the review as areas of concern, were earmarked for additional supplemental bait application. On Desecheo, the predominant valleys and cliffs run perpendicular to one another, such that flights that are parallel to one are perpendicular to the other. To mitigate concerns about the impact on bait density caused by bait shadows on steep terrain (i.e. more bait downslope than upslope) and the higher rat and non-target bait consumer densities in the valleys observed during the 2012 attempt, additional flights were flown parallel to the valleys and cliff features to achieve higher application rates in these areas.

4) Seeking regulatory approval to give the operational team sufficient flexibility to ensure a minimum application rate at every point on the island

Some areas of Desecheo received bait below the desired bait density during the 2012 attempt. For the second attempt, the operational team sought regulatory approval to achieve a minimum bait density across every point on the island. This allowed for the retreatment of any areas that were estimated to be below the desired minimum bait density and limited the total amount of bait that could be applied per application rather than the application rate.

The project review also noted that the bait application strategy used to minimise bait in the marine environment created a risk of bait gaps and/or lower than planned baiting rates between coastal and interior zones. Regulatory approval was sought to ensure sufficient bait was available to treat the interface between the interior and coastal zones. This provided the flexibility to achieve the desired minimum bait density while also minimising bait entering into the marine environment.

5) Responding to operational monitoring in real time

To ensure quality coverage, bait sowing rates were carefully monitored and the helicopter shut down every five loads to download GIS files and review progress. During each application, a GIS specialist produced bait application maps estimating bait densities achieved on the ground. These data were used to identify any possible errors in flight lines, GPS logging, or bait application rates. Any gaps, identified as areas larger than 20 × 20 m receiving less than 15% (5 kg/ha) of the target bait density, were re-treated.

Greater emphasis was placed on operational monitoring than in 2012, including the deployment of additional bait availability monitoring transects and ground-truthing of bait application rates across the island. Additionally, communications between the environmental monitoring and bait application teams were improved by conducting the bait loading on island so that key project personnel were in the same place. In 2012 bait loading was done in Rincón approximately 17 km away on the main island of Puerto Rico. Following the first bait application, and prior to the second, a review of all operational data was conducted to allow for adjustments to the operational strategy.

2016 eradication attempt

The second eradication attempt was conducted in March and April 2016. The baiting strategy used was similar to the 2012 attempt albeit with an increased application rate and additional supplemental treatments along the cliffs and valleys. As in 2012, the sowing rate during the 2016 attempt was increased from 30 kg/ha to 34 kg/ha to accommodate the 3-dimensional surface area to ensure the desired bait density on the ground.

To allow comparisons with bait availability data collected during the previous field trials and the 2012 attempt, 25 m^2 sample transects were monitored in the woodland and shrubland habitats using the previous protocols. Additionally, a circular hoop sampling method (1 m^2) was used to estimate bait density on the ground following each application and collect additional bait availability across five different treatment zones.

Confirmation

In April 2017, one year after implementation, staff returned to the island and deployed chew tags, tracking tunnels, and live traps to confirm the absence of rats. Additionally, images from trail cameras were collected and analysed.

RESULTS

A summary of key differences between the two attempts is outlined in Table 1.

Environmental conditions

On first arrival at Desecheo Island on February 19, 2012, initial impressions were that the island's vegetation was more lush and green than observed during the same period in 2009 and 2010. Personnel recorded a total of 25.5 mm of precipitation on Desecheo between 10 March and 2 April 2012. Opportunistic necropsies of a small number of rats (n = 6) found dead during the 2012 operation showed one female rat with three embryos, and a male and the same female showed subjectively significant abdominal fat. However, the monitoring team did not observe any small juvenile rats suggesting breeding was not widely occurring for any prolonged period beforehand.

Retrospective analysis of precipitation recorded at the Rincón, Puerto Rico station (the closest point to Desecheo) showed that rainfall between January and March 2012 was above the annual average, and in February 2012 precipitation was 2.9 times higher than the 34-year average and the third highest rainfall for the month of February since 1968 (NOAA, 2015). Further, the remote sensing

Table 1 Summary of key differences between the 2012 and 2016 eradication attempts on Desecheo.

Factor	2012	2016
Month	March	March/April
Rainfall 6 months prior	4603 mm	772 mm
Rainfall during	25.5 mm	35.56 mm
Rodent breeding	One pregnant female observed (n=6)	None observed (n=44)
Canopy Cover	Flush vegetation	Post-peak vegetation followed by unproductive flowering after 31 mm rain event
Target bait density	18 kg/ha, 9 kg/ha	30 kg/ha, 30 kg/ha
Average application rate	17.1 kg/ha, 9.1 kg/ha	40.3 kg/ha, 39.9 kg/ha
Interval between applications	9 days	22 days

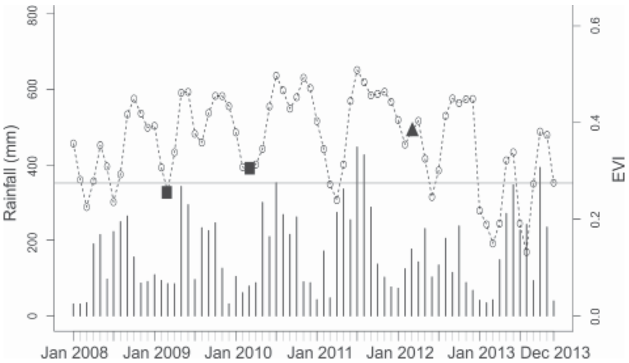

Fig. 3 MODIS Enhanced Vegetation Index (EVI) showing vegetation greenness vs rainfall in inches from Aguadilla, Puerto Rico. Dashed line represents EVI where higher EVI means greener and lower EVI means drier vegetation. Stacked lines represent rainfall in millimetres. The solid horizontal line represents the mean EVI for March, black squares EVI during placebo trials and black diamond EVI during the 2012 eradication.

analyses showed that March 2012 had significantly higher EVI than the same period during either of the 2009 or 2010 trials (Fig. 3).

During the assessment trips leading up to the 2016 attempt, observations showed that the island, while lush and green, was in post-peak greenness and starting to dry out in January 2016. This corresponded with a delayed and short wet season, likely the result of a record drought throughout much of Puerto Rico in 2015 (NOAA, 2015). By February there was a significant reduction in canopy cover; however, a significant rain event (31 mm in 24 hours) resulted in a large increase in canopy cover by March, mostly restricted to *Bursera* trees. Increased flowering was noticeable on some herbaceous shrubs and vine species; however, the fruits produced were not considered to be an alternative food source for rats. Increases in canopy cover continued through March, but by April 2016, after the second application of bait, most flower and fruit production had been abandoned. An irruption of caterpillars occurred after the second application, consuming much of the fresh *Bursera* growth (Shiels, et al., 2017).

Between 25 January and 10 April 2016, a total of 105.9 mm of precipitation was observed on Desecheo. Almost half of this precipitation was the result of two single events. In comparison, Rincón received a total of 239.3 mm of precipitation in the same period.

A total of 44 rats was captured during the 2016 attempt, all animals captured were adult size and none of the females showed signs of pregnancy, although some females showed indications (fat deposits and engorged uterine blood vessels) that breeding could have occurred soon after.

Bait application

During the 2012 eradication attempt, 3,588 kg of bait was applied on Desecheo as required by regulatory compliance, which resulted in an average application rate of 17.1 kg/ha and 9.1 kg/ha. An interval of nine days separated the first and second bait applications. Additionally, a total of 127 kg of bait was used in 107 bait stations placed along the ridges. The target application rates (18 kg/ha and 9 kg/ha) were at the upper limits allowed by regulatory requirements and the operational team was cautious in their approach with 1,000 kg of available bait unused. While the average application rates (total bait divided by island area) achieved were 17.1 kg/ha and 9.1 kg/ha, 76% of the island had a bait density on the ground below the target, with 8% less than half the target rate during the first application and 50% of the island below the target, with 4% less than half the target, during the second application.

In 2016, 10,650 kg of bait was applied according to regulatory compliance, resulting in an average application rate of 40.3 kg/ha and 39.9 kg/ha separated by 22 days. Regulatory approval was sought to allow for the retreatment of areas with less than the target bait density, ensuring a minimum bait density at every point across the island.

Table 2 Bait availability results from placebo trials and both eradication attempts on Desecheo. Bait availability is expressed as the 99% lower limit t-based confidence interval of mean bait availability to represent the “worst-case” scenario rather than the average case.

Habitat	Year	Plots	Target bait density (kg/ha)	Lower limit bait availability (kg/ha) after one day	Lower limit bait availability (kg/ha) after three days
1st Application					
Woodland	2009	6	18	0.5	0
	2010	9	18	3.2	0
	2012	5	18	6.9	0
	2016	6	45	7.8	0
Shrubland	2010	6	18	0	0
	2012	7	18	11.5	4.9
	2016	6	30	23.6	14.8
2nd Application					
Woodland	2009	6	18	-	-
	2010	9	18	-	-
	2012	5	9	0	0
	2016	8	45	36.0	24.0
Shrubland	2010	6	18	-	-
	2012	7	9	8.3	5.7
	2016	6	30	30.0	27.5

Using this strategy 11% of the island received less than the target bait density and 2% received less than half the target during the first application. During the second application 31% of the island received less than the target bait density and1% less than half the target.

A review of operational monitoring data following the first application during the 2016 attempt showed that bait disappeared faster than anticipated in the woodland valley habitat. In response, a total of 100 bait stations were installed in the valleys spaced at least 25 m apart. Bait stations were filled the day after aerial broadcast and elevated in trees wherever possible to reduce bait take by crabs. Each station was checked and replenished (if needed) three times during a three-week period. A total of 22.25 kg of bait was used in bait stations.

There was a small increase in the number of non-target carcasses observed between the 2012 (n = 4) and 2016 (n = 17) attempts, although a larger team was surveying the island for a longer duration in 2016. Few non-target species presented a high-risk exposure pathway so significant mortality was not expected following the 2016 attempt despite the increase in total bait applied to the island. Additionally, biological samples of rats, reptiles, and invertebrates were collected before and after the 2016 attempt to evaluate the persistence of brodifacoum in the environment four years after the 2012 attempt, if still detectable, and following the 2016 attempt, results of which will be reported elsewhere.

Bait availability

Observed bait availability was represented as the 99% lower limit confidence interval of mean bait availability (Table 2). The lower limit was used instead of mean availability to represent the "worst-case scenario" of bait availability rather than the average. During the first applications of both the 2012 and 2016 attempts, estimated bait availability reached zero in the woodland plots within three 24-hour periods despite the difference in application rates (Fig. 3).

Confirmation

A biosecurity monitoring trip was conducted seven months after the second attempt in November 2016 during which 10 A24 GoodNature traps, 40 bait stations, 10 Tomahawk live traps, 50 chew tags and 10 trail cameras were placed near possible landing sites. In April 2017, a total of 179 chew tags, 22 tomahawk live traps, 21 trail cameras and 20 tracking tunnels were placed across the island over a nine-day period for a total of 1,074; 124; 3,108; and 114 detection nights, respectively. No signs of rats were detected on any device during either monitoring trip. Following confirmation, monthly biosecurity monitoring trips between September 2017 and March 2018 continued to check the surveillance devices with no detections of rats.

DISCUSSION

The failure of the rat eradication on Desecheo in 2012 provided an excellent opportunity to better understand the reasons for failure, build upon the lessons learnt from other failed projects, and design a second attempt that addressed the key challenges. Keitt, et al. (2015) lays out a suite of recommendations to increase the probability of success for tropical rat eradications using aerial broadcast of brodifacoum based on reviews of several failed projects and input from a large group of experts. Desecheo was the first of these failed projects to be implemented a second time and enables review of the operational changes that contributed to operational success.

Environmental conditions

On tropical islands rainfall is a key driver of primary productivity and resulting elevated vegetation density is associated with an increase in rodent population densities (Harper & Bunbury, 2015). Like other dry tropical islands, primary productivity and resulting resource availability in the dry season on Desecheo (January–April) can be variable and is highly dependent on the amount of soil water recharge generated from successive rainfall events in the previous year's wet season (July–December) and the timing and amount of rain during the dry season. Environmental conditions leading up to the 2016 attempt were drier than those in 2012, primarily as an artefact of long term drought conditions experienced in 2015, resulting in lower primary productivity, less resource availability, and lower probability of rodent breeding. We feel that these 'favourable' conditions contributed to project success and had conditions leading up to the 2016 attempt been like those observed in 2012 the project would have been postponed.

Even though environmental conditions, and their subsequent implications for project success, are difficult to predict, the subjective assessments conducted on Desecheo were critical to the steering committee's confidence in proceeding with the bait application. They provided an opportunity to critically evaluate project risk and, more importantly, considered the consequences of postponement in advance of a final go/no-go decision. Where possible, future projects can improve stakeholder confidence by identifying the primary environmental drivers that pose risks to project success and developing a process that evaluates these risks to inform a final go/no-go decision. Projects should identify the worst-case scenario of alternative resource availability, non-target bait competitor abundance and rodent breeding, and plan accordingly.

Desecheo was relatively easy to access during day trips, but the deployment of an automated rain gauge and time-lapse cameras and use of remote sensing data provided valuable information on climatic conditions that could be replicated on remote islands. On islands where variability in environmental conditions pose a risk to operational efficacy projects should consider using these tools and others to better evaluate these risks. At the very least, projects can improve the collective knowledge of the challenges facing tropical rodent eradications by documenting and reporting observed environmental conditions, and subsequent perceived risks, leading up to and during implementation.

Bait availability

The review of the first attempt identified inadequate overall or localised baiting rates as one of the more likely causes of failure to eradicate rats. As described in Keitt, et al. (2015) eradications should strive to make bait available to rats for at least four consecutive 24-hour periods to maximise the probability that all rats are exposed to a lethal dose. The interpretation of the bait availability data for the 2012 attempt used mean bait availability to determine sufficient bait availability rather than the lower limit of 99% confidence intervals. Reinterpretation of the placebo trials using the 99% lower limit confidence interval method estimated that with a rate of 18 kg/ha the lower limit of bait availability would reach zero within two to three days. This was further supported by data from the 2012 attempt where the lower limit of bait availability went to zero by the third day after bait application (Fig. 3).

During the 2016 attempt bait availability observed in the transect sampling (25 m^2) roughly followed observations from the 2012 attempt where bait disappeared more quickly

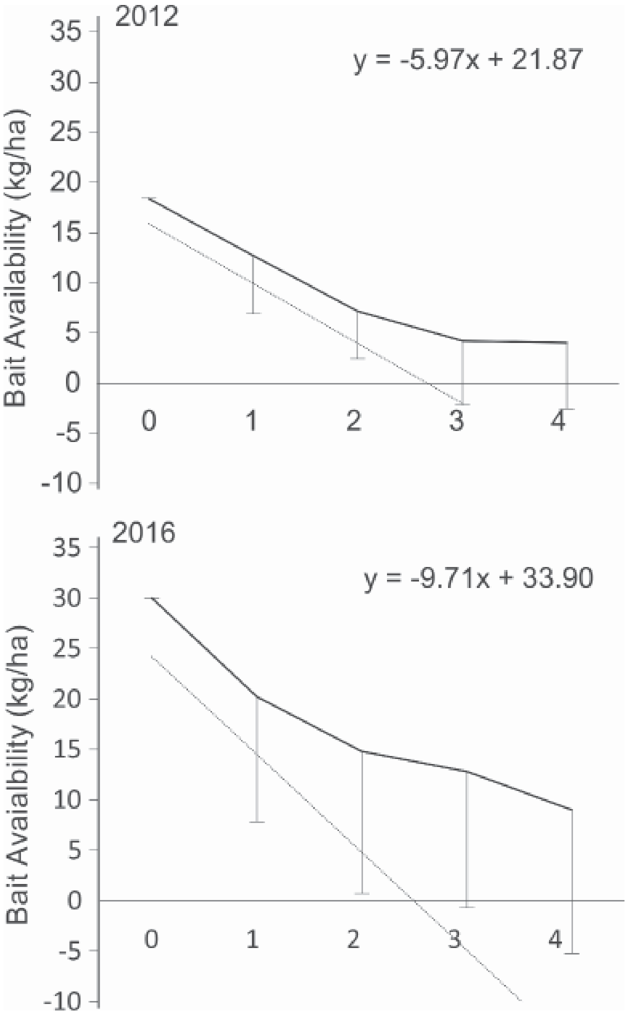

Fig. 4 The mean bait availability from the 25 m² bait availability plots set in the woodland habitats in 2012 and 2016. Day 0 represents the broadcast date and Day 1 represents the first 24-hour period during which bait is available to bait consumers (i.e. day 1 ends 24 hours after the end of bait application). The error bars represent the lower limit for a 99% t-based confidence interval. The trend line represents the bait disappearance rate based on the lower limit.

in the woodland than in the shrubland habitat. Although the revised strategy intended for bait to be available for at least four consecutive 24 hour periods after bait application, the lower limit of bait availability in the woodland plots was similar to the 2012 attempt and reached zero by the third day after bait application despite the nearly two-fold increase in application rate. This highlights some of the challenges tropical rodent eradications face in the presence of non-target bait competitors, supports the methods proposed in Pott et al. (2015) for evaluating bait availability data, and suggests that the higher application rate used in the 2016 attempt may have been necessary in the hermit crab-dense woodland habitat to ensure sufficient bait availability for all rats.

Regulatory approval

One of the criticisms of the 2012 attempt was that some areas received lower than the prescribed rates during the first application, particularly inside the coastal edge. This was potentially a consequence of the complex regulatory environment in the United States and the strategy employed to minimise bait spread into the marine environment. The desired target application rate on the ground was very near the maximum rate permitted by regulation and the operational team needed to strike a balance between achieving the desired application rate while staying within permitted limits. This was not an issue unique to Desecheo as, in general, aerial eradications conducted in the Unites States tend to use less bait than planned, compared to projects conducted elsewhere that use more than planned (Will, et al., 2019).

Leading up to the 2016 attempt the operational team aimed to address this challenge by engaging regulatory partners early in the project process as part of the project steering committee. The operational team justified, and sought approval for, a strategy that focused on achieving a site-specific minimum application rate based on the best available science. The justified strategy estimated the amount of bait needed to achieve a minimum rate at every point across the island, the amount of bait needed for overlapping flights necessary to minimise bait spread into the marine environment while minimising the chance of gaps along the coastal edge, and an additional amount of bait to fill unanticipated gaps and undertreated areas. Particularly in complex regulatory environments, future projects should consider seeking site-specific regulatory approval based on a justified strategy that maximises project success and bait quantities derived from a predicted flight plan.

Operational strategy

The justification for increasing the interval between applications was to reduce the risk posed by the scenario of pups emerging three weeks after the first application. The justification for using the same application rate in both applications was to ensure bait availability in the presence of non-target bait competitors. It should be noted that several tropical island eradications elsewhere have been successful with shorter gaps between bait applications. For example, in Mexico seven projects were successful with durations of 7–10 days between applications (Samaniego-Herrera, et al., 2014, 2017) even though rat breeding was confirmed. Additionally, an interval of three weeks could incur considerable operational costs while personnel and equipment are on standby. Alternatively, rodent breeding risks could be mitigated by increasing the application rate so that bait was available for a longer period, or conduct a third application; however, these would need to be balanced against associated non-target risk. As Keitt, et al. (2015) note, the recommendations should not be considered hard and fast rules as every island is different and we still have much to learn about tropical ecosystems.

The decision to apply additional bait in the valleys and on cliffs was based on the perceived risks justified from observations in 2012. These concerns appear somewhat validated because bait disappeared quickly in the woodland plots following the first application in 2016 despite the increased higher application rate from the 2012 attempt. It is difficult to evaluate what impact this strategy decision had on operational success but this stresses the importance of selecting bait application rates based on the best available science. Additionally, future projects should consider an additional treatment to increase confidence in areas of concern.

Operational monitoring

Intentionally slowing down the bait application in 2016 and reviewing bait density estimate maps improved the quality of the bait application and ensured that significantly less of the island was below the minimum bait density than during the 2012 attempt. Future projects should consider this strategy particularly on small islands where a single load treats a significant proportion of the island, and using bait density estimate maps to identify gaps or low treatment areas (Will, et al., 2019).

The second attempt put emphasis on near-real time information sharing to inform decision-making during the operation. While there is limited opportunity for adaptive management during aerial eradications, where success or failure is largely determined on the day, projects should put processes in place to ensure that data from the field are available to inform operational decision making and risk assessments during project implementation. Comprehensive operational monitoring allows managers to implement any available response options and, more importantly, allows stakeholders to understand project risk as the implementation unfolds.

CONCLUSION

Although we are unlikely to determine the influence environmental conditions, bait applications rates, or the interval between applications have on project success without experimentation, the variability in conditions observed on Desecheo during the 'dry' season and the consistently high rate of bait disappearance in crab-dense areas highlight the importance of understanding an island's ecosystem prior to implementing tropical eradications. The second attempt on Desecheo provided a significant opportunity to reconsider operational strategies for tropical eradications and marks the first of the high-profile failures to be successfully redone following the global review of tropical rodent eradications. The synthesis of recommended guidelines in Keitt, et al. (2015), and the process of reviewing project risks at pre-determined times, were necessary for increasing stakeholder confidence to make a second attempt. Ultimately, the rationale employed during the successful 2016 attempt should increase global confidence in rodent eradications on tropical islands.

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REFERENCES

Evans, M., Herbert, H.J. and Rohnke, K. (1991). 'Observations on the status of the herpetofauna of Desecheo Island National Wildlife Refuge, Puerto Rico'. In: J.A. Moreno (ed.) *Status y Distribución de los Reptiles y Anfibios de la Región de Puerto Rico. Publicación Científica Miscelánea No. 1*, pp. 34–36. San Juan, Puerto Rico: Departamento de Recursos Naturales.

Evans, M.A. (1989). 'Ecology and removal of introduced rhesus monkeys: Desecheo Island National Wildlife Refuge, Puerto Rico'. *Puerto Rico Health Sciences Journal* 8(1): 139–156.

Figuerola-Hernandez, C., Swinnerton, K.J., Holmes, N., Monsegur-Rivera, O., Herrera Giraldo, J.L., Wold, C., Hanson, C., Silander, S. and Croll, D. (2017). 'Resurgence of *Harrisia portoricensis* (Cactaceae) on Desecheo Island after the removal of invasive herbivores: Implications and recommendations for the management of the species'. *Endangered Species Research* 34: 339–347.

Hanson, C., Hall, T.J., DeNicola, A.J., Silander, S., Keitt, B. and Campbell, K. (2019). 'Rhesus macaque eradication to restore the ecological integrity of Desecheo National Wildlife Refuge, Puerto Rico'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 249–255. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Harper, G.A. and Bunbury, N. (2015). 'Invasive rats on tropical islands: Their population biology and impacts on native species'. *Global Ecology and Conservation* 3: 607–627.

Herrera-Giraldo, J.L., Figuerola-Hernández, C.E., Holmes, N.D., Swinnerton, K., Bermúdez-Carambot, E.N., González-Maya, J.F. and Gómez-Hoyos, D.A. (2019) 'Survival analysis of two endemic lizard species before, during and after a rat eradication attempt on Desecheo Island, Puerto Rico'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 191–195. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Jones, H.P., Holmes, N.D., Butchart, S.H., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E. and Burbidge, A.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences* 113(15): 4033–4038.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

Meier, A.J., Noble, R.E. and Raffaele, H.A. (1989). 'The birds of Desecheo Island, Puerto Rico, including a new record for Puerto Rican Territory'. *Caribbean Journal of Science* 25 (1–2): 24–29.

NOAA. (2015). *Climate Prediction Center: National Drought Predicition Center*. <http://www.cpc.ncep.noaa.gov/>. Accessed 11 November 2015.

Pott, M., Wegmann, A.S., Griffiths, R., Samaniego-Herrera, A., Cuthbert, R.J., Brooke, M.d.L., Pitt, W.C., Berentsen, A.R., Holmes, N.D., Howald, G.R., Ramos-Rendón, K. and Russell, J.C. (2015). 'Improving the odds: Assessing bait availability before rodent eradications to aid in selecting bait application rates'. *Biological Conservation* 185 :27–35.

R Core Team (2016). *R: A Language and Environment for Statistical Computing*. Vienna, Austria: R Foundation for Statistical Computing.

Roerink, G., Menenti, M. and Verhoef, W. (2000). 'Reconstructing cloudfree NDVI composites using Fourier analysis of time series'. *International Journal of Remote Sensing* 21(9): 1911–1917.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Samaniego-Herrera, A., Russell, J.C., Choquenot, D., Aguirre-Muñoz, A. and Clout, M. (2014). 'Invasive Rodents on Tropical Islands: Eradication Recommendations from Mexico'. In: R.M. Timm and J.M. O'Brien (eds.) *26th Vertebrate Pest Conference*, pp. 43–50. Davis: University of California.

Samaniego-Herrera, A., Aguirre-Muñoz, A., Bedolla-Guzmán, Y., Cárdenas-Tapia, A., Félix-Lizárraga, M., Méndez-Sánchez, F., Reina-Ponce, O., Rojas-Mayoral, E. and Torres-García, F. (2017). 'Eradicating invasive rodents from wet and dry tropical islands in Mexico'. *Oryx* 10.1017/s0030605316001150: 1–12.

Shiels, A.B., Haines, W.P., Swinnerton, K.J., Silander, S., Figuerola-Hernández, C., Will, D., García-Cancel, J.G. and Torres-Santana, C.W. (2017). 'Sudden appearance and population outbreak of *Eunica monima* (Lepidoptera: Nymphalidae) on Desecheo Island, Puerto Rico'. *Florida Entomologist* 100(1): 176–179.

Tershy, B.R., Shen, K.-W., Newton, K.M., Holmes, N.D. and Croll, D.A. (2015). 'The importance of islands for the protection of biological and linguistic diversity'. *BioScience* 65(6): 592–597.

Towns, D.R., Atkinson, I.A. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?' *Biological Invasions* 8(4): 863–891.

Veitch, C.R., Clout, M.N. and Towns, D.R. (eds.) (2011). *Island invasives: eradication and management.* Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Wetmore, A. (1918). 'The birds of Desecheo Island, Porto Rico'. *Auk* 35: 333–340.

Will, D., Howald, G., Holmes, N., Griffiths, R. and Gill, C. (2019). 'Considerations and consequences when conducting aerial broadcast applications during rodent eradications'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 71–78. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Woodbury, R.C., Martorell, L.F. and Tuduri, J.G. (1971). 'Flora of Desecheo Island, Puerto Rico'. *Journal of Agriculture of the University of Puerto Rico* 55(4): 478–505.

The eradication of black rats (*Rattus rattus*) from Dog Island, Anguilla, using ground-based techniques

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Abstract Rat eradication techniques developed in New Zealand are a proven method for removing invasive rodents from islands worldwide. This technology moved rapidly from ground-based bait station operations to aerial application of rodenticides. Rat eradications on tropical islands using similar methods, have not always been as successful as those in temperate regions. As most previous eradications in the Caribbean have been on islands smaller than 50 ha, the eradication of black rats (*Rattus rattus*) from 207 ha Dog Island was a significant increase in size. Reptile and seabird populations on Dog Island had been in decline for a number of years and black rats were identified as the most likely factor. Following the feasibility study in 2007, the Dog Island Recovery Project was launched in 2011. This was a multiple-year project incorporating a ground-based eradication with establishment of biosecurity procedures to prevent reinvasion, alongside long-term monitoring of native species. Bait stations with cereal-based wax blocks containing brodifacoum at 0.005% w/w were established on a 30–50 m grid over the island. Interference with bait stations by non-target invertebrates, particularly crabs, was high and bait stations required moving or elevating to avoid this. However, there was no evidence of any non-target animals being killed or injured by the bait. Eradication success was confirmed in 2014.

Keywords: biosecurity, black rat, brodifacoum, Dog Island, eradication, monitoring, *Rattus rattus*

INTRODUCTION

Dog Island (207 ha) is located 13 km north-west of Anguilla (18.2783°N, 63.2533°W) in the north-east Caribbean and consists of one main island and three smaller offshore cays, East, Mid and West Cay (Sanders, 2006; Hodge, et al., 2008). Designated as an Important Bird Area, the Royal Society for the Protection of Birds (RSPB), Anguilla National Trust (ANT) and Fauna & Flora International (FFI) have monitored the seabird colonies on Dog Island for nearly 10 years and recorded that seabird populations, particularly sooty terns (*Onychoprion fuscatus*) and magnificent frigatebirds (*Fregata magnificens*), had been declining (Campbell, 1991; Sanders, 2006; Holliday, et al., 2007; Hodge, et al., 2008; Daltry, 2010). Dog Island also has a much reduced endemic reptile community consisting of the Anguilla Bank ground lizard (*Pholidoscelis plei*), the Anguilla Bank tree lizard (*Anolis gingivinus*), two species of dwarf gecko (Anguilla Bank dwarf gecko (*Sphaerodactylus parvus*) and Leeward Island banded gecko (*S. sputator*)), and the Anguilla Bank skink or slipperyback skink (*Spondylurus powelli*); surveys in 2009 failed to observe any dwarf geckos or skinks (Hodge, et al., 2003; Daltry, 2010; Hedges & Conn, 2012). Black (ship) rats (*Rattus rattus*) were identified as the most likely factor influencing this decline through predation on eggs, and young or small individuals. Rats are known to have devastating effects on seabird and reptile populations, causing extinctions on numerous islands worldwide (Moors & Atkinson, 1984; Atkinson, 1985; Towns, et al., 2006; Jones, et al., 2008, Harper & Bunbury, 2015). Many islands have been successfully cleared of rats, including more than 30 in the Caribbean, with a subsequent increase in bird and reptile populations (Day & Daltry, 1996; Daltry, 2000; Daltry, et al., 2001; Thomas & Taylor, 2002; Towns & Broome, 2003; Jones, et al., 2008; Varnham & Daltry, 2006; Howald, et al., 2007; Varnham, 2010).

The Dog Island Restoration Project partnership (consisting of Anguilla National Trust (ANT), Anguilla Department of Environment (DOE), Fauna & Flora International (FFI), the Royal Society for the Protection of Birds (RSPB) and the island owner, Anguilla Development Company) commissioned the development of an operational plan to eradicate black rats from Dog Island in 2011 (Bell, 2011) based on an earlier feasibility assessment (Varnham, 2007). Wildlife Management International Limited (WMIL) directed the eradication with the assistance of international volunteers and ANT, DOE, FFI and RSPB staff. The three-phase Dog Island Recovery Project (Phase I eradication of black rats; Phase II long-term monitoring of native species and Phase III biosecurity) began in January 2012.

METHODS

Study area

Dog Island is a low-lying (highest point: 29 m asl), rocky island with three small offshore islets (Mid Cay, West Cay and East Cay). There are several long, sandy beaches, two saline ponds and the rest of the coastline is rocky or has low cliffs (< 8 m high). The island lies within a Marine Protected Area, covering an area of approximately 10 km^2 around the island (Hodge, et al., 2008). The island is popular with visiting yachts and tourist vessels from Anguilla or Saint Martin/Sint Maarten.

Dog Island was originally covered in dry forest or woodland, with shorter vegetation in coastal areas exposed to salt spray, but today is dominated by low, thorny scrub (e.g. *Lycium americanum* and *Castela erecta*) and prickly pear cacti (*Opuntia* spp.) due to herbivory by feral goats (*Capra hircus*). Larger trees including manchineel (*Hippomane mancinella*), sea grape (*Coccoloba uvifera*), white cedar (*Tabebuia heterophylla*) and buttonwood mangrove (*Conocarpus erectus*) can be found around the coastline and occasionally inland.

The island is recognised as an Important Bird Area because it is globally significant for a large number of breeding seabirds, in particular sooty terns and magnificent frigate birds, (Sanders, 2006; Hodge, et al., 2008). Other

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seabirds include brown boobies (*Sula leucogaster*), laughing gulls (*Larus atricilla*), masked boobies (*S. dactylatra*), brown noddies (*Anous stolidus*) and red-billed tropicbirds (*Phaethon aethereus*) (Holliday, et al., 2007; Hodge, et al., 2008; Daltry, 2010). The commonest land-birds on Dog Island are Caribbean elaenias (*Elaenia martinica*), bananaquits (*Coereba flaveola*) and black-faced grassquits (*Tiaris bicolor*). The island is also frequently used by migratory species travelling between North and South America (Holliday, et al., 2007; Daltry, 2010; Ross, 2011). A total of 48 resident and migratory bird species were confirmed on Dog Island by Richard Brown and Giselle Eagle from January to March 2012 (Bell, 2012).

Dog Island also has an important, albeit reduced, community of endemic lizards (Hodge, et al., 2003; Hedges & Conn, 2012). Notable missing reptiles are the globally threatened Anguilla racer (*Alsophis rijgersmaei*) and Lesser Antillean iguana (*Iguana delicatissima*), which were presumably present in the past. Dog Island would have been connected by a land bridge to Anguilla and indeed the rest of the Anguilla Bank well into the late Pleistocene, likely until 5,000 years ago. Three globally threatened species of marine turtles – hawksbill (*Eretmochelys imbricata*), green (*Chelonia mydas*) and leatherback (*Dermochelys coriacea*) – nest on and forage around Dog Island. Freshly dug holes have been recorded on many of the island's beaches (Hodge, et al., 2003; Daltry, 2010).

There have been few studies of the invertebrate fauna of Dog Island. Varnham (2007) collected samples using pitfall traps, but her specimens have not been identified to date (Daltry, 2010). Hermit crabs (*Coenobita clypeatus*) are present in high numbers, particularly around the coast.

Feral goats are present on the island; a remnant of more extensive grazing practices (Daltry, 2010). These goats are the property of the landowner but are hunted regularly, with or without permission. There are no known native mammals on Dog Island, but it could potentially support native bat species.

It is not known when black rats became established on Dog Island; but this is likely to have occurred sometime after 1613 when rats were first recorded in the Caribbean region (Harper & Bunbury, 2015). There is a history of human habitation on the island (i.e. stone walls and ruins) and rats may have reached Dog Island during this occupation or when ships were wrecked along the shores. Rats have been implicated as causing major impacts on island biodiversity (Towns, et al., 2006; Jones, et al., 2008) and they are known to have effects on important species on Dog Island. House mice (*Mus musculus*) have never been recorded on Dog Island (Varnham, 2007; Hilton & Connor, 2008; Bell, 2011).

Eradication operation

The eradication operation was planned to take place in the dry season (between January and May), when natural foods for the rats are in short supply and when there was little, or no risk of the operation being interrupted by tropical storms or hurricanes. The eradication option adopted for this project was a ground-based poison programme using protective bait stations to reduce risk to non-target species, particularly the reptile and feral goat populations. The eradication programme ran from 8 February 2012 to 4 April 2012 and included establishing the bait station grid, poisoning, monitoring and biosecurity. Biosecurity monitoring ran monthly between April 2012 and February 2014. The final check, species monitoring, and rat-free declaration ran from 10 to 19 February 2014. A core team of ten people completed the eradication, six people completed the biosecurity monitoring and a four-person team completed the final check. Each operational task was undertaken and completed as follows:

Bait station grid

A series of parallel tracks was cut through the vegetation on Dog Island, by a local contracting firm from Anguilla, between 20 November 2011 and 10 February 2012. Three additional lines were completed between 27 February and 6 March 2012. The contracting firm used two mechanised tools; machetes and rakes, to complete the task. One third of the island (the north-eastern end) had lines that were 30 m apart where the scrub was lower and easier to cut and the rest of the island, where scrub was much denser and more difficult to clear, had lines that were 40 m apart; bait stations were placed every 30 m along these lines. Areas of manchineel were not cut by the contracting firm (under arrangement with ANT as they did not want to deal with the toxic plant) during the track cutting phase of the project, but the main areas of manchineel were completed by the eradication team over a one-week period (10–15 February 2012) during the grid establishment phase, and two smaller stands of manchineel were completed over two four-day periods during the baiting phase (2–6 and 12–16 March 2012). Protective gloves and clothing and full-face masks were worn by the team when cutting tracks through the manchineel to avoid the sap and fumes which can irritate or blister the skin and cause breathing issues.

The bait station grid was established between 8 February and 15 February 2012. Bait stations were made from 1.5-litre plastic bottles (with the top and bottoms removed) donated by the public on Anguilla. These stations were pegged to the ground with wire "legs" to prevent movement by wind and/or stabilised with rocks or other material to reduce interference by feral goats. Bait was placed in the centre of the station through either end of the bottle.

Bait stations were placed out on the baiting grid. Mid and East Cays (not shown in the figures) were baited, but bait was laid on the surface (i.e. under vegetation and rocks) as feral goats were not present on the cays and there were few other non-target species present on these offshore islets. Each station was marked with flagging tape to ensure visibility in thicker vegetation.

The entire grid of 1,714 stations was established before being individually numbered and mapped using GPS and added to a GIS-linked database (Fig. 1).

Poisoning

Brodifacoum was used in two formulations: Klerat® (Syngenta, UK), a 20 g, wax-based wax block containing the bittering agent Bitrex™, and Pestoff® (Animal Control Products, NZ), a 24 g grain-based block bait. Both had 0.005% active ingredient and were dyed blue (or green/

Fig. 1 Bait station grid on Dog Island, Anguilla (bait station positions are marked by a black dot).

blue), to be less attractive to birds (Caithness & Williams, 1971; Hartley, et al., 1999; Weser & Ross, 2013).

The poisoning programme commenced on 16 February 2012 and continued through to 30 March 2012. Baits were present in each station throughout the poisoning programme and replaced as required. Two bait blocks were constantly available in each main island bait station throughout the programme. Klerat® was used as the main bait (16 February–20 March and 26–30 March). Pestoff® was only used for checks 20 and 21 (21–25 March) to target any surviving or rats that had avoided Klerat® for any reason.

The bait stations on Dog Island were checked and serviced every 1–4 days. However, the stations on the offshore cays were only checked twice, during suitable weather, on team changeover days when the boat was available. Thus, they had more bait per station (10 blocks) than on the main island. To present the data on bait-take gained from these varied bait station checks we grouped the data into 25 periods or checks (mean (±SEM) = 1.44 ± 0.14 days between checks, range 1–4 days) shown as days from baiting (Fig. 2).

Bait take was recorded in field notebooks by bait station number and the species believed to have consumed or removed the bait as confirmed by sign in and around the bait station (i.e. pieces or fragments with rat teeth marks or crab claw marks, etc.). These data were entered into a database and large-scale maps showing active stations were produced in real-time to enable the team to effectively monitor bait take activity and target any "hot spots". All rat carcasses found were collected and returned to base for incineration to reduce risk for non-target scavengers.

Mitigation measures such as using bait stations to prevent access by goats to the bait, moving bait stations if crabs interfered with the stations and raising the bait stations into vegetation were used to reduce the risk of primary and secondary poisoning to non-target species.

Monitoring

Three distinct periods of monitoring were undertaken as the project progressed. Monitoring points consisted of materials attractive to rats (e.g. chocolate flavoured wax or resin, candles and soap) and tracking tunnels. Intensive monitoring using 3,428 points at 15–20 m spacing was carried out from 12 March 2012 to 4 April 2012 to detect any surviving rats. This was followed by a 22-month period of long-term monitoring using 167 commercial lockable plastic bait stations (placed around the coastline of Dog Island as long-term biosecurity stations) from 5 April 2012 to 9 February 2014. These biosecurity stations were established at high risk areas on the island; around the coast and at seabird breeding sites (Bell, 2012). The final check, using 626 monitoring points and biosecurity stations, was carried out between 10 and 19 February 2014. WMIL, FFI and ANT staff and volunteers carried out the intensive and final checks and ANT staff and volunteers maintained the long-term monitoring. All stations were individually numbered and any evidence of activity (e.g. teeth marks or foot prints) was recorded in field notebooks by number and the species believed to have consumed the wax or soap or marked the tracking plate.

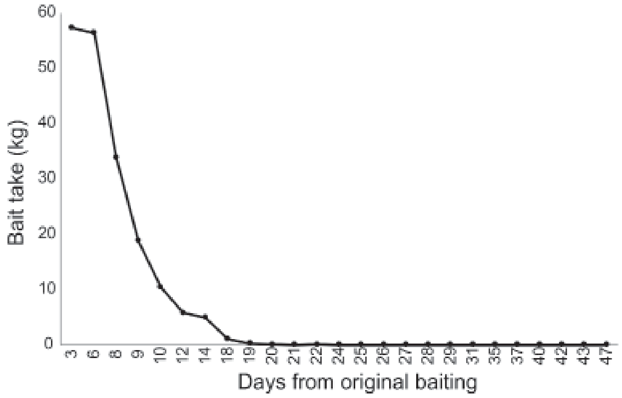

Fig. 2 Amount (in kg) of bait consumed by black rats (Rattus rattus) at each bait check (marked by black dot) during the black rat eradication on Dog Island, Anguilla (Day 1 was 16 February 2012).

Monitoring items were placed inside and outside each biosecurity station as well as halfway between each biosecurity station. Sand traps smoothed out to detect rat foot prints were established on beaches and inner island tracks. Checks for active rat runs and activity (i.e. identifying evidence of predation or scavenging on carcasses, chews on plants, droppings, etc.) at high-risk sites (i.e. ruins, seabird colonies, etc.) were also undertaken.

Each monitoring site was checked regularly, either separately, or during the poisoning phase, together with the poisoning bait station grid. Any rat and non-target species sign found on detection devices was recorded and added to the database.

RESULTS

Bait take was high over most of the island. Green/blue rat droppings appeared within three days and rats consumed 189 kg of bait. The bait take pattern was typical of other bait station rat eradication campaigns (Thomas & Taylor 2002). It was very high in the days immediately after the first bait loading (checks 1–3) and dropped to a relatively low level 20 days after initial baiting (check 10). A small increase was recorded at day 22 after initial baiting (check 12) but dropped away, reaching zero bait take on day 26 after the initial baiting (check 15) (Fig. 2).

Throughout the poisoning phase, 89% of bait stations were visited by rats, with 58% active within nine days of the initial baiting. The high number of active bait stations during the first two bait rounds shows that the rats quickly accepted the bait over most of the island.

The average number of blocks removed was 6.18 (± 0.07) blocks per station (Range: 0–16.05). As shown by Fig. 3, bait take was not evenly distributed over the entire island, with the greatest level of bait take at the eastern end where the main sea bird colony was situated, and the centre of the island. Bait take was also recorded on all the offshore stacks. Rats were also quickly eradicated from the cays as bait was still present when the second baiting visit was undertaken.

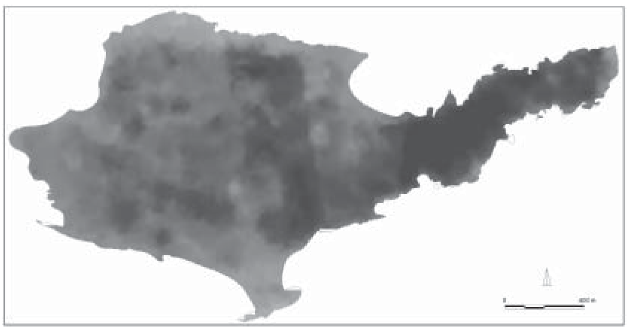

Fig. 3 Distribution of total bait take by black rats (*Rattus rattus*), as bait blocks consumed per station, during the black rat eradication on Dog Island, Anguilla. Darker shading indicates higher levels of bait uptake by rats.

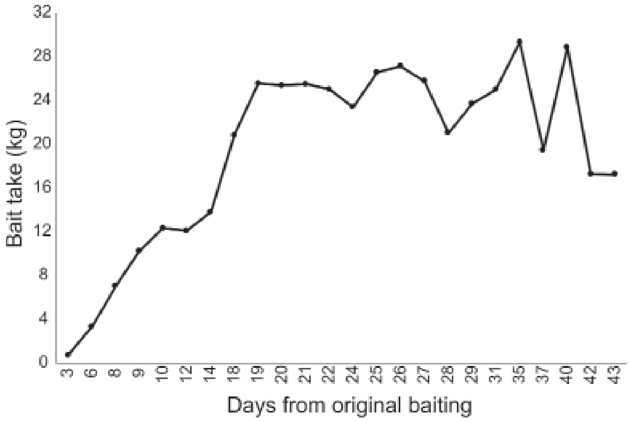

Fig. 4 Amount of bait (in kg) consumed by hermit crabs (*Coenobita clypeatus*) at each bait check (marked by black dot) during the black rat (*Rattus rattus*) eradication on Dog Island, Anguilla.

There was substantial interference by hermit crabs (Fig. 4). The average number of blocks eaten by crabs was estimated at 14.7 (± 0.3) blocks per station (Range 0–48.3). Crabs took a few days to become habituated to the bait, but then crab activity levels remained high (i.e. from days 19–41 over 50% of the stations were visited by crabs each round) (Fig. 4). There were 1,586 (92.5%) bait stations that had crab activity during the poisoning phase and only 128 stations were not affected. Crabs ate an estimated 467.4 kg of Klerat® bait throughout the eradication. There was no evidence that crabs were adversely affected by the bait. Anticoagulants are considered unlikely to affect invertebrates, as most have an open circulation system and have different physical and chemical clotting systems compared to vertebrates (Pain, et al., 2000).

Other non-target species had interfered with the bait to lesser amounts; goats consumed 0.04 kg (2 blocks) of Klerat® bait, ground lizards 0.26 kg (10.8 blocks) of Pestoff® bait, ants 10.9 kg of Klerat® bait and other insects 0.33 kg of Klerat® bait.

No animals, other than rats, exhibited signs of poisoning and no suspicious mortalities were recorded over the 11-week operation. The team was trained to observe non-target behaviour and collect any carcasses. There were 160 rat carcasses collected on the surface during the operation. These carcasses were collected and incinerated on the island to prevent availability to non-target species.

Monitoring for rat presence continued island-wide for two years after the end of the poisoning operation. The last rats were detected on 13 March 2012 during the overlap between the poisoning and intensive monitoring phases and these rats were successfully targeted using Klerat® by 30 March 2012. No rats or sign were detected during any phase of the long-term or final check monitoring. Dog Island was declared rat-free in May 2014.

DISCUSSION

The success of the Dog Island black rat eradication shows that a well-planned, adequately resourced, well-executed programme, supported by the landowner and directed by experienced operators and completed during the dry season can eradicate black rats from a large, arid, tropical island using a ground-based bait station operation. Dog Island is now the largest Caribbean island to be cleared of invasive rats and we believe that similar techniques could be utilised on other, even larger, islands in the Caribbean region.

Once the poison grid was established, the island was cleared of rats within four weeks (25 days from initial baiting). Bait-take showed that the rat population was not evenly distributed across the island. Apparently high concentrations of rats where the seabird colonies are present suggests rats were likely to have been having an effect on these nesting seabirds.

Importantly there were no known non-target species affected by this operation despite intensive searches for carcasses and a high level of interference by land crabs and to a lesser degree by ground lizards, invertebrates and, on one occasion, a goat which ate two blocks of bait. This stands in marked contrast to other operations that have inadvertently poisoned a variety of birds and other native wildlife (e.g. Howald, et al., 2007, Fisher, et al., 2011, Pitt, et al., 2015). Our choice of bait was a critical factor to this success; the primary bait used was Klerat® which was consistently untouched by any vertebrate other than rats (whereas the goat and lizards ate Pestoff® bait only). Klerat® has been equally successful in almost all the previous rat eradications in the Caribbean completed or managed by the authors and others (Day & Daltry, 1996; Daltry, 2000; Garcia, et al., 2002; Varnham, 2003; Varnham & Daltry, 2006; Witmer, et al., 2007; Varnham, 2010).

Ecological surveys conducted prior to the rat eradication operation identified a suite of ecological indicators on Dog Island that were consistent with the impacts of black rats, including the suppressed diversity and abundance of land birds, lizards and plants (Daltry, 2010). Audubon's shearwaters (*Puffinus lherminieri*) were first confirmed nesting on the island in 2012, within a few weeks of eradicating the rats (Bell, 2012). Preliminary surveys in 2014 found significant increases in a number of native species since the rats were eradicated; a two-fold increase in the density of ground lizards, three-fold increase in abundance of land-birds and a three-fold increase in burrow occupancy of Audubon's shearwaters (Bell & Daltry, 2014). Further increases were recorded during routine monitoring in 2016 and are predicted to continue over the next 10–20 years. Birds, lizards, goats, vegetation and invertebrates should be monitored for the next 20 years to detect and assess longer-term changes to the Dog Island ecosystem.

Unfortunately, as long as goats remain on Dog Island, some of the benefits of removing rats may be significantly reduced or fail to occur at all (Daltry, 2010). By preferentially eating all but the most spiny and toxic plants, the goats are maintaining an artificial, plagioclimax vegetation of thorny scrub across most of the interior of the island, which has low diversity and supports relatively few animals. Our cross-island transects, for example, revealed these interior areas had an extremely low density of lizards (Bell & Daltry, 2014). Another major concern about the goat herd is that it attracts parties of hunters who pose a biosecurity risk because their vessels and gear could provide pathways for rodents and other pests to invade the island.

While eradicating rats from Dog Island is a considerable achievement, it is important to stress that keeping this island rodent-free will require constant vigilance and commitment from all agencies, interested parties and the Anguillan community to prevent, detect and respond to any incursions. Prevention of rat re-infestation should be the primary aim. The greatest risk of reinvasion by rats reaching Dog Island is with private vessels, charter boats and fishing boats, particularly those that moor overnight, from Anguilla or the other nearby islands such as the Prickly Pear Cays and Saint Martin. This is especially so when equipment and food are brought to the island. Permanent biosecurity stations have been established on Dog Island and these will be maintained indefinitely by trained ANT staff. An incursion response plan has also been developed by ANT to deal with any rats that may be detected in the future. This shows that the local conservation agencies are totally committed to the restoration of this important Caribbean island.

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REFERENCES

Atkinson, I.A.E. (1985). 'The Spread of Commensal Species of *Rattus* to Oceanic Islands and their Effects on Island Avifaunas'. In: P.J. Moors (ed.) *Conservation of Island Birds,* pp. 35–81. ICBP Technical Publication 3. Cambridge, UK: International Council for Bird Preservation (ICBP).

Bell, E. (2011). *Dog Island Restoration Project: Operational Plan for the Eradication of Black Rats (*Rattus rattus*) from Dog Island, Anguilla*. New Zealand: Wildlife Management International Ltd.

Bell, E. (2012). *Dog Island Restoration Project: Technical Report for the Eradication of Black Rats (*Rattus rattus*) from Dog Island, Anguilla*. New Zealand: Wildlife Management International Ltd.

Bell, E. and Daltry, J. (2014). *Dog Island Restoration Project: Twoyear Assessment Following the Eradication of Black Rats (*Rattus rattus*) from Dog Island, Anguilla*. New Zealand: Wildlife Management International Ltd and UK: Fauna & Flora International.

Caithness, T. and Williams, G. (1971). 'Protecting birds from poisoned baits'. *New Zealand Journal of Agriculture* 122: 38–43.

Campbell, E. (1991). 'The effect of introduced roof rats on bird diversity of Antillean Cays'. *Journal of Field Ornithology* 62(3): 343–348.

Daltry, J. (2000). *The Second Rat Eradication on Praslin Island, Saint Lucia: August–September 2000.* Jersey: Saint Lucia Forest and Lands Department, Saint Lucia and the Durrell Wildlife Conservation Trust.

Daltry, J., Bloxam, Q., Cooper, G., Day, M., Hartley, J., Henry, M., Lindsay, K. and Smith, B. (2001). 'Five years of conserving the 'world's rarest snake', the Antiguan racer *Alsophis antiguae*'. *Oryx* 35: 119–127.

Daltry, J. (2010). *Observations on the Ecology of Dog Island, Anguilla, 2010.* Anguilla Development Company, Anguilla National Trust, Department of Environment (Government of Anguilla) and DEFRA-Fera (UK Government).

Day, M. and Daltry, J. (1996). 'Rat eradication to conserve the Antiguan Racer'. *Aliens* 3: 14–15.

Fisher, P., Griffiths, R., Speedy, C. and Broome, K. (2011). 'Environmental monitoring for brodifacoum residues after aerial application of baits for rodent eradication'. In: C.R. Veitch, M.N. Clout and D.R Towns (eds.) *Island invasives: eradication and management*, pp. 300–304. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Garcia, M., Diez, C. and Alvarez, A. (2002). 'The eradication of *Rattus rattus* from Monito Island, West Indies'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 116–119. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Harper, G. and Bunbury N. (2015). 'Invasive rats on tropical islands: Their population biology and impacts on native species'. *Global Ecology and Conservation* 3: 607–627.

Hartley, L., O'Connor, C., Waas, J. and Matthews, L. (1999). 'Colour preferences in North Island robins (*Petroica australis*): Implications for deterring birds from poisonous baits'. *New Zealand Journal of Ecology* 23: 255–259.

Hedges, S. and Conn, C. (2112). 'A new skink fauna from Caribbean islands (Squamata, Mabuyidae, Mabuyinae)'. *Zootaxa* 3288: 1–244.

Hilton, G. and Connor, R. (2008). *Mouse Detection on Dog Island*. Anguilla: Anguilla National Trust and UK: Royal Society for the Protection of Birds.

Hodge, K., Censky, E. and Powell, R. (2003). *The Reptiles and Amphibians of Anguilla, British West Indies*. The Valley, Anguilla: The Anguilla National Trust.

Hodge, K., Holliday, S. and Mukhida, F. (2008). 'Anguilla'. In: D. Wege, and V. Anadon–Irizarry (eds.) *Important Bird Areas in the Caribbean*, pp. 27–34. Cambridge, UK: BirdLife International.

Holliday, S., Hodge, K. and Hughes, D. (2007). *A Guide to the Birds of Anguilla.* Sandy, Bedfordshire, UK: Royal Society for the Protection of Birds.

Howald, G., Donlan, C., Galvan, J., Russell, J., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershey, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Jones, H., Tershy, B., Zavaleta, E., Croll, D., Keitt, B., Finkelstein, M. and Howald, G. (2008). 'Review of the global severity of the effects of invasive rats on seabirds'. *Conservation Biology* 22: 16–26.

Moors, P.J. and Atkinson, I.A.E. (1984). 'Predation on Seabirds by Introduced Animals and Factors Affecting its Severity'. In: J.P. Croxall, P.G.H. Evans and R.W. Schreiber (eds.) *Status and Conservation of the World's Seabirds,* pp. 667–690. ICBP Technical Publication 2. Cambridge, UK: International Council for Bird Preservation (ICBP).

Pain, D., Brooke, M., Finnie, J. and Jackson, A. (2000). 'Effects of brodifacoum on the land crabs of Ascension Island'. *Journal of Wildlife Management* 64: 380–387.

Pitt, W., Berentsen, A., Shiels, A., Volker, S., Eisemann, J., Wegmann, A. and Howald, G. (2015). 'Non–target species mortality and the measurement of brodifacoum rodenticide residues after a rat (*Rattus rattus*) eradication on Palmyra Atoll, tropical Pacific'. *Biological Conservation* 185: 36–46.

Ross, T. (2011) *Dog Island: Brief Information and Data*. Cambridge, UK: Fauna & Flora International.

Sanders, S. (2006). *Important Bird Areas in the United Kingdom Overseas Territories.* Sandy, UK: Royal Society for the Protection of Birds.

Thomas, B. and Taylor, R. (2002). 'A history of ground-based rodent eradication techniques developed in New Zealand, 1959–1993'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 301–310. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Towns, D. and Broome, K. (2003). 'From small Maria to massive Campbell: Forty years of rat eradications from New Zealand islands'. *New Zealand Journal of Zoology* 30: 377–398.

Towns, R., Atkinson, I. and Daugherty, C. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?'. *Biological Invasions* 8: 863–891.

Varnham, K. (2003). *Eradication of Black Rats (*Rattus rattus*) from Sandy Cay, British Virgin Islands.* UK: Royal Society for the Protection of Birds.

Varnham, K. and Daltry, J. (2006). *Eradication of Invasive Black Rats (Rattus rattus) from Green and York Islands, Antigua: June–July 2006'.* UK: Royal Society for the Protection of Birds and UK: Fauna & Flora International.

Varnham, K. (2007). *Eradicating Introduced Black Rats (*Rattus rattus*) from Dog Island, Anguilla: A Feasibility Study*. UK: Royal Society for the Protection of Birds.

Varnham, K. (2010). 'Invasive rats on tropical islands: their history, ecology, impacts and eradication'. *RSPB Research Report No. 41*. UK: Royal Society for the Protection of Birds.

Witmer, G., Boyd, F. and HillisStarr, Z. (2007). 'The successful eradication of introduced roof rats (*Rattus rattus*) from Buck Island using diphacinone, followed by an irruption of house mice (*Mus musculus*)'. *Wildlife Research* 34: 108–115.

Weser, C. and Ross, J. (2013). 'The effect of colour on bait consumption of kea (*Nestor notabilis*): Implications for deterring birds from toxic baits'. *New Zealand Journal of Zoology* 40(2): 137–144.

Recovery of introduced Pacific rats following a failed eradication attempt on subtropical Henderson Island, South Pacific Ocean

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Abstract Rodent eradications in tropical environments are often more challenging and less successful than those in temperate environments. Reduced seasonality and the lack of a defined annual resource pulse influence rodent population dynamics differently than the well-defined annual cycles on temperate islands, so an understanding of rodent ecology and population dynamics is important to maximise the chances of eradication success in the tropics. Here, we report on the recovery of a Pacific rat (*Rattus exulans*) population on Henderson Island, South Pacific Ocean, following a failed eradication operation in 2011. We assessed changes in the rat population using capture rates from snap-trapping and investigated seasonality by using capture rates from live-trapping. Following the failed eradication operation in 2011, rat populations increased rapidly with annual per capita growth rates, r, of 0.48–5.95, increasing from 60–80 individuals to two-thirds of the pre-eradication abundance within two years, before decreasing (r = -0.25 – -0.20), presumably as the population fluctuated around its carrying capacity. The long-term changes in rat abundance may, however, be confounded by short-term fluctuations: four years after the eradication attempt we observed significant variation in rat trapping rates among months on the plateau, ranging from 36.6 rats per 100 corrected trap-nights in mid-June to 12.6 in late August. Based on mark-recapture, we also estimated rat density fluctuations in the embayment forest between 20.4 and 42.9 rats ha^{-1} within one month in 2015, and a much lower rat density on the coral plateau fluctuating between 0.76 and 6.08 rats ha^{-1} in the span of two months. The causes for the short-term density fluctuations are poorly understood, but as eradication operations on tropical and subtropical islands become more frequent, it will be increasingly important to understand the behaviour and ecology of the invasive species targeted to identify times that maximise eradication success.

Keywords: introduced species, island restoration, Pitcairn Islands, rodents, spatially explicit capture-recapture

INTRODUCTION

The removal of introduced rodents from islands is an increasingly important tool for the conservation of island biodiversity, and has been successful in hundreds of cases (Lorvelec & Pascal, 2005; Howald, et al., 2007; Bellingham, et al., 2010; Russell & Holmes, 2015). Introduced rodents have been eradicated from >580 islands (Keitt, et al., 2015; DIISE, 2016) and rodent eradications are one of the most cost-effective methods of preserving island biodiversity (Howald, et al., 2007; Jones, et al. 2016).

The success rate of rodent eradications has improved as eradication tools and methods become more refined. However, failures still occur, especially on tropical islands where conditions that can increase the risk of eradication failure, such as aseasonal breeding, are more likely (Varnham, 2010; Holmes, et al., 2015). While undesirable, these unsuccessful projects still provide an opportunity to advance conservation science, often through *post hoc* review of operational planning and implementation (Keitt, et al., 2015). However, they also present potentially unique occasions to further understand invasion biology. For example, the population dynamics of surviving rodent populations following such failed eradication attempts are seldom studied (Hein & Jacob, 2015) despite being useful for predicting population dynamics during new invasions (Nathan, et al., 2015).

In particular, there is currently little knowledge on how much time elapses before tropical rodent populations can reach an island's presumed carrying capacity after a severe population bottleneck, but such information could be useful to inform the post-operation monitoring interval that determines whether an eradication operation has been successful or not (Samaniego-Herrera, et al., 2013). On temperate islands, two years encompasses two rat breeding seasons, and is typically sufficient to determine an eradication operation's success. In the tropics, rats have a less constrained timing of breeding, and a breeding cycle as short as four months, so a shorter time may be required to reliably detect a recovering rat population, particularly in wetter conditions (Keitt, et al., 2015).

For many widespread invasive rodents, however, there is a lack of basic ecological knowledge about densities, and the factors affecting the large variation in abundance that is evident for highly versatile invasive rodents (Harper & Bunbury, 2015). Henderson Island (24°20'S, 128°19'W), in the subtropical Pitcairn Islands of the South Pacific, was subject to an aerial poison bait-based eradication attempt of the introduced Pacific rat (*Rattus exulans*) in 2011 (Torr & Brown, 2012). The eradication was unsuccessful, but the cause of the failure was neither operational shortcoming nor due to resistance of rats to brodifacoum pellets (Torr & Brown, 2012; Amos, et al., 2016, Brooke, 2019).

Here, we report on the population recovery of *R. exulans* on Henderson Island up to four years following a failed eradication attempt, and provide information on short-term seasonality in density of Pacific rats using live trapping and a spatially explicit capture-recapture (SECR) framework. We use the obtained estimates in a rapid eradication assessment (Russell, et al., 2017) to provide guidance on the length of a post-operation monitoring period after which an eradication could be considered successful with 95% certainty. These data provide a robust overview of the short- and long-term population variability on an aseasonal sub-tropical island that will inform future conservation management.

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MATERIALS AND METHODS

Study area

Henderson Island, a UNESCO World Heritage Site, is a 43 km^2 raised coral atoll in the Pitcairn Islands, South Pacific Ocean, with a tropical climate (Spencer, 1995; Weigelt, et al., 2013). The island was subjected to an unsuccessful aerial eradication attempt of the introduced Pacific rat (*Rattus exulans*) in 2011 (Torr & Brown, 2012). While the ultimate cause of the eradication failure remains unknown, resistance of rats to brodifacoum pellets or operational errors are not considered factors (Torr & Brown, 2012; Amos, et al., 2016).

We conducted our study at the northern end of the island's plateau and along the two accessible beaches, North Beach and East Beach. The plateau substrate is fossilised coral with a uniform, dense native vegetation consisting of mostly of *Pandanus tectorius*, *Xylosma suaveolens* and *Psydrax odorata* (Waldren, et al., 1995). The beach and embayment forest ("beach back") areas have a sandy substrate with a mixed low vegetation and small stands of introduced coconut (*Cocos nucifera*) (Waldren, et al., 1995).

Rat snap-trapping and long-term abundance indices

We estimated rat abundance indices in 2009 (September), 2012 (May and November), 2013 (August), and 2015 (October and November) on the plateau and embayment forest areas of North and East beaches of Henderson using snap-traps, though the precise methods differed because of logistical and time constraints. In all years, however, we set traps between 16:00–18:00 h (all times UTC-8), and checked the following morning between 08:00–10:00 h (Table 1).

We recorded the traps' contents (rat, crab, or snapped and empty) to calculate an index of abundance as the number of rats caught per 100 corrected trap-nights (100 CTN; Nelson & Clark, 1973), where

CTN = Total trap-nights – Trap-nights lost (equation 1);

Trap-nights lost = ½ × (crab captures + snapped traps) (equation 2).

Estimating long-term rat population change

Based on the rat abundance indices derived from snap-trapping, we estimated the annual per capita growth rate, *r*, using the formula:

$$r = \frac{\log\left(\frac{N_t}{N_{t-1}}\right)}{t}$$

(equation 3), where *N* is the population estimate at time *t* and *t*-1, and *t* is the elapsed time, in years, between the two estimates.

To estimate a population growth rate, which requires non-zero values in each time interval if no immigration is assumed, we scaled abundance indices derived from snap-trapping to an island population size. This approach allowed us to have all the population estimates on the same scale, and to include the very small population size in 2011. We extrapolated population size based on live- and snap-trapping data from 2009: based on live-trapping, there were approximately 28 rats/ha (95% confidence interval: 23–40 rats/ha) in the embayment forest of North Beach on Henderson (Cuthbert, et al., 2012), which corresponded to 31.7 rats 100 CTN^{-1} in the same habitat. We extrapolated the density estimate to an approximate population size of 120,000 (range: 104,000–172,000) rats on the island, assuming equal density across all habitat types, and used the relationship between this extrapolated population size and the snap-trapping rate to extrapolate population sizes in other years. For 2011, when no snap-trapping data were available, we used the population estimate of 60–80 individuals that was estimated to have survived the eradication attempt in 2011 based on genetic markers (Amos, et al., 2016). In each year, we proportioned the total population to the three different habitats in which we measured rat abundance based on their relative area: North Beach embayment forest (7 ha), East Beach embayment forest (7 ha), and the island plateau (4,290 ha), and the initial population based on the abundance indices in these three habitats in 2009. We assumed that trapability was constant among years.

Rat live trapping and density estimation

To obtain a robust estimate of rat density and to document short-term fluctuation in rat density over six months, we implemented a spatial capture-mark-recapture programme in 2015 (Oppel, et al., 2019). Rats were live-trapped on the plateau from 28 May to 16 October 2015 during seven primary sessions of 10 consecutive trap-nights each, followed by a window of 10–15 days with no trapping between primary sessions. We established a trap network placed along 3.5 km of cleared paths (Fig. 1), and traps were arranged at distances from 3–20 m along 343 locations, with a different subset of 250 trap locations used during each primary session. Because our original traps (Sherman and Elliott aluminum boxes; model LFA, 23 × 9 × 8 cm, H.B. Sherman Traps Inc., Tallahassee, Florida, USA) were easily damaged by crabs, they were replaced by larger and more robust Tomahawk cage traps in September (27 × 16 × 13 cm, Key Industries, Auckland, New Zealand).

In the embayment forest at North Beach, rats were live-trapped during three primary occasions of 6–10 trap-nights each between 1 August and 19 September 2015 using 38

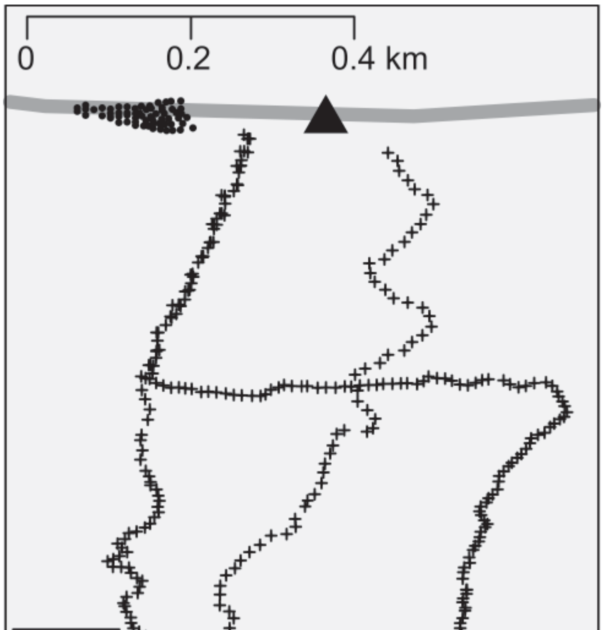

Fig. 1 Map of the path network on the north end of Henderson Island showing the locations of all live traps used in 2015 (+), snap-trapping grid on North Beach (●) and the location of the research camp (▲).

Table 1 Snap trap arrangements and bait used on Henderson Island from 2009–2015.

Habitat	Start date	End date	Bait	Trap spacing
Embayment forest – East Beach	12 Sep 09	21 Sep 09	Coconut, peanut butter, rolled oats	10–15 m
	14 Aug 11	23 Aug 11		
	23 Nov 12	30 Nov 12	Peanut butter, coconut	10–15 m
	21 Aug 13	28 Aug 13	Peanut butter, coconut	10–15 m
Embayment forest – North Beach	12 Sep 09	21 Sep 09	Coconut, peanut butter, rolled oats	10–15 m
	14 Aug 11	23 Aug 11		
	03 May 12	06 May 12	Peanut butter, rolled oats, chocolate	8–10 m
	23 Nov 12	30 Nov 12	Peanut butter, coconut	10–15 m
	21 Aug 13	28 Aug 13	Peanut butter, coconut	10–15 m
	21 Oct 15	31 Oct 15	Coconut, *Pandanus*	20 m
	01 Nov 15	14 Nov 15	Coconut, *Pandanus*	20 m
Plateau	12 Sep 09	21 Sep 09	Coconut, peanut butter, rolled oats	10–15 m
	14 Aug 11	23 Aug 11		
	03 May 12	06 May 12	Peanut butter, rolled oats, chocolate	8–10 m
	23 Nov 12	30 Nov 12	Peanut butter, coconut	10–15 m
	21 Aug 13	28 Aug 13	Peanut butter, coconut	10–15 m
	21 Oct 15	31 Oct 15	Coconut, *Pandanus*	20 m
	01 Nov 15	14 Nov 15	Coconut, *Pandanus*	20 m

traps arranged in a 6 × 6 configuration with traps spaced 10 m apart, and we expanded the trapping grid to 63 traps in a 7 × 9 configuration for the last primary session.

Before the first primary session in each habitat, traps were deployed, but not opened, for approximately five days to allow rats to overcome neophobia. For each ten-day trapping period, traps were baited with a small (approximately 1 × 1 cm) cube of fresh coconut between 16:00–18:00 h, and checked the following morning between 08:00–10:00 h.

During trap checks each captured rat was fitted with a uniquely numbered ear tag, or the number of an existing ear tag was recorded, and the rat was released next to the trap. We recorded the trap location for each capture and recorded whether traps were available to capture rats or had been rendered ineffective (e.g. by crabs). We estimated a capture index (rats/100 corrected trap nights) for the plateau and the embayment forest for each trap-night using the same equation as above to correct for inactivated traps.

To estimate rat densities, we used spatially-explicit capture-recapture models, which have been used successfully for other rat density estimations on islands (Russell, et al., 2011; Ringler, et al., 2014; Harper, et al., 2015). We assumed that rat home ranges were randomly located with respect to trap locations and stationary within a given primary session, and that the central location of the home range was adequately described by a homogenous Poisson distribution (Efford, 2004; Borchers, 2012). Capture probability of rats at a given trap was based on the distance of the rat's home range centre from the trap and was modelled with a half-normal function in the embayment forest (Borchers & Efford, 2008; Harper, et al., 2015) and a negative exponential function on the plateau where the distribution of rat movements included a long tail of some very large movements >500 m. We estimated density using the function 'secr.fit' in the package 'secr' (Efford, 2016) using R 3.2.5 (R Core Team, 2017) for each habitat and primary session separately, thus allowing for density, capture probability, and the movement parameter, σ, to vary over time and habitat. We did not consider trap dependence. We report estimates of density, capture probability and σ with 95% confidence intervals.

Rapid eradication assessment

During the eradication operation in 2011, a team remained on the island for three months after the bait drop (from August to November), and any future eradication operation will require a similar post-operational period to monitor non-target species (Oppel, et al., 2016). We therefore estimated whether rat monitoring at all 406 trap locations of our two networks could conclude that an eradication had been successful with 95% certainty if no rat was detected during three 10-day trapping sessions up to three months after the bait drop. We also explored whether certainty could be increased if a larger area was covered with traps, and simulated a 30 × 30 m trapping grid over 10%, 30%, and 50% of Henderson Island. We used our empirical estimates of population growth rate and rat roaming behaviour in a rapid assessment tool (REA Shiny; Russell, et al., 2017) assuming a prior probability of success of 83.9% (Russell & Holmes, 2015), no reinvasion (Amos, et al., 2016), and rat dispersal distances of up to 500 m (Oppel, et al., 2019). We present the probability of successful eradication that could be inferred given that no rat was detected during the specified survey effort.

RESULTS

Rat abundance estimates and long-term population recovery

We trapped rats from 11 August to 21 September 2009, catching 233 rats in 734.5 corrected trap-nights overall, or 31.7 rats/100 CTN, with little difference among habitats

(29.0–33.4 rats/100 CTN; Table 2). The eradication attempt in August 2011 reduced the Henderson rat population to 60–80 individuals (Amos, et al., 2016), and eight months after the eradication one rat was caught on the plateau in 96.5 corrected trap nights. From 23–30 November 2012, we caught 9.2–14.8 rats/100 CTN across all three habitats (Table 2).

In 2013, we caught 20.0–73.2 rats/100 CTN, and the abundance index exceeded the pre-eradication estimate in the embayment forest (by more than 100% on North Beach), while the population on the plateau was 62% of pre-eradication levels.

In October 2015, we caught again more rats/100 CTN in embayment forest habitat on North Beach (42.9 rats/100 CTN) than on the plateau (13.2 rats/100 CTN) corresponding to an abundance index similar to pre-eradication conditions in the embayment forest, but only 41% of pre-eradication levels on the plateau (Table 2).

Based on these rat abundance indices, the rat population appears to have recovered rapidly, with annual per capita growth rates ranging from 0.48 to 5.95 (Table 2) during the recovery phase. The estimated number of rats reached peaks of 113%, 219%, and 62% of pre-eradication levels on East Beach, North Beach, and the plateau, respectively, by 2013 (Table 2), two years after the eradication attempt. The annual population growth rate has decreased since 2013 and was slightly negative between 2013 and 2015 (Table 2).

Short-term fluctuation in rat density

Overall in 2015, we recorded a total of 2,826 rat captures in 7,552 corrected trap-nights in our live-trapping network on the plateau and 319 captures in 684 corrected trap-nights in the embayment forest. Trapping rates in the embayment forest were much higher than on the plateau and less variable over time (Fig. 2). On the plateau, the trapping index declined from 36.6 rats/100 CTN in early July to 12.6 rats/100 CTN in late August (Fig. 2). The subsequent increase to 75.8 rats/100 CTN occurred after switching Sherman traps with Tomahawk traps, and any population increase is therefore confounded by a potentially more effective trap type. To account for habitat- and time-specific variation in capture probability, we estimated rat density using spatially explicit capture-recapture models for each primary session.

Rat density in the embayment forest was about 10× higher than that on the plateau (Fig. 2), and there were significant temporal fluctuations in both habitat types: apparent rat densities declined by 50% within one month in the embayment forest, and by 85% within two months on the plateau before recovering to 80% of the original density another three months later (Fig. 2). Lower rat densities on the plateau coincided with increased rat roaming distances (σ), which were generally larger on the plateau than in the embayment forest (Fig. 2). Despite some very long rat movements on the plateau, only three individuals were recorded in both the embayment forest and on the plateau (Fig. 1).

Table 2 Abundance indices of *Rattus exulans* on Henderson Island increased markedly following an eradication attempt in August 2011. August 2011 population estimate from Amos, et al. (2016); 2009 rat population from Brooke, et al., (2010a), and resulting population estimates are calculated from the relationship between rats/100 corrected trap nights (CTN, see text for details) and population size. Annual per capita growth rate, r, is based on exponential population growth.

Habitat	Start – end date	Corrected trap nights	Rats caught	Rats 100 CTN^{-1}	No of rats[a]	% of original popn	Time between surveys (months)	Annual growth rate *r*
Embayment forest – East Beach	12 – 21 Sep 2009	252	73	29.0	210		-	
	14 – 23 Aug 2011	-	-	-	5	2%	22.8	
	23 – 30 Nov 2012	83	11	13.3	96	46%	15.0	2.36
	21 – 28 Aug 2013	92	30	32.6	236	113%	8.7	1.24
Embayment forest – North Beach	12 – 21 Sep 2009	374	125	33.4	210		-	
	14 – 23 Aug 2011	-	-	-	5	2%	22.8	
	03 – 6 May 2012	88.5	0	0.0	7	3%	8.4	0.48
	23 – 30 Nov 2012	67.5	10	14.8	93	44%	6.6	4.70
	21 – 28 Aug 2013	82	60	73.2	460	219%	8.7	2.21
	21 – 31 Oct 2015	49	21	42.9	269	128%	25.8	-0.25
	01 – 14 Nov 2015	149.5	36	24.1	151	72%	1.0	NA[b]
Plateau	12 – 21 Sep 2009	108.5	35	32.3	120,120		-	
	14 – 23 Aug 2011	-	-	-	60	0%	22.8	
	03 – 6 May 2012	96.5	1	1.0	3,859	3%	8.4	5.95
	23 – 30 Nov 2012	272	25	9.2	34,225	28%	6.6	3.97
	21 – 28 Aug 2013	703.5	141	20.0	74,633	62%	8.7	1.08
	21 – 31 Oct 2015	836.5	110	13.2	48,967	41%	25.8	-0.20
	01 – 14 Nov 2015	511.5	50	9.8	36,400	30%	1.0	NA[b]

[a]Number of rats extrapolated from the relationship between the abundance index and original population estimate and the mean (95% credible interval) from the state-space model used to calculate r. See text for details.
[b]Populations and population changes for October–November 2015 were not calculated because snap trapping removes individuals from the population and biases abundance indices for short time periods.

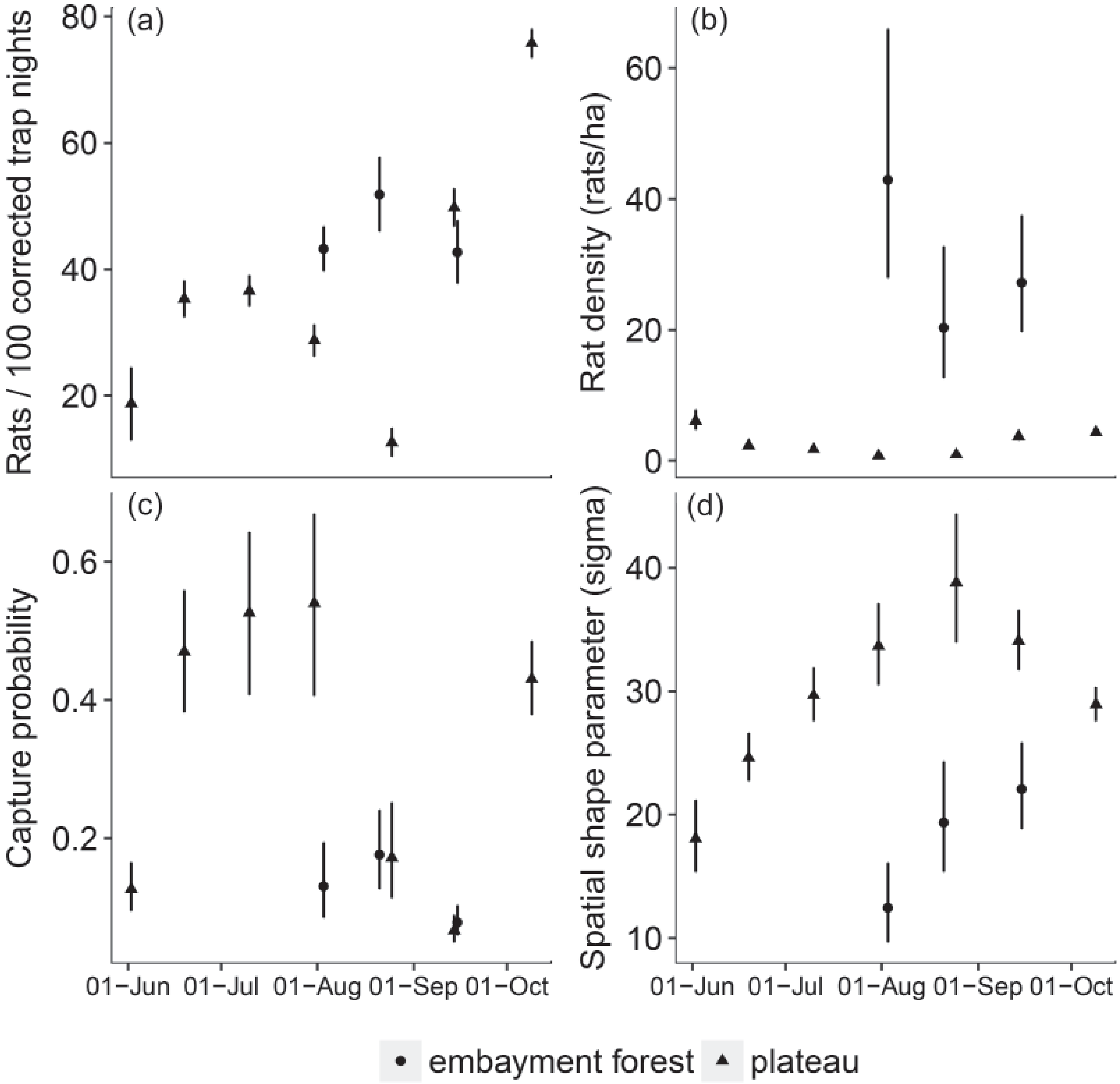

Fig. 2 Rat abundance (a), density (b), capture probability (c), and spatial shape parameters (c) from spatially-explicit capture-recapture analysis of Pacific rats in two habitats on Henderson Island in 2015 (black circles: embayment forest, open circles: plateau).

Rapid eradication assessment

Using the same trapping array as in 2015 for three 10-day trapping sessions at monthly intervals following a future hypothetical eradication attempt on Henderson Island would be insufficient to declare an eradication successful with 95% certainty. When we simulated a larger trapping array, we found that we could only conclude with 95% confidence that the eradication had been successful if no rat was detected within three months on a trapping grid covering at least 30% of Henderson Island. The model indicated that in order to be confident that the eradication had been successful within three months, we would require a 30 × 30 m trapping grid covering one third of the island.

DISCUSSION

Population recovery after a failed eradication

The rat population on Henderson increased rapidly for at least the first 15 months following the eradication operation, with high annual per capita growth rates up to August 2013 (Table 2). As the population approached or exceeded pre-eradication abundances, the growth rate decreased between August 2013 and October 2015, possibly as a result of the population fluctuating around a carrying capacity. These growth rates are broadly similar to the maximum annual growth rates of other rat species (Hone, et al., 2010), and are useful to estimate the probability of success of an eradication during follow-up monitoring (Russell, et al., 2017). Owing to variability in trapping methods and locations, there is considerable uncertainty about the exact size of the rat population; however, our density estimates in the embayment forest in 2015 indicate that rat density in this habitat was very similar to the pre-eradication density estimated in the same habitat at 24–40 rats/ha in 2009 (Cuthbert, et al., 2012). Based on the updated density estimates from the plateau in 2015, we estimate the rat population on North Beach in 2015 to be ~150–300 rats, which is similar to the estimates from the extrapolated relationship between density and abundance indices (Table 2). On the plateau, however, density fluctuated considerably throughout the year (Table 3), and extrapolating to the 4,290 ha of plateau habitat resulted in an estimate of ~3,300–26,000 rats, which is lower than the 48,000 estimate from the relationship with the abundance index. We assumed the relationship between density (rats ha^{-1}) and the abundance index (rats 100 CTN^{-1}) was linear but, on the plateau, this is clearly not the case. Estimating density, however, entails significant work over several months, whereas an abundance index can be determined fairly quickly, in a matter of days. Further work should investigate factors that influence the relationship between these two metrics.

There have been few studies on the recovery of *Rattus* spp. following eradication attempts. In urban Baltimore, Maryland, USA, *R. norvegicus* recovered to pre-control numbers within about 12–18 months (Emlen Jr., et al., 1948), and *R. fuscipes* in Australian eucalypt patches returned to pre-removal densities within two years (Lindenmayer, et al., 2005). In both cases, immigration was the likely cause of the rapid increase (though see Banks, et al., 2011). Genetic analysis from Henderson shows that there was no reinvasion, and that all rats present are descended from 60–80 survivors of the failed 2011 eradication operation (Amos, et al., 2016). Our results demonstrate the rapid recovery of an island population of introduced rodents in the absence of immigration. The time for rodent populations to either recover or reach pre-eradication levels (15–24 months), was similar to the experimental invasion of Saddle Island, New Zealand by mice (*Mus musculus*), where immigration may have supplemented mouse populations (Nathan, et al., 2015), and the time from arrival to near-saturation of black rats (*Rattus rattus*) on Taukihepa, New Zealand (24–36 months; Bell, et al., 2016).

Temporal and spatial variation in rat population density

The shape of the recovery curve of the rat population on Henderson is difficult to determine from the intermittent trapping efforts and due to the high short-term variability. In 2015 we documented three-fold fluctuations in live trapping indices and even larger differences in rat density within just two months (Fig. 2), indicating that there may be pronounced seasonal changes among the rat population that could potentially mask or confound any long-term trajectories. There may also be considerable spatial heterogeneity in rat densities, and rats on Henderson do travel large distances (Oppel, et al., 2019), which further complicate interpretations from sampling a relatively small area of the available habitat. Some rats are relatively territorial, moving <200 m, and others roaming >1000 m (Oppel, et al., 2019).

Our finding that a decrease in rat density coincided with increasing movement rates of rats (Fig. 2) adds a further complication to the long-term comparison of simple trap indices that do not account for capture probability and rat movements. Tropical rodent populations are known to undergo large population fluctuations, which can be driven by short-term changes in resource availability (Adler, 1998; Madsen & Shine, 1999) or extreme climatic events (Ujvari, et al., 2016). We did not observe pronounced plant resource fluctuations in 2015, and most tree species had individuals at various stages of flowering and fruiting between June and October 2015, though invertebrate abundance likely varied through the season (Lavers, et al., 2016). There was also neither a noticeable drought, nor an unusually heavy rainfall event during that period that could have explained the apparent intermittent reduction in the rat population. In the beach embayment forest, the major reduction of rat density between early and late August coincided with the temporal availability of Murphy's petrel (*Pterodroma ultima*) chicks, which may have led to temporary immigration of rats, but would have been unable to sustain a rat population for more than a few days (Brooke, et al., 2010b). Although we do not know whether higher mortality, lower fecundity, or both contributed to the apparent temporal fluctuation that we observed, or whether rats' probability of capture changed significantly over time, the timing of any future eradication operation should coincide with a naturally occurring nadir in the population trajectory to improve the probability of success. The eradication operation in 2011 therefore appears to have been optimally timed if rat population fluctuations are similar every year, but more research is required to examine whether rat populations exhibit predictable seasonality on sub-tropical islands such as Henderson.

The population abundance indices of *R. exulans* on Henderson (14–32 rats/100 CTN with snap-trapping, 12–75.8 rats/100 CTN with live-trapping) appear to be higher than abundance indices of other island rat population. For example, the *R. exulans* abundance index on Hawaii was only 5.65 rats /100 CTN (Sugihara, 1997), presumably because the species is subject to competition and predation (Moller & Craig, 1987); on the Marianas, the trapping rate was also much lower than on Henderson with 3.7 rats/100 CTN (Yackel Adams, et al., 2011). On Honuea, French Polynesia, indices ranged from 5–20 rats/100 CTN, often lower than conspecific *R. rattus* (up to 35 rats/100 CTN; Russell, et al., 2015). Abundance indices of the much larger *R. norvegicus* ranged from 3–9 rats /100 CTN (Drever, 2004; Harper, et al., 2005; Bond & Eggleston, 2015), and those of *R. rattus* from 1.6–35 rats/100 CTN across their range (Blackwell, et al., 2002; Shiels, 2010; Russell, et al., 2015), but reached up to 94.1/100 CTN on some nearshore islands in New Zealand (Russell & MacKay, 2005), and ranged from 60–80 rats/100 CTN on Surprise Island, New Caledonia (Caut, et al., 2009). The

Table 3 Mean (± 95% confidence interval) rat density, capture probability, and movement parameter, σ, in two habitats of Henderson Island in June–October 2015 estimated with spatially-explicit capture-recapture models. Note that different detection functions were used in the embayment forest (half-normal) and on the plateau (negative exponential), and that σ values are not directly comparable.

Habitat	Time period	Density (rats ha^{-1})	Capture probability	σ
Beach embayment – North Beach	Early August	42.92 (27.92–65.98)	0.13 (0.09–0.19)	12.48 (9.67–16.11)
	Late August	20.37 (12.67–32.73)	0.18 (0.13–0.24)	19.35 (15.41–24.29)
	September	27.2 (19.73–37.48)	0.08 (0.06–0.1)	22.08 (18.85–25.86)
Plateau	Early June	6.08 (4.76–7.76)	0.13 (0.1–0.17)	18.06 (15.39–21.19)
	Late June	2.33 (1.95–2.78)	0.47 (0.38–0.56)	24.6 (22.74–26.61)
	July	1.77 (1.49–2.11)	0.53 (0.41–0.64)	29.67 (27.57–31.92)
	Early August	0.76 (0.6–0.95)	0.54 (0.41–0.67)	33.64 (30.52–37.09)
	Late August	0.94 (0.75–1.17)	0.17 (0.11–0.25)	38.78 (33.92–44.33)
	September	3.73 (3.15–4.4)	0.07 (0.05–0.1)	34.05 (31.75–36.51)
	October	4.36 (3.92–4.85)	0.43 (0.38–0.49)	28.92 (27.56–30.34)

much higher trapping rate of *R. exulans* on Henderson is possibly because the species is smaller than congeners and is not subject to either predation or competition because no other mammals or avian predators exist on Henderson, and there is minimal dietary overlap with Henderson's birds (Brooke & Jones, 1995; Jones, et al., 1995; Trevelyan, 1995; Lavers, et al., 2016). In addition, the relatively high temperature and greater resource availability on tropical islands is generally well known to increase rat population size compared to temperate islands (Harper & Bunbury, 2015; Russell & Holmes, 2015). It is important to note, however, that abundance indices of rats exhibit a wide range depending on the rat species, environment, and the presence of competitor or predator species, seasonality, trap type, and the layout and spacing of traps.

Different approaches used to estimate densities also complicate the comparison across different islands (Harper & Bunbury, 2015). Despite the relatively high snap- and live-trapping rates on Henderson, our estimate of rat density is surprisingly low, especially on the coral plateau, where large rat movements were observed (Oppel, et al., 2019) that may have led to high trapping indices despite low density. But even the 10-times higher density in the beach embayment forest appears to be at the lower end of the range found for *R. exulans* on tropical islands (1.2–288 rats/ha; Harper & Bunbury, 2015). A potential explanation for this apparent discrepancy might be that Henderson Island is a relatively nutrient-poor coral atoll, where the maximum population size could be lower compared to more fertile tropical islands. Due to the potential nutrient limitation, the use of a highly attractive bait (coconut) may result in relatively high trapping rates, especially on the plateau where coconut is generally unavailable. Coconut has been implicated as an important factor affecting the eradication success on tropical islands (Holmes, et al., 2015). Our data also suggest that coconut may have facilitated a rapid recovery of the surviving rat population: In 2009, trapping rates were similar in the beach embayment forest and on the plateau, but on all occasions after the eradication attempt the snap-trapping rates on the plateau were considerably lower than in the embayment forest (Table 2). Rats on Henderson would often gnaw into de-husked coconuts, or those opened by land crabs on the beaches. We speculate that the lush embayment forest with abundant coconut may have facilitated a faster return to pre-eradication rat population densities than the scrubby plateau forest where coconut, though present, is scarce compared to the embayment forest.

Previous investigations using live- and snap-trapping indicated that live traps do not have a higher capture probability than snap traps for *R. exulans* and *R. rattus* (Russell, et al., 2015). We observed a much higher trapping rate with live traps than with snap traps in 2015, and our spatially explicit capture-recapture model indicated that rats living along our trail network had an almost 100% probability of being captured at least once in a trap. The variation in capture probability between different trap types highlights the need for consistent monitoring using identical approaches to facilitate valid comparisons over time.

Assessing the probability of eradication success

The failure of the 2011 eradication attempt was opportunistically recorded eight months after the bait drop. Our assessment indicated that an intensive monitoring programme for three months following a bait drop would not have allowed the reliable conclusion that the eradication had been successful, unless at least 30% of the island were covered with a 30 × 30 m trapping grid. Such a trapping effort is unrealistic on Henderson Island. Rapid eradication assessments have so far only been conducted for small islands where the survey effort covered the entire area habitable by rats (Samaniego-Herrera, et al., 2013; Russell, et al., 2017) but because Henderson Island is a fairly large island with impenetrable vegetation it is logistically unrealistic to install a monitoring network across an area sufficiently large to enable an early declaration of success. Depending on where rats that survive an eradication attempt occur in relation to the trap array at the northern end of the island, the potential benefit of post-operational monitoring to facilitate a rapid assessment of success is questionable. By the time surviving rats may be discovered with the limited trap array, the population would have likely grown to a size that would require a new eradication rather than allow a rapid follow-up to kill any remaining survivors. Although post-operational rat trapping on Henderson Island may be useful to rapidly discover eradication failure, it is unlikely that it would allow the confident conclusion of eradication success.

In summary, we found that rat abundance increased rapidly between the failed eradication operation in August 2011 and August 2013 before decreasing from August 2013 to October 2015. Rats on Henderson Island reached two-thirds of their pre-eradication abundance 24 months following their failed eradication, but our estimates of rat density on the plateau of the island suggest that rat density may have been substantially lower than previously assumed (Brooke, et al., 2010a). Studies of failed eradication operations, and the recovery of introduced rodent populations are rare, but of great conservation and operational importance. Our study highlights rodent population fluctuations on a relatively short time-scale, and a better understanding of the regularity and underlying drivers of these fluctuations would be useful to schedule an eradication operation so as to maximise eradication success. As eradication operations on tropical and subtropical islands become more frequent, it will be increasingly important to understand the behaviour and ecology of the invasive species targeted, and more work in this area is required if we are to replicate success on temperate islands.

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REFERENCES

Adler, G.H. (1998). 'Impacts of resource abundance on populations of a tropical forest rodent'. *Ecology* 79(1): 242–254.

Amos, W., Nichols, H.J., Churchyard, T. and Brooke, M.de L. (2016). 'Rat eradication comes within a whisker! A case study from the South Pacific'. *Royal Society Open Science* 3: 160110.

Banks, S.C., Dujardin, M., McBurney, L., Blair, D., Barker, M. and Lindenmayer, D.B. (2011). 'Starting points for small mammal population recovery after wildfire: Recolonisation or residual populations?' *Oikos* 120: 26–37.

Bell, E.A., Bell, B.D. and Merton, D.V. (2016). 'The legacy of Big South Cape: Rat irruption to rat eradication'. *New Zealand Journal of Ecology* 40(2): 212–218.

Bellingham, P.J., Towns, D.R., Cameron, E.K., Davis, J.J., Wardle, D.A., Wilmshurst, J.M. and Mulder, C.P.H. (2010). 'New Zealand island restoration: Seabirds, predators, and the importance of history'. *New Zealand Journal of Ecology* 34(1): 115–136.

Blackwell, G.L., Potter, M.A. and McLennan, J.A. (2002). 'Rodent density indices from tracking tunnels, snap-traps and Fenn traps: Do they tell the same story?' *New Zealand Journal of Ecology* 26(1): 43–51.

Bond, A.L. and Eggleston, C.J. (2015). 'Application of a non-invasive indexing method for introduced Norway rats (*Rattus norvegicus*) in the Aleutian Islands, Alaska'. *Biodiversity and Conservation* 24: 2551–2563.

Borchers, D.L. and Efford, M.G. (2008). 'Spatially explicity maximum likelihood methods for capture-recapture studies'. *Biometrics* 64: 377–385.

Borchers, D. (2012). 'A non-technical overview of spatially explicit capture–recapture models'. *Journal of Ornithology* 152(2): 435–444.

Brooke, M. de L. and Jones, P.J. (1995) 'The diet of the Henderson fruit-dove *Ptilinopus insularis*. Island Field observations of fruit choice'. *Biological Journal of the Linnean Society* 56: 149–165.

Brooke, M.de L., Cuthbert, R.J., Henricson, A., Torr, N., Warren, P. and O'Keefe, S. (2010a). *Towards Rat Eradication on Henderson Island: Fieldwork Report, August – September 2009*. Unpublished report. Sandy, UK: Royal Society for the Protection of Birds.

Brooke, M.de L., O'Connell, T.C., Wingate, D., Madeiros, J., Hilton, G.M. and Ratcliffe, N. (2010b). 'Potential for rat predation to cause decline of the globally threatened Henderson petrel *Pterodroma atrata*: Evidence from the field, stable isotopes and population modelling'. *Endangered Species Research* 11: 47–59.

Brooke, M. de L. (2019). 'Rat Eradication in the Pitcairn Islands, South Pacific: a 25-year Perspective'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 95–99. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Cuthbert, R.J., Brooke, M.de L. and Torr, N. (2012). 'Overcoming hermit-crab interference during rodent-baiting operations: A case study from Henderson Island, South Pacific'. *Wildlife Research* 39(1): 70–77.

DIISE (2016). 'The Database of Island Invasive Species Eradications, developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group. University of Auckland and Landcare Research New Zealand'. <http://diise.islandconservation.org>. Accessed 1 July 2016.

Drever, M.C. (2004). 'Capture rates of Norway rats (*Rattus norvegicus*) in a seabird colony: A caveat for investigators'. *Northwestern Naturalist* 85: 111–117.

Efford, M. (2004). 'Density estimation in live-trapping studies'. *Oikos* 106(3): 598–610.

Efford, M.G. (2016). 'secr: Spatially Explicit Capture-recapture in R. R Package Version 2.10-3'. <https://CRAN.R-project.org/package=secr>.

Emlen Jr., J.T., Stokes, A.W. and Winsor, C.P. (1948). 'The rate of recovery of decimated populations of brown rats in nature'. *Ecology* 29(2): 133–145.

Harper, G.A., Dickinson, K.J.M. and Seddon, P.J. (2005). 'Habitat use by three rat species (*Rattus* spp.) on Stewart Island/Rakiura, New Zealand'. *New Zealand Journal of Ecology* 29(2): 251–260.

Harper, G.A. and Bunbury, N. (2015). 'Invasive rats on tropical islands: Their population biology and impacts on native species'. *Global Ecology and Conservation* 3: 607–627.

Harper, G.A., van Dinther, M., Russell, J.C. and Bunbury, N. (2015). 'The response of black rats (*Rattus rattus*) to evergreen and seasonally arid habitats: Informing eradication planning on a tropical island'. *Biological Conservation* 185: 66–74.

Hein, S. and Jacob, J. (2015). 'Recovery of small rodent populations after population collapse'. *Wildlife Research* 42(2): 108–118.

Holmes, N.D., Griffiths, R., Pott, M., Alifano, A., Will, D., Wegmann, A.S. and Russell, J.C. (2015). 'Factors associated with rodent eradication failure'. *Biological Conservation* 185: 8-–16.

Hone, J., Duncan, R.P. and Forsyth, D.M. (2010). 'Estimates of maximum annual population growth rates (r_m) of mammals and their application in wildlife management'. *Journal of Applied Ecology* 47(3): 507–514.

Howald, G.R., Donlan, C.J., Galván, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21(5): 1258–1268.

Jones, H.P., Holmes, N.D., Butchart, S.H.M., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P.E., Cuthbert, R.J., Ebbert, S.E., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R. & Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences of the United States of America* 113: 4033–4038.

Jones, P.J., Schubel, S., Jolly, J.N., Brooke, M. de L., and Vickery, J.A. (1995) 'Behaviour, natural history, and annual cycle of the Henderson Island rail *Porzana atra* (Aves: Rallidae)'. *Biological Journal of the Linnean Society* 56: 167–183.

Keitt, B.S., Griffiths, R., Boudjelas, S., Broome, K.G., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 182: 17–26.

Lavers, J.L., McClelland, G.T.W., MacKinnon, L., Bond, A.L., Oppel, S., Donaldson, A.H., Duffield, N.D., Forrest, A.K., Havery, S.J., O'Keefe, S., Skinner, A., Torr, N. and Warren, P. (2016). *Henderson Island Expedition Report: May–November 2015*. RSPB Research Report 57. Sandy, UK: RSPB Centre for Conservation Science.

Lindenmayer, D.B., Cunningham, R.B. and Peakall, R. (2005). 'The recovery of populations of bush rat *Rattus fuscipes* in forest fragments following major population reduction'. *Journal of Applied Ecology* 42: 649–658.

Lorvelec, O. and Pascal, M. (2005). 'French attempts to eradicate non-indigenous mammals and their consequences for native biota'. *Biological Invasions* 7: 135–140.

Madsen, T. and Shine, R. (1999). 'Rainfall and rats: Climatically-driven dynamics of a tropical rodent population'. *Australian Journal of Ecology* 24(1): 80–89.

Moller, H. and Craig, J.L. (1987). 'The population ecology of *Rattus exulans* on Tiritiri Matangi Island, and a model of comparative population dynamics in New Zealand'. *New Zealand Journal of Zoology* 14(3): 305–328.

Nathan, H.W., Clout, M.N., MacKay, J.W.B., Murphy, E.C. and Russell, J.C. (2015). 'Experimental island invasion of house mice'. *Population Ecology* 57: 363–371.

Nelson, L. and Clark, F.W. (1973). 'Correction for sprung traps in catch/effort calculations of trapping results'. *Journal of Mammalogy* 54(1): 295–298.

Oppel, S., Bond, A.L., Brooke, M.de L., Harrison, G., Vickery, J.A. and Cuthbert, R.J. (2016). 'Temporary captive population and rapid population recovery of an endemic flightless rail after a rodent eradication operation using aerially distributed poison bait'. *Biological Conservation* 204B: 442–448.

Oppel, S., McClelland, G.T.W., Lavers, J.L., Churchyard, T., Donaldson, A.H., Duffield, N.D., Havery, S., Kelly, J., Proud, T., Russell, J., Vickery, J.A. and Bond, A.L. (2019). 'Seasonal variation in movements and survival of invasive Pacific rats on a sub-tropical island: implications for eradication'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 200–208. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

R Core Team (2017). *R: A Language and Environment for Statistical Computing. Version 3.4.0.*Vienna, Austria: R Foundation for Statistical Computing.

Ringler, D., Russell, J., Jaeger, A., Pinet, P., Bastien, M. and Le Corre, M. (2014). 'Invasive rat space use on tropical islands: Implications for bait broadcast'. *Basic and Applied Ecology* 15(2): 179–186.

Russell, J. and MacKay, J. (2005). *Ship Rat Reinvasion of Tawhitinui Island*. Unpublished report. Auckland, NZ: University of Auckland.

Russell, J.C., Ringler, D., Trombini, A. and Le Corre, M. (2011). 'The island syndrome and population dynamics of introduced rats'. *Oecologia* 167: 667–676.

Russell, J.C., Caut, S., Anderson, S.H. and Lee, M. (2015). 'Invasive rat interactions and over-invasion on a coral atoll'. *Biological Conservation* 185: 59–65.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Russell, J.C., Binnie, H.R., Oh, J., Anderson, D.P. and Samaniego-Herrera, A. (2017). 'Optimizing confirmation of invasive species eradication with rapid eradication assessment'. *Journal of Applied Ecology* 54: 160–169.

Samaniego-Herrera, A., Anderson, D.P., Parkes, J.P. and Aguirre-Muñoz, A. (2013). 'Rapid assessment of rat eradication after aerial baiting'. *Journal of Applied Ecology* 50: 1415–1421.

Shiels, A.B. (2010). 'Ecology and impacts of introduced rodents (*Rattus* spp. and *Mus musculus*) in the Hawaiian Islands'. Ph.D. thesis. Manoa: University of Hawai'i.

Spencer, T. (1995). 'The Pitcairn Islands, South Pacific Ocean: Plate tectonic and climatic contexts'. *Biological Journal of the Linnean Society* 56: 13–42.

Sugihara, R.T. (1997). 'Abundance and diets of rats in two native Hawaiian forests'. *Pacific Science* 51(2): 189–198.

Torr, N. and Brown, D. (2012). 'Henderson Island Restoration Project Post-operational Report. Unpublished report. Sandy, UK: Royal Society for the Protection of Birds.

Trevelyan, R.J. (1995) 'The feeding ecology of Stephen's lory and nectar availability in its food plants'. *Biological Journal of the Linnean Society* 56: 185–197.

Ujvari, B., Brown, G., Shine, R. and Madsen, T. (2016). 'Floods and famine: Climate-induced collapse of a tropical predator-prey community'. *Functional Ecology* 30(3): 453–458.

Varnham, K. (2010). *Invasive Rats on Tropical Islands: Their History, Ecology, Impacts and Eradication*. RSPB Research Report No. 41. Sandy, UK: Royal Society for the Protection of Birds.

Waldren, S., Florence, J. and Chepstow-Lusty, A.J. (1995). 'A comparison of the vegetation communities from the islands of the Pitcairn Group'. *Biological Journal of the Linnean Society* 56: 121–144.

Weigelt, P., Jetz, W. and Kreft, H. (2013). 'Bioclimatic and physical characterization of the world's islands'. *Proceedings of the National Academy of Sciences of the United States of America* 110(38): 15307–15312.

Yackel Adams, A.A., Stanford, J.W., Wiewel, A.S. and Rodda, G.H. (2011). 'Modelling detectability of kiore (*Rattus exulans*) on Aguiguan, Mariana Islands, to inform possible eradication and monitoring efforts'. *New Zealand Journal of Ecology* 35(2): 145–152.

Bait colour and moisture do not affect bait acceptance by introduced Pacific rats (*Rattus exulans*) at Henderson Island, Pitcairn Islands

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Abstract Rodent eradications are a useful tool for the restoration of native biodiversity on islands, but occasionally these operations incur non-target mortality. Changes in cereal bait colour could potentially mitigate these impacts but must not compromise the eradication operation. Changing bait colour may reduce mortality of Henderson crakes (*Zapornia atra*), an endemic globally threatened flightless bird on Henderson Island, Pitcairn Islands, South Pacific Ocean. Crakes had high non-target mortality in a failed 2011 rat eradication operation and consumed fewer blue than green cereal pellets. We examined which cereal bait properties influenced its acceptance by captive Pacific rats (*Rattus exulans*) on Henderson Island. We held 82 Pacific rats from Henderson Island in captivity and provided them with non-toxic cereal bait pellets of varying properties (blue or green, moist or dry). We estimated the proportion of rats consuming bait using logistic generalised linear mixed models. We found no effect of sex, females' reproductive status, bait colour or bait moisture on rats' willingness to consume baits. Rats' bait consumption was unaffected by cereal bait properties (colour or moisture). The use of blue bait is unlikely to affect future eradication operational success but may reduce non-target mortality of Henderson crakes. Timing cereal bait distribution in relation to precipitation may also reduce crake mortality without compromising palatability to rats.

Keywords: baits, Henderson Island, island restoration, non-target safety, rat eradication

INTRODUCTION

The eradication of introduced rodents is a common conservation intervention, especially on islands, and has been accomplished on > 580 islands worldwide (DIISE, 2016), with benefits to native biodiversity (Lavers, et al., 2010; Buxton, et al., 2014; Jones, et al., 2016). In some cases, eradication operations may result in non-target morality, and mitigation can include housing captive populations of species likely to be affected, or by using cereal pellets that are less palatable to non-target species (Empson & Miskelly, 1999; Hoare & Hare, 2006; Pitt, et al., 2015; Oppel, et al., 2016a; Oppel, et al., 2016b). In such cases, it is crucial that the mitigation measures do not decrease the likelihood of a successful rodent eradication operation, and that the rodents are exposed to a sufficient quantity of cereal bait, are susceptible to the rodenticide used, and will consume a sufficient dose.

Rarely, though more so in the tropics and subtropics, these eradication operations fail to remove rodents for a variety of operational, biological, and environmental reasons (Holmes, et al., 2015). Eradication success in the tropics is generally lower than in temperate systems because there is less seasonal change in the environment, and consequently a less predictable period of food-resource limitation, which is the ideal time for an eradication operation (Holmes, et al., 2015; Russell & Holmes, 2015). Consequently, understanding which factors may influence rodents' acquisition of a lethal dose of bait are crucial for improving the probability of success in tropical systems (Lamoreux, et al., 2006).

Factors affecting bait acceptance by Pacific rats (*Rattus exulans*), a common introduced rodent in the Pacific Ocean tropics and subtropics (Atkinson, 1985; Varnham, 2010; Keitt, et al., 2015), are poorly known. A number of factors can influence rat food choice, including physical characteristics such as bait colour and hardness (Booth, et al., 1974; Clapperton, 2006; Hegab, et al., 2014). Murine rodents (including rats and mice) have colour vision, including sensitivity in the UV range (Jacobs, 1993; Jacobs, 2009), and there is evidence that cereal bait colour does affect the likelihood of acceptance by rats (Hegab, et al., 2014).

Blue or green cereal pellets are the most effective at reducing avian non-target mortalities, but there is considerable variation in bait attraction among species. Kea (*Nestor notabilis*) and weka (*Gallirallus australis*) were less likely to eat green pellets than blue (Hartley, et al., 2000; Weser & Ross, 2013), whereas North Island robins (*Petroica longipes*) and Henderson crakes (*Zapornia atra*) were less likely to consume blue pellets than green (Hartley, et al., 1999; Oppel, et al., 2016b). Henderson Island, part of the Pitcairn Islands in the South Pacific Ocean, was the site of a failed eradication operation for Pacific rat in 2011, which also resulted in non-target mortality of Henderson crakes (Amos, et al., 2016; Oppel, et al., 2016a). Subsequent work found that Henderson crakes consumed less blue bait than green, and did not consume dry pellets (Oppel, et al., 2016b).

Here we report on the factors affecting Pacific rat bait acceptance on Henderson Island, Pitcairn Islands, South Pacific Ocean. Our goal was to compare Pacific rats' acceptance of both moist and dry cereal pellets of these two colours to determine whether measures to reduce the non-target mortality of Henderson crakes might affect success of future eradication operations on Henderson.

METHODS

Rat capture & acclimatisation

Rats were captured on Henderson Island's plateau using either Tomahawk (27 × 16 × 13 cm, Metal Rat Cage Trap, Key Industries, Auckland, New Zealand) or Sherman (22.9 × 8.9 × 7.6 cm, Sherman Traps Inc., Tallahassee, Florida, USA) live traps baited with a 2 × 2 cm piece of coconut (*Cocos nucifera*). Individual rats' knowledge of coconut prior to its presentation during the cage trial is presumed to be limited, because the areas where rats were captured were > 500 m away from the nearest coconut grove on

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the island. We chose coconut as bait because it was easily available and highly effective in attracting rats, while alternative baits (*Pandanus tectorius* fruit, peanut butter, chocolate, semolina, soap, and mixtures thereof) largely failed to capture sufficient individuals. Although the choice of coconut as trap bait may have predisposed some trapped rats to exhibit universal acceptance of coconut in the trials (see Results), particularly given the large movements possible in this population (Oppel, et al., 2019), the use of a non-natural food source (e.g., peanut butter, chocolate) may have resulted in capturing only curious or bold rats who will readily accept new food items, which may have biased our assessment of acceptance rate of bait pellets, another novel food item (Booth*,* et al.*,* 1974).

We held 81 rats in captivity for 12 days each during October–November 2015. Rats were weighed using a spring balance to the nearest 1 g, fitted with a uniquely numbered ear tag, and their sex was determined from external anatomy. They were first allowed to acclimatise for four days in sex-specific communal wire cages (70 × 60 × 30 cm) of up to four individuals, where they were fed commercial rodent food (Rabbit and Guinea Pig Muesli, Topflite, Oamaru, New Zealand) *ad libitum* in a single ceramic food dish and water was provided by both a commercial water dispenser (Critter Canteen, SuperPet, Walnut Creek, California, USA) and a large clamshell. Each cage contained four hollow *Pandanus tectorius* logs, with small, loose pieces of coconut bark providing cover and visual barriers. After four days, rats were weighed as before, and moved to individual wire cages (70 × 50 × 30 cm) with the same food and water regime, and environmental enrichment. Rats were considered to have acclimated if, after three days in individual cages, their mass differed by < 10% compared to their mass at capture, and we observed no anomalous behaviour. Rats that had lost > 10% of body mass were allowed two additional days to acclimate and were then reassessed using the same criteria. Any individuals that did not acclimate were not subjected to the experimental trial and were euthanised by cervical dislocation and used in other research (Lavers*,* et al.*,* 2016).

Bait acceptance trial

Individuals which acclimated were assigned randomly to one of four treatment groups based on combinations of bait colour (green or blue) and moisture (moist or dry); cereal bait pellets (nontoxic ~2 g Pestoff 20R, Animal Control Products, Whanganui, New Zealand) were otherwise identical. The pellets were surface coated with dye and did not lose their colour after soaking in water. The green dye was a composite of tartrazine powder (with the Chemical Abstracts Service (CAS) number 1934-21-0), Brilliant Blue powder (CAS number 3844-45-9) and sodium sulphate (CAS number 7757-82-6), and the blue dye was Hexacol Indigo Carmine Supra Blue R2613 (CAS number 860-22-0). Cereal pellets were presented to rats either dry, as manufactured, or moist, after being soaked in water for three hours. Rats received fresh cereal bait pellets and natural foods daily.

Between 06:00-08:00 (UTC-8) on Day 1 of the trial, rats were provided with sufficient natural food (coconut, *Myrsine hosakae* seeds, and whole *Pandanus tectorius* fruit) to ensure that they were sated but not consuming any single food item completely, and a single cereal bait pellet. Food consumption was assessed between 06:00-08:00 on Day 2: if the rat had consumed a significant portion of a pellet (~1 g), the individual's trial was ended and the rat was euthanised. If the bait was not eaten, the rat received a fresh batch of natural food of the same amount as on Day 1 while the number of pellets was increased to five in an attempt to overcome neophobia and increase the number of trial rats consuming bait. This procedure was continued for two more days if needed, with rats receiving 10 and 20 pellets on Days 3 and 4, respectively. Rats that had not eaten bait after Day 4 were given five bait pellets and all natural foods, with the exception of coconut (which was accepted universally), on experimental Day 5. The trial ended the next day regardless of outcome.

Food consumption was monitored daily and all remaining food from the previous night removed and the cage inspected to ensure no natural food item was completely consumed, and any remaining bait pellets counted. The remains of any partially eaten pellets were inspected and the amount eaten estimated to the nearest 25%. Where a natural food item was completely consumed, the result was ignored and the test repeated with the same individual.

All captive rats were humanely euthanised by cervical dislocation at the end of the trial. Females were examined internally to determine reproductive status: breeding was indicated by the presence of foetal pups, a highly vascularised uterus, or lactation.

Statistical analysis

We used logistic generalised linear mixed-effects models (GLMMs) with a logit link to test whether bait acceptance (yes/no) varied as a function of the following fixed factors: sex (female/male), bait colour (blue/green), and bait moisture (moist/dry), and trial day. We treated 'individual' as a random effect to account for potential serial autocorrelation (Bolker*,* et al.*,* 2009). We included main effects only, as the effective sample size would reduce the statistical power to detect the effect of interactions in our dataset. We constructed a series of models with varying biologically meaningful combinations of the terms above, as well as an intercept-only model (Table 1) and evaluated

Table 1 The ranked set of candidate models for examining captive Pacific rats' acceptance of bait on Henderson Island. Models with $\Delta AIC_c < 2$ were considered competitive (i.e. the top 3).

Model	k	AIC_c	ΔAIC_c	w_i
Intercept only	2	214.80	0.00	0.33
Moisture	3	215.89	1.10	0.19
Sex	3	216.62	1.82	0.13
Colour	3	216.85	2.05	0.12
Sex + Moisture	4	217.72	2.92	0.08
Moisture + Colour	4	217.96	3.16	0.07
Sex + Colour	4	218.68	3.89	0.05
Day	5	219.15	4.35	0.03
Sex + Colour + Moisture	5	219.80	5.00	0.03
Day + Moisture	6	220.54	5.75	0.02
Day + Sex	6	221.11	6.31	0.01
Day + Colour	6	221.30	6.50	0.01
Day + Sex + Moisture	7	222.49	7.69	0.01
Day + Moisture + Colour	7	222.66	7.87	0.01
Day + Sex + Colour	7	223.23	8.43	<0.01
Day + Sex + Colour+Moisture	8	224.62	9.82	<0.01

k: number of parameters, AIC_c: Akaike's Information Criterion adjusted for small sample size, ΔAIC_c: difference between each model and the most parsimonious model, w_i: Akaike model weight.

them in a multi-model selection framework using Akaike's Information Criterion adjusted for small sample size (AIC_c) (Burnham & Anderson 2002). Models with $\Delta AIC_c < 2$ were considered competitive. All models were fit using Laplace approximation in the package lme4 (Bates, et al., 2014) in R 3.3.0 (R Core Team, 2017), and we present mean parameter estimates (β) ± standard error.

RESULTS

We captured 82 rats of which 81 acclimated to the captive trial. Overall, 48% of captive rats (n = 39) consumed the non-toxic cereal pellets. The intercept-only model, where bait consumption varied among individuals, but not with any other factors, received the most support, but models that included the single terms for sex, bait colour, and bait moisture had $\Delta AIC_c < 2.0$ (Table 1). Using each of these single-factor models, there was no difference in bait acceptance between sexes (males: β = 0.532 ± 0.012, females: β = 0.536 ± 0.014), and no effect of bait colour (blue: β = 0.533 ± 0.018, green: β = 0.534 ± 0.018), or moisture (dry: β = 0.540 ± 0.015, moist: β = 0.529 ± 0.012; Fig. 1). Bait acceptance did not differ with females' reproductive status (calculated parameter estimates for breeding: β = 0.539 ± 0.033, n = 17; not breeding: β = 0.543 ± 0.025, n = 22; one individual not of breeding age: β = 0.520). All models that included trial day had $\Delta AIC_c > 4$, so were not considered further (Table 1).

DISCUSSION

We found no evidence for bait colour, moisture, sex, or reproductive status affecting the consumption of bait pellets by captive Pacific rats. Blue and green bait pellets are frequently used in rodent eradication operations (Clapperton, et al., 2015), and the use of blue pellets may therefore reduce the reported non-target mortality among Henderson crakes (Oppel, et al., 2016b) without affecting the efficacy of rat eradication operations.

Rats ingested dry and moist pellets equally, which is important operationally as Henderson crakes do not consume dry pellets (Oppel, et al., 2016b). While rainfall patterns on Henderson are unpredictable and aseasonal (Spencer, 1995), targeting any future eradication operation at periods of low rainfall is unlikely to affect the outcome for rodents, but may reduce the risk of non-target mortality. In the longer term (i.e. longer than the four days used in our captive trial), rainfall will increase the degradation rates of bait, thereby reducing rats' exposure to bait, regardless of their inherent preference to consume bait that is dry or wet (Berentsen, et al., 2014).

The aseasonal breeding often found on tropical islands means that baiting operations are more likely to include breeding females than operations in temperate regions. Concern has been expressed that pregnant and lactating rodents are less likely to eat bait if their nutritional needs are not met by the bait matrix (Keitt, et al., 2015), though this also assumes that rats could identify the nutritional content of bait pellets without consuming a lethal dose (i.e., one pellet; Amos, et al., 2016). Our results suggest that not only are female Pacific rats as likely to consume bait pellets as males, but that, at least for this particular bait formulation, females' reproductive status is unlikely to influence bait acceptance.

Importantly, while only 48% of trial rats consumed bait pellets, this general acceptance rate cannot be used to infer potential acceptance rates in free-ranging rats during an eradication operation. Evidence of higher or lower bait acceptance rates in the wild than in cage trials is equivocal (Clapperton, 2006) but several important limitations

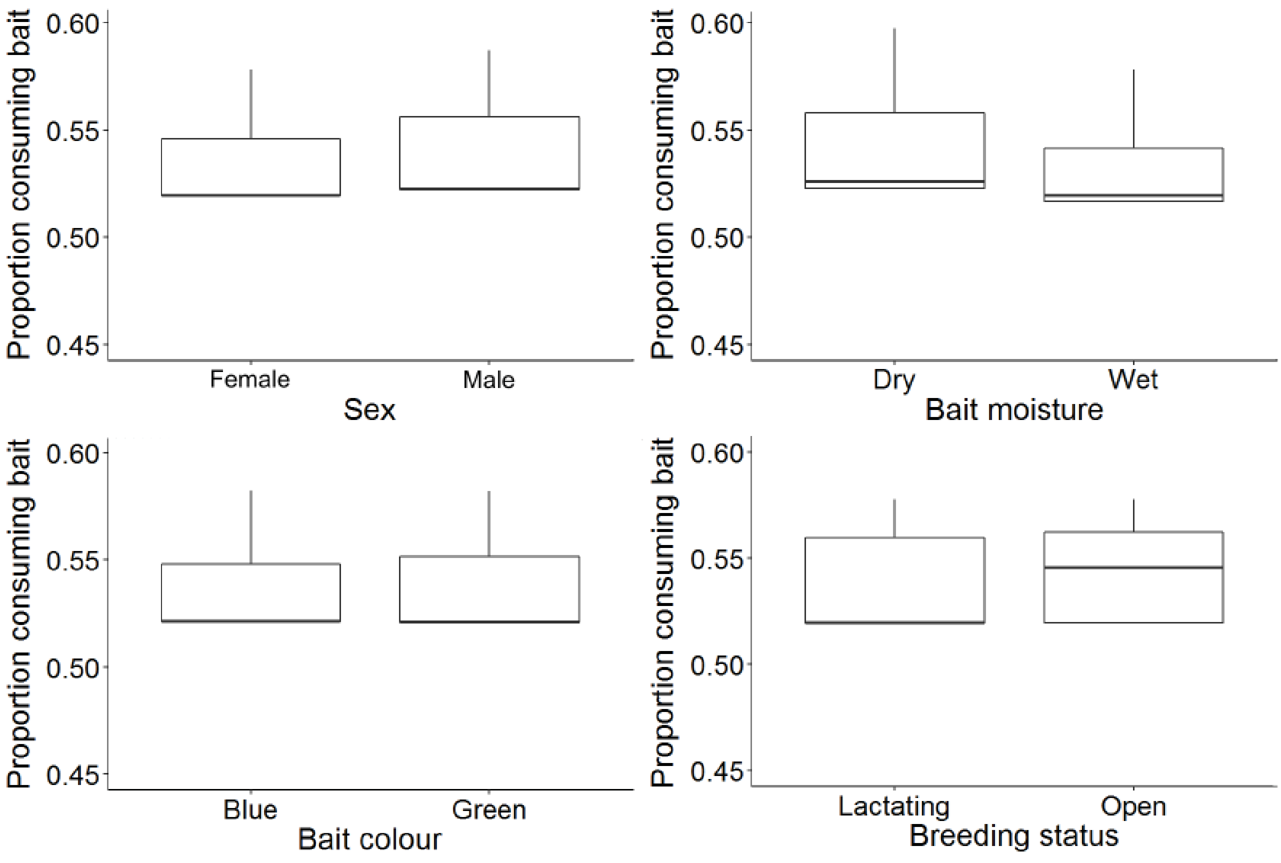

Fig. 1 There was no effect of rats' sex, breeding status, bait colour, or bait moisture on the proportion of Pacific rats consuming bait. Values are from generalised linear mixed models. Dark bars are the median, boxes are the interquartile range, and whiskers the range.

of cage trials have the potential to lower acceptance below what would be typical in the wild. For example, the provisioning of commercial rodent food during the acclimatisation phase ensures an unbiased test of food preference because test subjects are not food stressed (which results in selection of food items based on dietary deficiencies rather than food palatability). However, the chemical composition of food plays an important role in diet selection in free-ranging rats, with individuals self-selecting food based on physiological need (Rozin, 1976). Another major limitation of cage trials, specifically with regard to acceptance of novel food items such as bait, is the absence of social learning. While rats are predominantly solitary foragers, the identification and adoption of novel food items is heavily influenced by social interactions with conspecifics (Galef Jr, 1996). Cage trials should therefore only be considered as useful tools for identifying potential problems that can be explored further by field trials. On Palmyra Atoll, for example, rats were found to prefer coconut over cereal bait pellets in cage trials, but later field trials found adequate bait uptake, and a toxic cereal bait eradication was successful (Buckelew*,* et al*.,* 2006; Alifano & Wegmann, 2011).

CONCLUSIONS

Our findings suggest Pacific rats have no preference between green or blue bait pellets, nor if bait is moist or dry. This suggests that individual variation is a significant driver of bait acceptance, regardless of other demographic parameters (Nathan, 2016). While a baiting operation timed when rats are breeding carries increased risks and is preferably avoided, pregnant or lactating females are as likely to accept bait as non-pregnant females. Any future operation on Henderson Island should use blue bait pellets, and time the operation for dry conditions, in order to reduce non-target mortality of Henderson crakes without affecting rat bait acceptance.

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REFERENCES

Alifano, A. and Wegmann, A.S. (2011). *The Ecotoxicology and Palatability of Two Rodenticide Bait Products: Field-based Assessment at Palmyra Atoll*. Santa Cruz, California: Island Conservation.

Amos, W., Nichols, H.J., Churchyard, T. and Brooke, M.de L. (2016). 'Rat eradication comes within a whisker! A case study from the South Pacific'. *Royal Society Open Science* 3: 160110.

Atkinson, I.A.E. (1985). 'The Spread of Commensal Species of *Rattus* to Oceanic Islands and their Effects on Island Avifaunas'. In*:* P.J. Moors (ed.) *Conservation of Island Birds*. pp. 35–81. Cambridge: ICBP Technical Publication No. 3.

Bates, D., Machler, M., Bolker, B. and Walker, S. (2014). 'Fitting Linear Mixed-Effects Models using lme4'. <*arXiv*:http://arxiv.org/abs/1406.5823>.

Berentsen, A.R., Pitt, W.C., Eisemann, J.D. and Engeman, R.M. (2014). 'Longevity of rodenticide bait pellets in a tropical environment following a rat eradication program'. *Environmental Science and Pollution Research* 21(3): 2283–2288.

Bolker, B.M., Brooks, M.E., Clark, C.J., Geange, S.W., Poulsen, J.R., Stevens, M.H.H. and White, J.-S.S. (2009). 'Generalized linear mixed models: a practical guide for ecology and evolution'. *Trends in Ecology and Evolution* 24: 127–135.

Booth, D.A., Stoloff, R. and Nicholls, J. (1974). 'Dietary flavor acceptance in infant rats established by association with effects of nutrient composition'. *Physiological Psychology* 2(3A): 313–319.

Buckelew, S.L., Howald, G.R., Wegmann, A.S., Sheppard, J., Curl, J., McClelland, P., Tershy, B., Swift, K., Campbell, E. and Flint, B. (2006). *Progress in Palmyra Atoll Restoration: Rat Eradication Trial 2005.* Santa Cruz, California, USA: Island Conservation.

Burnham, K.P. and Anderson, D.R. (2002). *Model Selection and Multimodel Inference: A Practical Information-theoretic Approach,* New York: Springer.

Buxton, R.T., Jones, C.J., Moller, H. and Towns, D.R. (2014). 'Drivers of seabird population recovery on New Zealand islands after predator eradication'. *Conservation Biology* 28(2): 333–344.

Clapperton, B.K. (2006). 'A Review of the Current Knowledge of Rodent Behaviour in Relation to Control Devices'. *Science for Conservation* 263. Wellington, NZ: Department of Conservation.

Clapperton, B.K., Day, T.D., Morgan, D.K., Huddart, F., Cox, N. and Matthews, L.R. (2015). 'Palatability and efficacy to possums and rats of pest control baits containing bird repellents'. *New Zealand Journal of Zoology* 42: 104–118.

DIISE (2016). 'The Database of Island Invasive Species Eradications'. Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group. University of Auckland and Landcare Research New Zealand. <http://diise.islandconservation.org>. Accessed 1 July 2016.

Empson, R.A. and Miskelly, C.M. (1999). 'The risks, cost and benefits of using brodifacoum to eradicate rats from Kapiti Island, New Zealand'. *New Zealand Journal of Ecology* 23(2): 241–254.

Galef Jr, B.G. (1996). 'Social enhancement of food preferences in Norway rats: a brief review'. In*:* C.M. Heyes and B.G. Galef Jr (eds.) *Social Learning in Animals: The Roots of Culture,* pp. 49–64. Amsterdam: Elsevier.

Hartley, L., O'Connor, C., Waas, J. and Matthews, L. (1999). 'Colour preferences in North Island robins (*Petroica australis*): Implications for deterring birds from poisonous baits'. *New Zealand Journal of Ecology* 23(2): 255–259.

Hartley, L., Waas, J., O'Connor, C. and Matthews, L. (2000). 'Colour preferences and coloured bait consumption by weka *Gallirallus australis*, an endemic New Zealand rail'. *Biological Conservation* 93(2): 255–263.

Hegab, I.M., Pan, H., Dong, J., Wang, A., Yin, B., Yang, S. and Wei, W. (2014). 'Effects of physical attributes and chemical composition of novel foods on food selection by Norway rats (*Rattus norvegicus*)'. *Journal of Pest Science* 87(1): 99–106.

Hoare, J.M. and Hare, K.M. (2006). 'The impact of brodifacoum on non-target wildlife: gaps in knowledge'. *New Zealand Journal of Ecology* 30(2): 157–167.

Holmes, N.D., Griffiths, R., Pott, M., Alifano, A., Will, D., Wegmann, A.S. and Russell, J.C. (2015). 'Factors associated with rodent eradication failure'. *Biological Conservation* 185: 8–16.

Jacobs, G.H. (1993). 'The distribution and nature of colour vision among the mammals'. *Biological Reviews* 68(3): 413–471.

Jacobs, G.H. (2009). 'Evolution of colour vision in mammals'. *Philosophical Transactions of the Royal Society B: Biological Sciences* 364(1531): 2957–2967.

Jones, H.P., Holmes, N.D., Butchart, S.H.M., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P.E., Cuthbert, R.J., Ebbert, S.E., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R. and Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences of the United States of America* 113(15): 4033–4038.

Keitt, B.S., Griffiths, R., Boudjelas, S., Broome, K.G., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 182: 17–26.

Lamoreux, J.F., Morrison, J.C., Ricketts, T.H., Olson, D.M., Dinerstein, E., McKnight, M.W. and Shugart, H.H. (2006). 'Global tests of biodiversity concordance and the importance of endemism'. *Nature* 440(7081): 212–214.

Lavers, J.L., Wilcox, C. and Donlan, C.J. (2010). 'Bird demographic responses to predator removal programs'. *Biological Invasions* 12: 3839–3859.

Lavers, J.L., McClelland, G.T.W., MacKinnon, L., Bond, A.L., Oppel, S., Donaldson, A.H., Duffield, N.D., Forrest, A.K., Havery, S.J., O'Keefe, S., Skinner, A., Torr, N. and Warren, P. (2016). *Henderson Island Expedition Report: May–November 2015.* RSPB Research Report 57. Sandy, UK: RSPB Centre for Conservation Science.

Nathan, H.W. (2016) 'Detection Probability of Invasive Ship Rats: Biological Causation and Management Implications'. Ph.D. thesis. Auckland, NZ: University of Auckland.

Oppel, S., Bond, A.L., Brooke, M.de L., Harrison, G., Vickery, J.A. and Cuthbert, R.J. (2016a). 'Temporary captive population and rapid population recovery of an endemic flightless rail after a rodent eradication operation using aerially distributed poison bait'. *Biological Conservation* 204B: 442–448.

Oppel, S., Lavers, J.L., Bond, A.L. and Harrison, G. (2016b). 'Reducing the primary exposure risk of Henderson crakes (*Zapornia atra*) during aerial broadcast eradication by selecting appropriate bait colour'. *Wildlife Research* 43: 298–303.

Oppel, S., McClelland, G.T.W., Lavers, J.L., Churchyard, T., Donaldson, A.H., Duffield, N.D., Havery, S., Kelly, J., Proud, T., Russell, J., Vickery, J.A. and Bond, A.L. (2019). 'Seasonal variation in movements and survival of invasive Pacific rats on a sub-tropical island: implications for eradication'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 200–208. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Pitt, W.C., Berentsen, A.R., Shiels, A.B., Volker, S.F., Eisemann, J.D., Wegmann, A.S. and Howald, G.R. (2015). 'Non-target species mortality and the measurement of brodifacoum rodenticide residues after a rat (*Rattus rattus*) eradication on Palmyra Atoll, tropical Pacific'. *Biological Conservation* 185: 36–46.

R Core Team (2017). *R: A Language and Environment for Statistical Computing. Version 3.4.0 [computer program].* Vienna, Austria: R Foundation for Statistical Computing.

Rozin, P. (1976). 'The selection of foods by rats, humans, and other animals'. *Advances in the Study of Behavior* 6: 21–76.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Spencer, T. (1995). 'The Pitcairn Islands, South Pacific Ocean: plate tectonic and climatic contexts'. *Biological Journal of the Linnean Society* 56 :13–42.

Varnham, K. (2010). *Invasive Rats on Tropical Islands: Their History, Ecology, Impacts and Eradication*. RSPB Research Report No. 41. Sandy, UK: Royal Society for the Protection of Birds.

Weser, C. and Ross, J.G. (2013). 'The effect of colour on bait consumption of kea (*Nestor notabilis*): Implications for deterring birds from toxic baits'. *New Zealand Journal of Zoology* 40(2): 137–144.

Containment of invasive grey squirrels in Scotland: meeting the challenge

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Abstract Saving Scotland's Red Squirrels (SSRS), launched in 2009, is a project to stop the decline of core populations of Scotland's native red squirrel. It is a partnership project between Scottish Wildlife Trust, Scottish Natural Heritage, Forestry Commission Scotland, RSPB Scotland, Scottish Land & Estates and the Red Squirrel Survival Trust. The aim is the containment of the invasive non-native grey squirrel, which poses a dual threat to red squirrels through competition and disease transmission. Grey squirrels have replaced red squirrels over much of their former range in England, Wales, Ireland and central Scotland. SSRS controls grey squirrels at a landscape-scale in three strategically selected zones: in north-east Scotland, where the aim is eradication of an isolated grey squirrel population; coast to coast along the Highland Boundary Fault the aim is to prevent northwards incursion of grey squirrels into the Scottish Highlands and Argyll, where red squirrel is still the only species; and in southern Scotland, the aim is now to prevent replacement of priority red squirrel populations by focussing control in areas identified as having the best prospects for the long-term maintenance of red squirrel populations. Control methods involve live cage-trapping combined with humane dispatch. The control network comprises SSRS and Forestry Commission controllers, private landowners supported by EU/government funding and a large number of individual volunteers. The work is dependent on wide public acceptance and active volunteer support. To date SSRS has been successful at significantly reducing grey squirrel geographic range and occupancy in NE Scotland and as well as reducing the incidence of grey squirrels north of the 'Highland Line' to no more than the occasional occurrence. In southern Scotland grey squirrel control has contributed to the maintenance of red squirrel populations despite the continued spread of squirrelpox in grey squirrels. The major challenge now is sustaining the level of grey squirrel control needed to secure Scotland's red squirrel populations in the long term. A new project phase started in 2017, focused on building community action networks until such a time as alternatives means of controlling grey squirrel numbers and disease impacts become widely available.

Keywords: adaptive management, community engagement, land manager, sustainability, trapping effort, volunteer

INTRODUCTION

Grey squirrels (*Sciurus carolinensis*) were introduced into Britain in the 1890s from the US and Canada, including to several release sites in Scotland (Middleton, 1930; Middleton, 1931). The impact of grey squirrels on native red squirrel (*Sciurus vulgaris*) populations was documented relatively early after their introduction (Middleton, 1931; Shorten, 1962), but their range expansion was initially quite modest (Gurnell, 1987). The role of grey squirrels in the replacement of red squirrels was possibly not fully recognised until the 1980s (Lloyd, 1983; Skelcher, 1997; Reynolds, 1998), by which time grey squirrels occupied much of southern and central England and Wales and central Scotland (Lloyd, 1983).

The factors leading to the replacement of red by grey squirrels have been the subject of extensive research (reviewed by Gurnell, et al., 2014b). The evidence indicates that competition with grey squirrels for food resources alone can account for the loss of red squirrels from many forests (Bryce, et al., 2001; Wauters, et al., 2002). However, added to this is the threat of squirrelpox virus, which is carried by grey squirrels and is highly pathogenic to red squirrels (Sainsbury, et al., 2000, Thomas, et al., 2003), greatly enhancing the speed of replacement (Rushton, et al., 2006).

Red squirrels have been protected under UK law since the 1930s and bounty schemes were enlisted to combat increasing grey squirrel numbers in the 1950s (Sheail, 1999). However, low-level, sporadic control has failed to halt the spread of grey squirrels (Lawton & Rochford, 2007). Grey squirrels are already widespread and abundant throughout much of the UK and eradication is not considered to be a realistic option (Gurnell & Pepper, 1993; Pepper & Patterson, 1998). EU Regulation 1143/2014 on Invasive Alien Species lists grey squirrels as species of Union concern, hence Member States are required to take concerted management action to ensure they do not spread any further and to minimise the harm they cause to the environment.

Large-scale control and containment of grey squirrels was originally seen as an interim approach, whilst more sustainable, long-term control measures were developed (Scottish Squirrel Group, 2004; Scottish Natural Heritage, 2010). However, in the absence of a squirrelpox vaccine or immuno-contraceptive, there has been growing support for targeted grey squirrel control to protect red squirrel populations (Scottish Natural Heritage, 2010). Following public consultation, a draft strategy for grey squirrel control in Scotland was published (Scottish Natural Heritage, 2010), which focuses on targeted control to maximise the benefits for red squirrels.

A collaborative project under the heading 'Saving Scotland's Red Squirrels' (SSRS), was formalised in 2009. It is a partnership comprising the Scottish Wildlife Trust (SWT), Scottish Natural Heritage, Forestry Commission Scotland, RSPB Scotland, Scottish Land & Estates and the Red Squirrel Survival Trust. SSRS has become the principal means of coordinating red squirrel protection in Scotland. The focus of SSRS is on applied conservation action, but SSRS has made a concerted effort to collate records and monitor squirrel populations and has worked closely with researchers to inform an adaptive approach. A key challenge to SSRS has been to assess the efficacy and sustainability of control measures. This paper explores some of the work carried out to address these challenges and highlights some of the learning to date.

THE SSRS APPROACH

Co-ordination of grey squirrel control

The main focus of SSRS activity is the co-ordination of grey squirrel control. SSRS aims to co-ordinate grey

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squirrel control across three strategic control zones (Fig. 1). The aims vary between zones reflecting the degree to which grey squirrels are already established. The Highlands of Scotland are currently free of grey squirrels; grey squirrels are long established in central Scotland (with introductions between 1892–1919) and had spread throughout much of the Central Lowlands by the 1980s; grey squirrels have spread into South Scotland from both the north and from northern England in recent years (Gurnell, et al., 2014a). The grey squirrel population in NE Scotland (Aberdeen and Aberdeenshire) has been recorded for about 30 years (Lloyd, 1983; Staines, 1986), and there is evidence it originates from a separate introduction rather than having spread from grey squirrels elsewhere in Scotland (Signorile, 2013). Squirrelpox was first reported to have crossed from northern England into Scotland in 2005, with the first cases in red squirrels observed in 2007 (McInnes, et al., 2009). The original SSRS aims are listed as follows, although these have been adapted in light of experience as is discussed later.

North-east (NE) Scotland – the original aim was to halt the spread of grey squirrels outwards from the city of Aberdeen;

Central Lowlands – SSRS's work in the Central Lowlands aims to contain the northward spread of grey squirrels into the Highlands and Argyll by carrying out control from coast to coast; along what is referred to as the 'Highland Line' a zone of control extending for some 160 km from just north of Glasgow to Montrose on the east coast.

Southern Scotland – the initial aim was to contain the spread of squirrelpox virus in south Scotland.

Grey squirrel control is currently carried out by a mixture of project staff, land managers and volunteers:

- Working with up to 197 landowners under five-year EU and Scottish Rural Development Programme funding (SRDP);
- Co-ordination via five conservation officers, six full-time and five part-time grey squirrel control officers;
- A trap-loan scheme involving up to 200 landowners, and 500 individuals.

Trapping is also carried out by Forest Enterprise Scotland at key sites on the National Forest Estate. Figure 2 illustrates the coverage of grey squirrel control initiated by SSRS by 2012 (from Tonkin, et al., 2015).

With a view to the long term, SSRS has sought to encourage land managers to carry out grey squirrel control on their own land. Regional conservation staff provide support to landowners applying for funding available through the SRDP to help cover the costs. SRDP contracts require land managers to operate an appropriate number of traps (as advised by SSRS staff) for a minimum number of sessions per year (usually five or six). All the traps are to be set for a minimum number of days (usually 10). SSRS control officers trap with landowner permission, in key gaps in the landowner protection network (Fig. 2). Most of the grey squirrel control occurs between April and the end of September, when grey squirrels are easier to catch. Due to the lack of specificity of other methods and animal welfare considerations (Central Science Laboratory, 2009), the SSRS Standard Operating Procedures specify the use of cage trapping and humane dispatch.

Trapping by SSRS control officers has typically followed the approach of five days pre-baiting and then trapping continuously until no or few further grey squirrels are caught. Traps are then revisited on a rotation. The traps are not located at a standardised density, but instead are grouped in areas of preferred grey squirrel habitats in the target zones. In South Scotland grey squirrel control by SSRS control officers was initially carried out in the area buffering known squirrelpox seropositive cases. The grey squirrels were then sent for laboratory testing for squirrelpox. Hence control effort was reactive and did not take place in the same locations across time.

In NE Scotland, in particular, SSRS is working in peri-urban and urban areas. Initially this created a challenge because large areas of wooded habitats in private gardens and parks were difficult to access, leaving reservoirs of grey squirrels. To address this challenge and to harness the community enthusiasm in other areas, SSRS has successfully instigated a trap-loan scheme. Under this scheme householders take responsibility for setting and monitoring traps (supplied by SSRS) in their garden. They

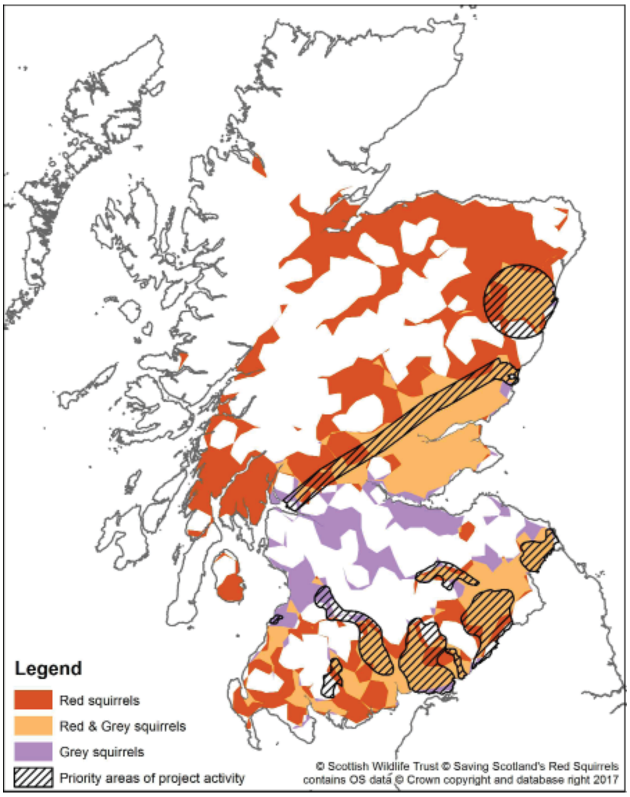

Fig. 1 Map of Saving Scotland's Red Squirrels project control zones.

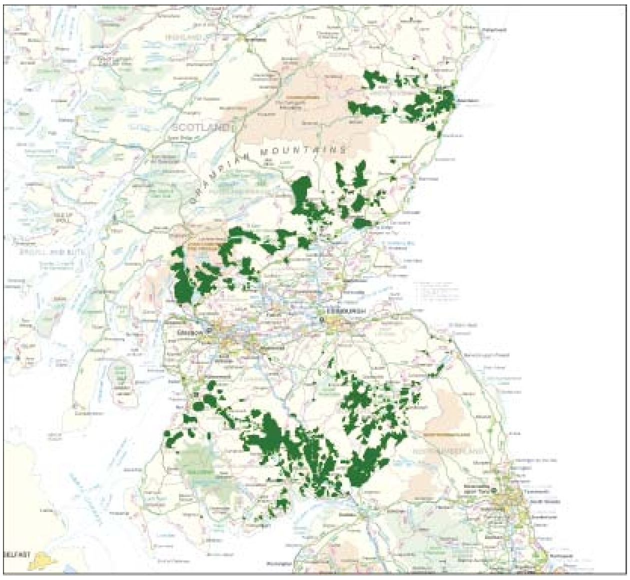

Fig. 2 Red squirrel protection network established across Scotland by 2012 (from Tonkin, et al., 2015).

are matched with a local trapper (gamekeeper or control officer) who is available to carry out humane dispatch. Project staff have also developed innovative trap designs for trapping grey squirrels in city parks to avoid drawing attention to traps.

Grey squirrel control in public spaces and, in particular, the involvement of volunteers requires a high degree of public acceptance, which is not guaranteed (Bertolino & Genovesi, 2003). A targeted approach to control was broadly supported in a public consultation (Scottish Natural Heritage, 2010) and public surveys in Aberdeen and Aberdeenshire have established that despite residents enjoying seeing grey squirrels, there is an appreciation of the need for grey squirrel control due to their impact on red squirrels (Ashbrook Research Consultancy Ltd, 2010).

Evaluation of control measures

Alongside establishing the network of grey squirrel control, SSRS has sought to collect evidence that this work is benefitting red squirrels. This has been critical for securing public funding for grey squirrel control. Three methods were employed by the SSRS in order to evaluate the efficacy of grey squirrel control:

- Evaluation of grey squirrel capture probability from trapping data;
- Annual (presence/absence) monitoring of red and grey squirrel occupancy in the three project areas, and
- Public sightings of squirrels across Scotland that have been catalogued since 2007.

Annual (presence /absence) monitoring of red and grey squirrels has been co-ordinated by SSRS in NE Scotland and the Central Lowlands since 2011 and since 2013 in South Scotland. The surveys are intended to assess if there are changes in squirrel distributions that can be attributed to the project. Nearly 200 volunteers have been mobilised to carry out these surveys. A sample of 2 km ×2 km grid squares or 'tetrads' are surveyed across each control zone. Four baited feeder-boxes are permanently located in woodland within each tetrad. Each feeder-box is checked by volunteers three times over a period of six weeks each spring. Hairs are identified under a microscope and each tetrad is consequently allocated to one of the following four categories: "red squirrels only", "grey squirrels only", "both species", or "neither" species (Fig. 3, Shikhorshidi & Tonkin, 2018). The number of tetrads has increased over time, but comparison of the same tetrads over time enables detection of changes in squirrel distributions. Changes between years have been explored using a replacement index as per Usher, et al. (1992). A positive index represents a change in tetrad occupancy in favour of grey squirrels and a negative index, a change in favour of red squirrels (Usher, et al., 1992).

A programme of squirrelpox surveillance has also helped guide the work in South Scotland. In 2012, grey squirrels were sampled from a systematic sample of locations across the whole of South Scotland to try and establish the full extent of exposure to the virus (10 grey squirrels are sampled from one 10 km square in every 20 km × 20 km square across the region).

RESULTS

Evaluation of grey squirrel capture probability

The project initially aimed to gather data on grey squirrel trapping across all three control zones, however, inconsistencies in recording between the different project delivery models has made it problematic to fully assess the cumulative trapping effort that has been achieved. The control officers' data are the most reliable. The format of other records varies, effort is not always systematically recorded and problems have been encountered (data gathering and ownership) in accessing results of trapping from the land mangers supported by SRDP funding. Forestry Enterprise Scotland controllers' data are included with the control officers' data where this has been possible. The minimum total number of grey squirrels controlled and the trapping effort achieved have been estimated from collated data (SWT pers. comm.). It is estimated that between 2009 and 2016:

- Control officers provided c. 214,000 trap days (the number of traps multiplied by the number of days for which traps are set) and controlled c. 13,000 grey squirrels (Table 1); and
- Up to 197 SRDP contracts were established (Scottish Government Statistics, 2014), with those reporting accounting for 1.1 million trap days and having controlled c. 18,000 grey squirrels Table 1).

Those in receipt of trap loans have not consistently reported trapping effort, however, the trap loan scheme in NE Scotland has made a larger contribution to the red squirrel protection network than elsewhere, with trap loans here accounting for the removal of more than 1,700 grey squirrels between 2010 and 2016 (SWT pers. comm.).

Due to the scale of the task to follow up on missing trapping information, SSRS have sought to collect as complete trapping data as possible for four demonstration areas in order to assess the cumulative impact on grey squirrel capture probability (a proxy for abundance). The size of demonstration areas is not equivalent but as an illustration of control effort (control officer and landowner data), the total number of trap days in 2014 in NE Scotland demonstration area (55 km^2) was 6,614 trap days, in Tayside (222 km^2) was 15,004 trap days, in Argyll & Trossachs (278 km^2) was 6,482 trap days and in South Scotland (604 km^2) was 15,206 trap days (Table 2). Only the NE Scotland and Tayside demonstration areas had generated sufficient time series of data for detailed analysis by 2013 as reported in Tonkin, et al. (2013).

Using all the available trapping data for the NE Scotland Demonstration Area between 2007 and June 2013 and the Tayside Demonstration Area from 2010 to 2012, a GLMM was used to explore the relationship between the probability of grey squirrel capture and a range of explanatory variables including the cumulative control effort for each trap location, taking account of nearby captures. There was found to be a significant negative effect of cumulative control effort on the probability of grey squirrel capture in both areas (Tonkin, et al., 2013). In Tayside the GLMM coefficient was -1.54 (CI -1.99 – -1.09), and in NE Scotland was -0.34 (CI -0.51 – -0.18) (both on link scale of logit model). Effects were found to be stronger in areas with the highest cumulative control effort. In these areas in Tayside the mean capture probability was close to zero and in NE Scotland was seven-fold lower than areas with relatively low effort (Tonkin, et al., 2013,). These results support the premise that trapping is having the desired effect of reducing grey squirrel abundance.

The reactive pattern of trapping in South Scotland in response to detecting squirrelpox, makes the data from this region problematic for assessing the impact of trapping on grey squirrel abundance. Added to this, despite the scale of trapping effort in South Scotland, it was apparent that the virus was still spreading (White & Lurz, 2014; Tonkin, et al., 2015). Hence SSRS sought the help of researchers to assess if containment of the virus was a realistic objective. White & Lurz (2014) used a spatially explicit population model to explore the spread of the disease under a range of control scenarios and levels of effort. Simulated control

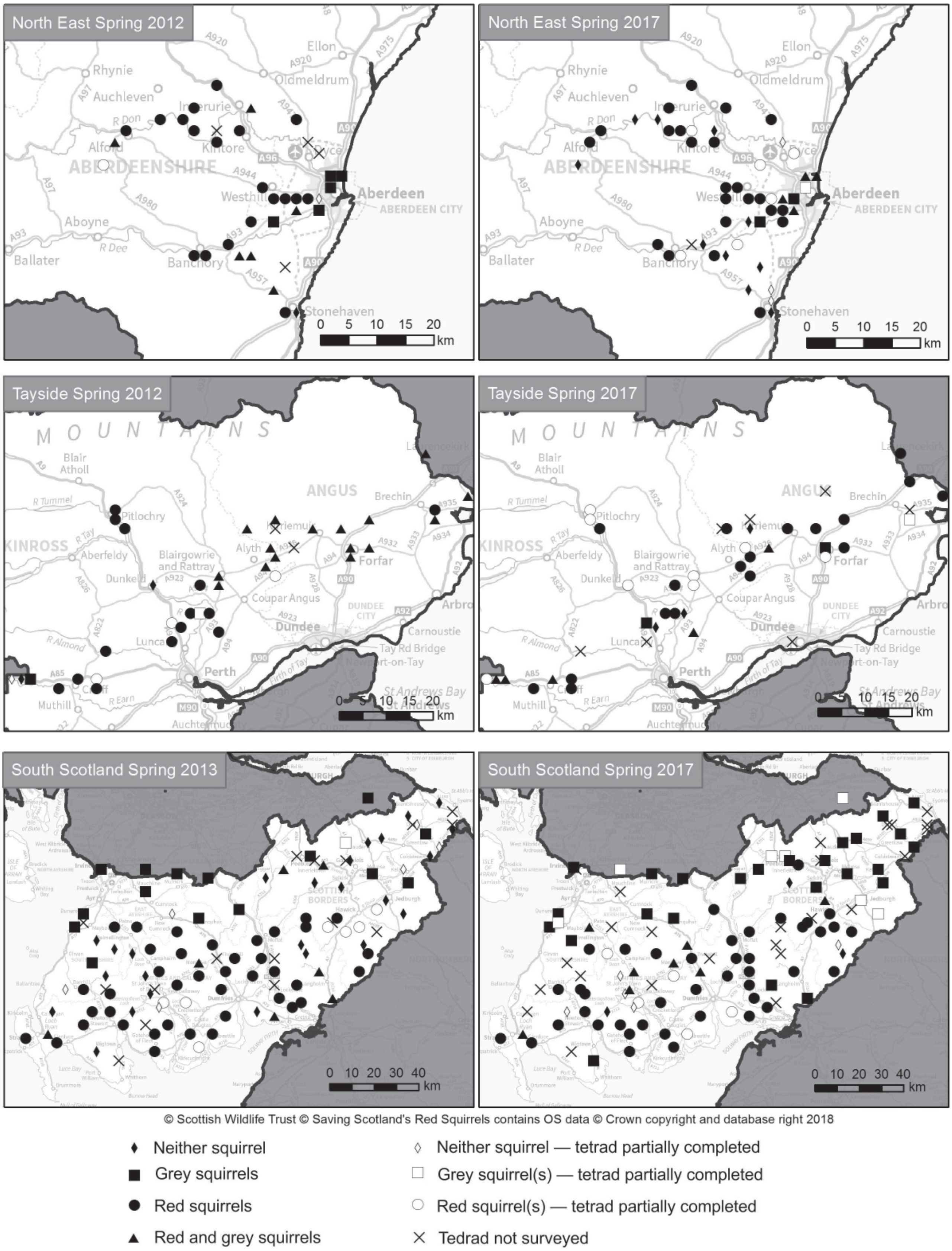

Fig. 3 Results of spring survey tetrads with both species, either species or none detected for a) North-east Scotland b) Tayside and (c) South Scotland for the years indicated (adapted from Shikorshidi & Tonkin 2018).

Table 1 Grey squirrel control achieved in all SSRS regions by control officers and land managers supported by SRDP funding.

	Grey Squirrel Officer Control		SRDP supported land manger control	
Year	Total no of trap days	No of greys captured	Total no of trap days	No of greys captured
2009	6,610	471	7,610	71
2010	18,615	1,637	41,365	510
2011	34,150	2,191	76,906	2,817
2012	40,783	2,797	237,738	3,630
2013	42,991	1,690	256,456	3,730
2014	32,889	1,819	222,031	2,713
2015	27,833	2,013	172,222	3,293
2016	9,776	490	89,316	1,029
Total	213,647	13,108	1,103,644	17,793
Average grey squirrel capture rate/100TN		6.14		1.61

was parameterised to assess the impact of current control measures; approximating the number of grey squirrels removed and adjusting the intensity of control in the model to mirror these levels by varying the area over which control was applied (White & Lurz, 2014). An alternative control scenario involving control along key dispersal routes was also assessed. The projections highlighted that current levels of control would not prevent the spread of the disease across Southern Scotland. Targeted control could help slow the spread of the virus but was unlikely to halt its spread in areas where grey squirrels are already established. However, the modelling also indicated that co-ordinated grey squirrel control should allow local red squirrel populations to persist and their density can recover after disease outbreaks in conifer dominated landscapes (White & Lurz, 2014; White, et al., 2016).

Annual (presence/absence) monitoring red and grey squirrels

The programme of annual presence/ absence monitoring indicates that red squirrel distributions have remained stable and that there have been some reductions in the range of grey squirrels in north Scotland and conversely some expansion in south Scotland (Fig. 3, Shirkhorshidi & Tonkin, 2018). The 2017 results of the tetrads in the north of Scotland as a whole (NE Scotland and Central Lowlands) show a significant change in favour of red squirrels when compared with 2012 (RI=-0.17, P=0.02). Contributing to this is a significant decrease in grey squirrel occupancy across the north and an increase in red squirrel distribution in the north-east, particularly in areas close to the City of Aberdeen; meanwhile red squirrel occupancy has been stable across the Highland Line (Shirkhorshidi & Tonkin, 2018). Although not significant, the overall changes in the south of Scotland have been in favour of grey squirrels (2013–2017 RI = 0.19). This reflects an increase in grey squirrel occupancy (largely outside SSRS areas of operation), whilst red squirrel occupancy appears to have been maintained (Shirkhorshidi & Tonkin, 2018). Whilst noting that squirrel populations experience fluctuations between years relating to seed crops, we interpret the overall trends in occupancy as an indication that SSRS's actions are helping to meet the project aims.

Public sightings of squirrels

Although they do not represent a systematic sample, public sightings help to harness public support and provide an early warning of range expansion of both red and grey squirrels. Sightings are mapped on the SSRS website (SSRS, 2018). For example, public reports have helped illustrate where red squirrels have returned to areas where they had not been recorded in the last 20 or 30 years following grey squirrel control, such as Aberdeen city parks (SSRS, 2017).

There has also been some standardisation and analysis of public sightings of squirrels between 1991 and 2010 across different regions of the UK (Gurnell, et al., 2014a). The data suggest red squirrel occupancy is declining in all regions over this period (at different rates), with the exception of Central Lowlands (east) of Scotland which fluctuates showing little overall change. However, an upward turn in red squirrel occupancy in the last two or three years is noted across all regions, especially South Scotland. Gurnell, et al. (2014a) indicate it is too early to speculate if the apparent upturn in the fortunes of red squirrels is as a result of grey squirrel control or other factors.

DISCUSSION AND FURTHER DEVELOPMENTS

We have described the evidence showing that sustained grey squirrel trapping can reduce grey squirrel abundance and occupancy at a landscape scale (Tayside and NE Scotland). Trapping data from Wales around the same time, indicates that sustained trapping can bring about reductions in grey squirrel populations at a landscape scale (Schuchert, et al., 2014). However, it has also been demonstrated that recolonisation can occur following intensive grey squirrel control after between one and three months (Lawton & Rochford, 2007; Schuchert, et al., 2014). Hence, SSRS are involved in further work to better quantify the level of control that might need to be sustained and, in particular, to put in place more sustainable delivery models.

The collaboration with researchers that started in South Scotland is now focussed on addressing the question of how much control may be required along the Highland Line to prevent grey squirrels from extending their range to the north. A spatially explicit population model (White, et al., 2017) has examined the impact of three levels of trapping intensity on grey squirrel populations in the Central Lowlands. Projections include the presence of squirrelpox virus (as a worst-case scenario) even though it has not yet been detected in this region. The potential density of red and grey squirrels in each 1 km × 1 km patch is derived based on average squirrel densities for the mixture of habitats encountered. Control is applied in targeted zones (typically 10 km ×10 km) and, mirroring trapping practice, can occur from 1 April to 30 September

(183 days), which is split into three 61 day (2 month) control periods. Trapping is applied in a responsive way to grid squares in which grey squirrels are present and in grid squares in a 2 km buffer zone in each of the three control periods. The model was run for three levels of trap intensity (TD = 0.3 – low; 0.5 – medium; 0.75 – high). This equates to 0.3 × 183 = 55 trap days per year (in a 1 km ×1 km grid square) in the low intensity scenario, 92 trap days per year (medium) and 137 trap days per year per (high), respectively.

In the low intensity scenario, trapping represents a kind of harvesting; there are abundant greys to catch and greys persist indefinitely. At medium intensity, control appears to be largely effective at preventing the northwards spread along the Highland Line. The model also highlights key dispersal routes where high intensity trapping is likely to be required (White, et al., 2017). Taking the Tayside Demonstration area (a key dispersal route) as an example the average, annual control effort predicted to be required (regions 7, 8, 9 in White, et al., 2017) is c. 18,000 trap days under the high intensity scenario. Given the modelled control area includes a slightly larger area than the Tayside demonstration area in order to prevent recolonisation, the levels of control suggested by the model are of the same order as control on the ground between 2012 and 2014 (Table 2) suggesting that this level of control effort needs to be maintained (White, et al., 2017).

Having successfully reduced the range, abundance and occupancy of grey squirrel populations in NE Scotland (Tonkin, et al., 2013; Shirkhorshidi & Tonkin, 2018), eradication of this isolated population now seems like a realistic prospect. However, some of the locations remaining untrapped are more challenging (smaller, fragmented and increasingly urban habitats). In 2014, SSRS set up an additional layer of monitoring to establish grey squirrel occupancy across the entire wooded network in the region. Feeder-box squirrel hair traps (n=223) have been distributed through all the suitable grey squirrel habitat patches in urban Aberdeen and the surrounding area. These data will allow analysis of grey squirrel occupancy (MacKenzie, et al., 2006) and better projections for the time and effort required to eradicate this isolated population. The monitoring will be complemented by rapid-response grey squirrel control.

SSRS's approach in South Scotland has adapted following the continued spread of squirrelpox virus and the model outcomes reported in White & Lurz (2014). The modelling indicates that co-ordinated grey squirrel control can help to protect red squirrel populations from the threat of squirrelpox virus in conifer dominated landscapes, where red squirrels typically occur at low densities, but importantly higher than those of grey squirrels. SSRS's control efforts have now shifted from the 'frontline' of squirrelpox detection to protecting identified priority areas for red squirrel conservation in South Scotland.

Table 2 Combined grey squirrel control effort (annual trap days) achieved by control officers and landowners supported by SRDP funding in the four demonstration areas 2009–2016.

	NE Scotland	Central Scotland		South Scotland
		Tayside	Argyll & Trossachs	
2009	2,465	NA	NA	4,987
2010	5,946	3,389	48	14,678
2011	7,878	8,201	360	17,912
2012	10,554	14,677	6,803	16,934
2013	6,178	16,158	6,721	20,079
2014	6,614	15,004	6,482	15,206
2015	7,500	9,780	6,590	5,114
2016	4,840	7,990	6,294	1,654*

*Landowner data not available

Sustaining the action

SSRS was initially funded for three years and eight months (2008–12), which was then extended by a further four years. The lead partner, SWT, secured a mixture of public and charitable funds to meet a project budget of just over £3 million covering the period 2008–16. In 2016, SWT secured a Heritage Lottery Fund Award of £2.46 million for the next 5 years (until 2022) towards a total project cost of £4.4 million. Hence, the costs have been roughly £0.5 million per annum to date. The piecemeal nature of project funding creates a challenge for sustaining co-ordinated grey squirrel control. Under the new SSRS phase, costs are anticipated to rise to c. £0.88 million per annum reflecting the additional activities aimed at ensuring the long-term sustainability of the control network and with a view to substantially reducing costs thereafter.

By September 2013, 197 landowners were in receipt of five-year SRDP funding at a cost of £4.5 million over the five-year period covered by the contracts. Although trapping by control officers is on average nearly four times more efficient (more captures per 100 trap days, Table 1) than SRDP-supported landowner grey squirrel control, landowner control provides five times more trapping effort than is provided by control officers (Table 1). Hence, being able to access public funding support has been hugely important. However, public funding is not without its challenges including: ease of access to the scheme for applicants; and ensuring trapping data are available to SSRS. Added to this there are uncertainties about the future of support upon leaving the European Union.

Quantifying the control effort needed to deliver SSRS's objectives has been challenging. However, SSRS's monitoring and associated modelling has supported that the levels achieved seem 'about right'. However, this equates to a substantial network of grey squirrel control that needs to be sustained.

Reflecting the successes to date and the challenges ahead, the next five-year phase is called Saving Scotland's Red Squirrels – Developing Community Action. This project's actions are geared towards long-term sustainability and how SSRS's work can be embedded in routine land management and community action, with a move away from reliance on project staff. Project funding at this level of investment is increasingly hard to find, hence there is an expectation that red squirrel conservation will increasingly rely on public delivery.

SSRS – Developing Community Action now aims to eradicate grey squirrels from NE Scotland within 10 years. In South Scotland the aims have been refined and focus on building the skills and resources available to local people and land managers working to control grey squirrels in identified priority areas. As part of this, a Community Hub information management system is being developed for staff and volunteers, which will better capture and integrate data from all sources and will allow improved feedback. For each priority area, an annual trapping programme is being developed that is capable of continuing to protect the red squirrel population. As yet it remains to be determined if the necessary levels of control can be sustained by these means. However, there is a shift in the focus of SSRS work from demonstrating the efficacy of control on to how can it be delivered.

Largely based on the evaluation of work co-ordinated by SSRS, the national policy position now recognises coordinated grey squirrel control as an integral part of the long-term approach to achieving the strategy aims (Scottish Squirrel Group, 2015).

The challenge of protecting Scotland's red squirrels remains significant given the scale and the ongoing nature of the work. However, the prospects for alternative/ or complementary approaches are also improving. Immuno-contraceptives and squirrelpox vaccines are actively being explored with the support of parallel initiatives under the 'UK Squirrel Accord' but are likely to be some years in development. New research into the role of pine marten on the dynamics of red and grey squirrels also offers promising insights in that as pine marten populations recover their range and densities, grey squirrel populations appear to be suppressed in the presence of this novel (to them) native predator, thereby reducing the levels of management control required to promote red squirrel persistence (Sheehy, et al., 2018).

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REFERENCES

Ashbrook Research Consultancy Ltd (2010). *Assessment of Public Attitudes to Grey Squirrel Control in Aberdeen.* Scottish Natural Heritage, Commissioned Report No. 384. Battleby, Perth: Scottish Natural Heritage.

Bertolino, S. and Genovesi, P. (2003). 'Spread and attempted eradication of the grey squirrel (*Sciurus carolinensis*) in Italy, and consequences for the red squirrel (*Sciurus vulgaris*) in Eurasia'. *Biological Conservation* 109(3): 351–358.

Bryce, J., Johnson, P.J. and Macdonald, D.W. (2001). 'Can niche use in red and grey squirrels offer clues for their apparent coexistence?' *Journal of Applied Ecology* 39: 875–887.

Central Science Laboratory (2009). *Review of Methods of Humane Destruction of Grey Squirrels (*Sciurus carolinensis*).* Scottish Natural Heritage Commissioned Report No, 317. Battleby, Perth: Scottish Natural Heritage.

Gurnell, J. (1987). *The Natural History of Squirrels*. London: Christopher Helm.

Gurnell, J. and Pepper, H. (1993). 'A critical look at conserving the British red squirrel (*Sciurus vulgaris*)'. *Mammal Review* 23: 125–136.

Gurnell, J., Lurz, P.W.W. and Bertolino, W. (2014a). 'The changing patterns in the distribution of red and grey squirrels in the north of England and Scotland between 1991 and 2010 based on volunteer surveys'. *Hystrix, the Italian Journal of Mammalogy* 25 (2): 83–89.

Gurnell, J., Lurz, P.W.W. and Wauters, L.A. (2014b). 'Years of Interactions and Conflict in Europe: Competition Between Eurasian Red Squirrels and North American Grey Squirrels'. In: C.M. Shuttleworth, P.W.W. Lurz and M.W. Hayward (eds.) *Red Squirrels: Ecology Conservation and Management in Europe,* pp. 19–37. Llangefni: European Squirrel Initiative.

Lawton, C. and Rochford, J. (2007). 'The recovery of grey squirrel (*Sciurus carolinensis*) populations after intensive control programmes'. *Biology and Environment: Proceedings of the Royal Irish Academy* Vol. 107B, No. 1: 19–29.

Lloyd, H.G. (1983). 'Past and present distributions of red and Grey squirrels'. *Mammal Review* 13: 69–80.

MacKenzie, D.I., Nichols, J.D., Royle, A.J., Pollock, K.H., Bailey, L.L. and Hines, J.E (2006). *Occupancy Estimation and Modlling: Inferring Patterns and Dynamics of Species occurrence.* Burlington, Massachusetts: Elsevier Academic Press.

McInnes, C.J., Coulter, L., Dagleish, M.P., Fiegna, C., Gilray, J., Willoughby, K., Cole, M., Milne, E., Meredith, A., Everest, DJ., and MacMaster, A. (2009). 'First cases of squirrelpox in red squirrels (*Sciurus vulgaris*) in Scotland'. *Veterinary Record* 164: 528–531.

Middleton, A.D. (1930). 'The ecology of the American grey squirrel (*Sciurus carolinensis* Gmelin) in the British Isles'. *Proceedings of the Zoological Society of London* 100: 809–843.

Middleton, A.D. (1931). *The Grey Squirrel*. London: Sidgwick & Jackson.

Pepper, H., and Patterson, G. (1998). *Red Squirrel Conservation*. Forestry Commission Practice Note 5. Edinburgh: Forestry Commission. <www.forestry.gov.uk/pdf/fcpn5.pdf/$file/fcpn5.pdf>.

Reynolds, J. (1998). 'Details of the geographic replacement of the red squirrel (*Sciurus vulgaris*) by the grey squirrel (*Sciurus carolinensis*) in Eastern England'. *Journal of Animal Ecology* 54: 149–162.

Rushton, S.P., Lurz, P.W.W. Gurnell, J., Nettleton, P., Bruemmer, C. and Shirley, M.D.F. (2006). 'Disease threats posed by alien species: the role of a poxvirus in the decline of the native red squirrel in Britain'. *Epidemiology and Infection* 134: 521–533.

Sainsbury, A. W., Nettleton, P., Gilray, J. and Gurnell, J. (2000). 'Grey squirrels have a high seroprevalence to a parapoxvirus associated with deaths in red squirrels'. *Animal Conservation* 3: 229–233.

Schuchert, P., Shuttleworth, C.M., McInnes, C.J., Everest, D.J. and Rushton, S.P. (2014). 'Landscape scale impacts of culling upon a European grey squirrel population: Can trapping reduce population size and decrease the threat of squirrelpox virus infection for the native red squirrel?'. *Biological Invasions* 16: 2381–2391.

Scottish Government Statistics (2014) 'Number and value of Rural Development Contract: Rural Priorities Options funding under the 2007–13 SRDP'. [*Statistics web resource*]. <www.gov.scot/Resource/0044/00449780.pdf>. Accessed 20 February 2018.

Scottish Natural Heritage (2010). *Protecting Scotland's Red Squirrels. The Consultation Response.* Battleby, Perth: Scottish Natural Heritage.

Scottish Squirrel Group (2004). *Scottish Strategy for Red Squirrel Conservation.* Battleby, Perth: Scottish Natural Heritage.

Scottish Squirrel Group (2015). *Scottish Strategy for Red Squirrel Conservation.* Battleby, Perth: Scottish Natural Heritage. https://www.nature.scot/scottish-strategy-red-squirrel-conservation-june-2015.

Sheail, J. (1999). 'The grey squirrel (*Sciurus carolinensis*) – A UK historical perspective on a vertebrate pest species'. *Journal of Environmental Management* 55: 145–156.

Sheehy, E., Sutherland, C., O'Reilly, C. and Lambin, X. (2018). 'The enemy of my enemy is my friend: native pine marten recovery reverses the decline of the red squirrel by suppressing grey squirrel populations'. *Proceedings of the Royal Society B* 285: 20172603. <://dx.doi.org/10.1098/rspb.2017.2603>. Accessed 9 April 2018.

Shorten, M. (1962). 'Squirrels, their biology and control'. *Ministry of Agriculture Fisheries and Food Bulletin* 184: 1–44.

Shirkhorshidi, M. and Tonkin, M. (2018) *Results of Spring 2017 Squirrel Surveys*. Saving Scotland's Red Squirrels report. Edinburgh: Scottish Wildlife Trust. <www.scottishsquirrels.org.uk/docs/087_506__ssrssquirrelsurvey2017_1517589214.pdf>.

Signorile, A.L. (2013). 'Genetic Determinants of the Expansion of Eastern Grey Squirrel Populations Across Europe'. PhD thesis. London: Imperial College London and Institute of Zoology, Zoological Society.

Skelcher, G. (1997). 'The Ecological Replacement of Red by Grey Squirrels'. In: J. Gurnell and P.W.W. Lurz (eds.) *The conservation of Red Squirrels,* Sciurus vulgaris *L*., pp. 67–78. London: Peoples Trust for Endangered Species.

SSRS (2017). 'New sightings signal red squirrels' return to Aberdeen city parks and gardens'. [website] (published 15/11/2017). <www.scottishsquirrels.org.uk/news/article/new-sightings-signal-red-squirrelsrsquo-return-to-aberdeen-city-parks-and-gardens/>.

SSRS (2018). 'Map of squirrel sightings' [website] <www.scottishsquirrels.org.uk/squirrel-sightings/squirrel-sighting-map.> Accessed 20 February 2018.

Staines, B. (1986). 'The spread of grey squirrels (*Sciurus carolinensis*) into north-east Scotland'. *Scottish Forestry* 40: 190–196.

Thomas, K., Tompkins, D.M., Sainsbury, A.W. Wood, A.R., Dalziel, R., Nettleton, P.F and McInnes, C.J. (2003). 'A novel poxvirus lethal to red squirrels (*Sciurus vulgaris*)'. *Journal of General Virology* 84: 3337–3341.

Tonkin, M., Balmforth, L. and Ramoo, K. (2013). *Evaluation of Grey Squirrel Control in the Saving Scotland's Red Squirrels Partnership Project (Phase Two) – Second Report 2009–2013.* Unpublished Report to SSRS Steering Group Partners. Edinburgh: Scottish Wildlife Trust.

Tonkin, M., Garritt, J., Bryce, J. and Cole, M. (2015). 'Species Management in Scotland 2007–2012: Red and Grey Squirrels'. In: M.J. Gaywood, P.J. Boon, D.B.A. Thompson and I.M. Strachan (eds.) *The Species Action Framework Handbook,* pp. 82–93. Battleby, Perth: Scottish Natural Heritage.

Usher, M.B., Crawford, T.J. and Banwell, J.L. (1992). 'An American invasion of Great Britain: The case of the native and alien squirrel (*Sciurus*) species'. *Conservation Biology* 6: 108–115.

Wauters, L.A., Gurnell, J., Martinoli, A. and Tosi, G. (2002). 'Interspecific competition between native Eurasian red squirrels and alien grey squirrels: does resource partitioning occur?'. *Behavioural Ecology and Sociobiology* 52: 332–341.

White, A. and Lurz, P.W.W. (2014). *A Modelling Assessment of Control Strategies to Prevent/Reduce Squirrelpox Spread.* Scottish Natural Heritage Commissioned Report 627, Battleby, Perth: Scottish Natural Heritage.

White, A., Lurz, P. W. W., Bryce, J., Tonkin, M., Ramoo, K., Bamforth, L., Jarrott, A. and Boots, M. (2016). 'Modelling disease spread in real landscapes: Squirrelpox spread in Southern Scotland as a case study'. *Hystrix, the Italian Journal of Mammalogy* 27 (1). DOI: 10.4404/hystrix-27.1-11657.

White, A, Lurz, P.W.W. and Boots, M. (2017). *Grey Squirrel Control Along the Highland Line: A Model Analysis.* NERC Innovation grant report (Ref: NE/M021319/1). <www.macs.hw.ac.uk/~awhite/SWT_SNH_HighlandLine_FinalReport.pdf>.

Testing auto-dispensing lure pumps for incursion control of rats with reduced effort on a small, re-invadable island in New Zealand

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Abstract In locations with a high potential for re-invasion, such as inshore islands, sustained control of invasive species is as important as the initial knock-down for the long-term recovery of native populations. However, ongoing trap maintenance and lure replenishment are barriers to minimising the time and financial costs of long-term suppression, even when automatic traps are used. Control of invasive mammal species is a high priority for the more than 200 islands within Rakiura National Park in southern New Zealand, many of which support nationally and internationally threatened endemic species and ecosystems. We previously used automatic, toxicant-free traps to control rats on Native Island, a 62 ha inshore island within the National Park, where tracking indices were 73% in mid-2013. After 18 months, tracking indices remained below 10%, and site visits were reduced to twice per year, following introduction of novel auto-lure pumps. Tracking indices for rats remained low after six months, then increased to 37% in May 2017. That increase, as well as small fluctuations in measured activity levels throughout the study, could indicate continued incursion from the mainland, highlighting the importance of continued suppression. Additional work is needed to determine the limitations of the automatic lure dispensers and optimise their use for long-term suppression of pest mammals in ecosystems that are highly vulnerable to re-invasion.

Keywords: conservation, invasion biology, invasive mammals, island biosecurity, Norway rat (*Rattus norvegicus*), Pacific rat (*Rattus exulans*), ship rat (*Rattus rattus*)

INTRODUCTION

Introduced mammals are one of the most significant threats to island ecosystems (Towns, et al., 2006; Bellingham, et al., 2010; Harper & Bunbury, 2015). In particular, rats (*Rattus* spp.) and other rodents have become major predators of endemic island species, causing several local extinctions (Courchamp, et al., 2003; Towns, et al., 2006; Bellingham, et al., 2010). Thus, they are a main target of eradication operations (Howald, et al., 2007; Glen, et al., 2013; Holmes, et al., 2015). However, while numerous offshore rat eradications have been undertaken successfully since the 1980s, eradication is more difficult in locations that are close enough to a non-controlled pest population to facilitate rapid, and inevitable, re-invasion (Russell, et al., 2008; Nathan, et al., 2015). At highly re-invadable sites, such as near-shore islands, a single operation can theoretically eliminate a population of invasive rats. However, that 'eradication' is only temporary. Sustained control is required in order to prevent re-establishment (Simberloff, 2011), which can occur rapidly and with only a few invaders (Russell, et al., 2008; Nathan, et al., 2015).

Most successful eradication operations on New Zealand islands – both of rats and of other pest mammals – have been undertaken using site-wide toxicant applications (Blackie, et al., 2013; Keitt, et al., 2015). However, toxicant-based methods are not as effective for sustained control in highly re-invadable sites as they are on relatively isolated, offshore islands. Importantly, a re-invading population of mammals has to achieve a minimum density in order for repeated toxicant use to be considered a cost-effective means of control (Warburton & Thomson, 2002), but that density threshold is higher than the maximum density under which many native species can successfully recover (Gillies, et al., 2003; Norbury, et al., 2015). Thus, conservation-motivated, long-term mammal suppression in re-invadable sites requires the availability of sustained-use, cost-effective methods. Throughout this paper, we use the terms 'suppression' and 'maintenance control' interchangeably to refer to any control method used to prevent the re-establishment of a population of pest mammals in an island due to incursion. However, the same principles can be applied within any controlled area that is at risk of being invaded, or re-invaded, from an adjacent, un-controlled population.

Unlike mammal control operations that rely on site-wide application of toxicants, traps can be left in situ and used for incursion control. However, current best-practice methods of trapping require continual re-baiting and, if a trap is triggered, re-setting of the trapping mechanism to remain effective (DOC, 2006). In addition, traps may be less effective at controlling low-density populations than they are at eradication of established, high-density populations (Thorsen, et al., 2000; Chappell, 2004). As a result, effective island biosecurity still requires regular surveillance and the availability of funding to undertake contingency response in the event of an incursion (Russell, et al., 2008). A relatively new technique for long-term control of invasive mammal populations is the use of automatic, or self-resetting, trapping and toxicant application mechanisms (reviewed in Campbell, et al., 2015). Like single-use trapping methods, self-resetting mechanisms – both traps and toxicant-delivery devices – can be designed with relatively high specificity, reducing the rate of non-target kills, relative to that realised following site-wide toxicant applications (Campbell, et al., 2015). Unlike single-use traps, automatic mechanisms can remove multiple pests before requiring maintenance and/or re-baiting (Blackie, et al., 2011; Blackie, et al., 2013; Murphy, et al., 2014; Carter, et al., 2016).

Automatic toxicant-delivery devices have been designed for stoats (*Mustela ermina*), possums (*Trichosurus vulpecula*) (Blackie, et al., 2016) and rats (Blackie, et al., 2013; Murphy, et al., 2014). Automatic, toxicant-free traps and corresponding long-life lures have been developed by Goodnature® Ltd for possums, rats, and stoats (Carter, et al., 2016; Carter & Peters, 2016), with the advantage that devices that do not rely on poisons may be more acceptable for control of invasive mammals in locations that support populations of native mammals (Campbell, et al., 2015). Self-resetting traps have been used to control sympatric populations of Norway rats (*Rattus norvegicus*), ship rats (*R. rattus*), and Australian brushtail possums

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on a single, near-shore island (Carter, et al., 2016) and to control ship rats and mice (*Mus musculus*) within an unprotected mainland site (Carter & Peters, 2016) in New Zealand. One pest control operation in Hawaii also found that automatic traps were more beneficial for decreasing predation of native species by rats than single-action snap-traps (Franklin, 2013).

The long-term financial costs of using automatic traps for control of invasive mammal populations are comparable to those of using basic Victor® snap-traps, especially when work is undertaken primarily by contractors, and slightly lower than the costs of using DOC-200 traps, heavy-duty, single-action tunnel traps commonly used for maintenance control (Carter, et al., 2016; Carter & Peters, 2016). The use of self-resetting traps greatly reduces the frequency at which site visits must be undertaken, relative to traditional methods of trapping that require regular rebaiting and resetting to maintain effectiveness (e.g., Franklin, 2013). However, the rate at which even long-life lures must be replenished in self-resetting traps – approximately monthly – is still higher than the rate at which pests are killed, following initial suppression of the population (Carter, et al., 2016). As a result, the costs of long-term suppression of pest mammals – in terms of both equipment and person-hours – are increased significantly by the investment in on-the-ground trap maintenance, even when self-resetting traps are used (Franklin, 2013; Glen, et al., 2013; Carter, et al., 2016; Carter & Peters, 2016).

The continued effectiveness of self-resetting traps relies largely on maintaining attractiveness of the highly viscous lure, which is contained within a plastic bottle housed inside the trap. When a targeted mammal population is relatively dense and the lure is consumed regularly, the force of gravity is sufficient to keep 'fresh' lure available. Once a population of invasive mammals has been knocked down, human intervention is required to ensure that uneaten lure does not become mouldy and unpalatable after being exposed to air. Thus, the mechanism of lure delivery itself remains a barrier to minimising costs of maintenance control. Here, we tested the use of auto-dispensing lure pumps for retaining lure freshness and maintaining low levels of rats on a previously-controlled inshore island, while significantly reducing the person-hours required for undertaking site visits.

MATERIALS AND METHODS

In November 2013, we installed 143 CO_2-powered, automatic rat traps (A24; Goodnature® Ltd, Wellington, New Zealand; https://www.goodnature.co.nz) on a 100 m × 50 m grid on Native Island (46°54'54″ E 168°09'25″ S) (Carter, et al., 2016), a mostly forested, 62 ha Scenic Reserve within Rakiura National Park in southern New Zealand (DOC, 2012). Because Native Island sits approximately 100 m from the coast of the main island of Stewart Island (also known as Rakiura), incursion by multiple species of pest mammals from the mainland following a control operation is inevitable (Atkinson, 1986). The presence of Norway rats, ship rats and brushtail possums has been confirmed on Native Island (DOC, 2012), and all three species were observed during establishment of the trapping network. In addition, Pacific rats (kiore, *Rattus exulans*) are present on the nearby mainland and may pose an additional incursion risk.

Each trap was initially baited with a bottle of non-toxic, peanut-based lure and checked approximately monthly, with lure bottles and CO_2 cartridges replenished every six months (Carter, et al., 2016). Due to lack of resources, the traps were not maintained for the eight months between September 2015 and May 2016. In May 2016, we replaced all CO_2 cartridges and replaced the standard lure bottles with novel auto-lure pumps. The auto-lure pump is a soft-sided lure bottle that uses hydrogen gas expansion to deliver 55 g of non-toxic lure over a period of six months (Fig. 1). The CO_2 cartridges and auto-lure pumps were replaced every six months.

During the initial control operation only, we used tracking tunnels (Pest Control Research [PCR] Ltd., Christchurch, New Zealand) with inked tracking cards (Black Trakka®, Gotcha Traps, Auckland, New Zealand) to monitor mammal activity within the trapping grid on Native Island and at a control site, located 3.5 km away on Stewart Island (Gillies & Williams, 2013). We estimated rat activity using tracking indices, with detection corrected for interference with the tracking cards by possums, where required (Gillies & Williams, 2013). Tracking tunnels were installed at 50 m intervals on Native Island in six lines of five tunnels each, and at the control site in three lines of five tunnels and two lines of ten tunnels. During each monitoring period, tracking tunnels were baited with peanut butter and tracking cards retrieved after 24 hours (Carter, et al., 2016). Following installation of the auto-lure pumps, rat activity was monitored at two subsequent intervals of approximately six months, at the Native Island site only.

RESULTS

During the initial control operation, tracking indices for rats on Native Island decreased from 73% to 7% within nine months of initiation of trapping and remained perpetually at or below 10% during the monitoring

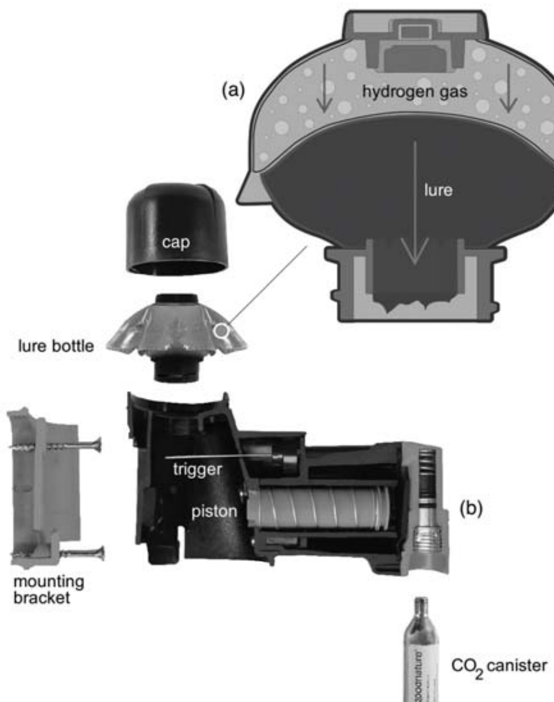

Fig. 1 Diagram of an (a) auto-lure pump and (b) deconstructed interior of an A24 self-resetting rat trap. Activation of the trigger by a rat as it accesses the lure causes rapid deployment of the CO2-powered piston, which strikes the skull and results in near-instantaneous death. The trap automatically resets after each strike, up to 24 times. Gradual expansion of hydrogen gas inside the soft-sided, auto-lure pump slowly delivers lure through the bottle opening over a period of six months. Image courtesy of Goodnature® Ltd (Wellington, New Zealand).

period, while tracking indices at the control site remained comparatively high (Fig. 2; see Carter, et al., 2016 for complete results). On the first monitoring visit following installation of the auto-lure pumps, tracking indices on Native Island were 7% but increased to 37% during the most recent site visit in May 2017 (Fig. 2). Rat activity was not monitored at the control site after the initial trapping operation. However, tracking indices within a separate, un-trapped area on Stewart Island were 40% in March 2017 (SIRCET, 2017). Between the first and second monitoring visits, air temperatures were between -1.5°C and 0.5°C of monthly regional (Southland) averages, varying between 8°C and14°C during the study months (Macara, 2013), and rainfall levels were at or below normal levels (Fig. 2).

DISCUSSION

This project was the first *in situ* test of auto-dispensing lure bottle technology, following a previous knockdown. One of the primary motivations for developing time-saving technologies for invasive mammal control – lack of sufficient available person-hours for maintaining traps and monitoring for incursions – was both the impetus for and the main limitation of this study. Because rat activity levels were not monitored for the year prior to installation of the auto-lure pumps, nor when they were installed, we cannot say definitively that they were as effective as standard lure bottles at maintaining low levels of rats. That is, the activity levels recorded in October 2016 could be indicative of no incursions, with switching of standard bait bottles for auto-lure pumps having no effect on the consistently low activity levels observed since at least August 2014. However, given the proximity of the study site to uncontrolled populations of multiple rat species, as well as fluctuating activity levels throughout the original control operation and slight increase observed in May 2015, that activity levels were still below 10% a year later is unlikely. More likely is that rat activity levels increased to some extent prior to installation of the auto-lure pumps and that the pumps effectively reduced activity levels during the first five months of their operation.

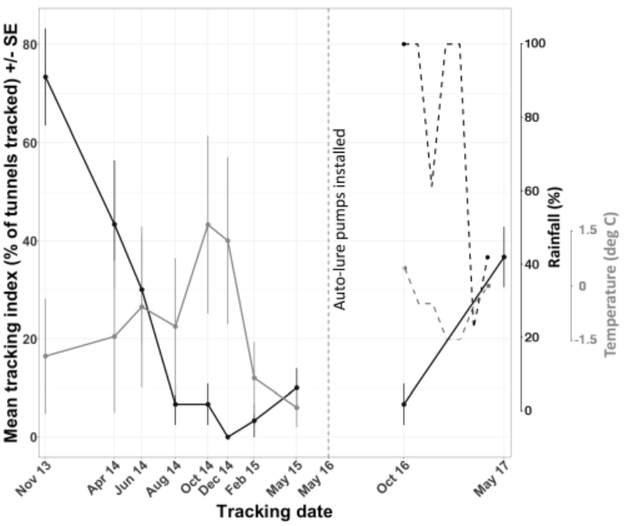

Fig. 2 Summary of monitoring data from tracking tunnels on Native Island (black) and the control site on Stewart Island (grey), with the introduction of auto-lure pumps indicated by the dashed vertical line. Except for the period from May 2015 – May 2016, the spacing of x-axis labels is proportional to the amount of time between monitoring dates. The percentage of tracking cards with interference by possums was high at the control site on Stewart Island throughout the initial trapping operation, so true activity of rats may be higher than indicated by the plot, especially in February and May of 2015. No data are available for the period May 2015 – October 2016, and no monitoring was undertaken at the control site after May 2015. Climate data are shown with dashed lines. The rainfall axis shows the approximate percentage of local rainfall in each month, relative to 'normal' conditions (i.e., a value of 100% is equal to normal). The temperature axis shows the deviation of local air temperatures from average conditions, with a value of 0 equal to the respective monthly mean. Weather information was obtained from NIWA 'Current climate' monthly summaries (https://www.niwa.co.nz/climate/nzcu/). Plot adapted from Carter, et al. (2016).

During the second, but not the first, monitoring visit to Native Island, the lure was noticeably mouldy and may have been less attractive to rats. Mould growth may be related to environmental conditions at the study site, which would suggest that the rate of gas expansion inside the auto-lure pump may be insufficient to keep the lure fresh in certain conditions. Climate has been implicated in the failure of mammal control operations across methods, with stationary bait stations being most similar to trapping. Bait station-based eradication failures have been associated with higher mean annual temperatures and increased variation in inter-annual precipitation in tropical locations, which become more important with increasing island size (Holmes, et al., 2015). High temperatures, in particular, are a significant predictor of failure across toxicant-based methods of rat eradication (Holmes, et al., 2015).

The importance of climate to the success of mammal control has been examined primarily in relation to the timing of toxicant application, particularly in the tropics, where more consistent food availability increases the difficulty of targeting rodents using attractant-based toxicants (Holmes, et al., 2015; Russell & Holmes, 2015). Air temperatures at our study location did not vary much from average monthly conditions, and more months were relatively 'dry' than 'wet,' compared with regional norms (Fig. 2). However, further research should be undertaken to determine whether abiotic environmental conditions constrain the efficacy of auto-lure pumps. If so, either (1) increasing the rate of gas expansion inside the auto-lure bottle or (2) increasing the rate of site visits during particular seasons or in climates normally conducive to mould growth may be required.

Assuming the number of successful control operations in incursion-vulnerable sites increases, so too will the costs of controlling invasive mammals. In highly re-invadable sites, true eradication is an impractical aim (Simberloff, 2011). Indeed, if mammal densities can be maintained at levels low enough to facilitate the recovery of native populations, then eradication becomes less immediately imperative. Thus, cost-effective suppression of pest mammals is a realistic goal for conservation of endemic island biodiversity. Technologies that minimise the time and financial investments required for long-term control will be key to maximising the area within which populations of invasive mammals can be controlled. More work is needed to optimise the use of auto-lure pumps and quantify their limitations. However, effective automatic lure delivery devices would be a transformative addition to the pest-control toolbox and should continue to be rigorously developed and tested.

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REFERENCES

Atkinson, I.A.E. (1986). 'Rodents on New Zealand's Offshore Islands: Distribution, Effects and Precautions Against Further Spread'. In: A.E. Wright and R.E. Beever (eds.) *The Offshore Islands of Northern New Zealand*, pp.13–40. Information Series 14. Wellington: New Zealand Department of Lands and Survey.

Bellingham, P.J., Towns, D.R., Cameron, E.K., Davis, J.J., Wardle, D.A., Wilmshurt, J.M. and Mulder, C.P.H. (2010). 'New Zealand island restoration: Seabirds, predators, and the importance of history'. *New Zealand Journal of Ecology* 34: 115–136.

Blackie, H.M., MacKay, J.W.B., Allen, W.J., Smith, D.H.V., Barrett, B., Whyte, B.I., Murphy, E.C., Ross, J., Shapiro, L., Ogilvie, S., Sam, S., MacMorran, D., Inder, S. and Eason, C.T. (2013). 'Innovative developments for long-term mammalian pest control'. *Pest Management Science* 70: 345–351.

Blackie, H., MacKay, J., Barrett, B., Inder, S., MacMorran, D., Bothwell, J., Clout, M. and Eason, C. (2016). A novel device for controlling brushtail possums (*Trichosurus vulpecula*). *New Zealand Journal of Ecology* 40: 60–64.

Blackie, H., Woodhead, I., Diegel, H., MacMorran, D., Shapiro, L., Murphy, E. and Eason, C. (2011). 'Integrating ecology and technology to create innovative pest control devices'. In: J. Jacob and A. Esther (eds.) *Proceedings of the 8th European Vertebrate Pest Management Conference*, pp. 152–153. Berlin: Julius Kühn-Institut.

Campbell, K.J., Beek, J., Eason, C.T., Glen, A.S., Godwin, J., Gould, F., Holmes, N.D., Howald, G.R., Madden, F.M., Ponder, J.B., Threadgill, D.W., Wegmann, A.S. and Baxter, G.S. (2015). 'The next generation of rodent eradications: innovative technologies and tools to improve species specificity and increase their feasibility on islands'. *Biological Conservation* 185: 47–58.

Carter, A.C., Barr, S., Bond, C., Paske, G., Peters, D. and van Dam, R. (2016), 'Controlling sympatric pest mammal populations in New Zealand with self-resetting toxicant-free traps: A promising tool for invasive species management'. *Biological Invasions* 18: 1723–1736.

Carter, A.C. and Peters, D. (2016). 'Self-resetting Traps Provide Sustained, Landscape-scale Control of a Rat Plague in New Zealand'. In: R. M. Timm and R. A. Baldwin (eds.) *Proceedings of the 27th Vertebrate Pest Conference*, pp. 308–313. Davis, CA: University of California.

Chappell, R. (2004). *Motutapere Island Rodent Invasion 2004*. Internal report, file ISL 0013. Coromandel, New Zealand: Department of Conservation.

Courchamp, F., Chapuis, J.-L. and Pascal, M. (2003). 'Mammal invaders on islands: impact, control and control impact'. *Biological Reviews* 78: 347–383.

DOC [Department of Conservation, New Zealand] (2006). *Island Biosecurity Best Practice Manual*. Wellington, New Zealand: Department of Conservation.

DOC [Department of Conservation] (2012). *Stewart Island / Rakiura Conservation Management Strategy and Rakiura National Park Management Plan 2011–2021*. Invercargill, New Zealand: Department of Conservation.

Franklin, K. (2013). *Informational Report on the Use of Goodnature A24 Rat Traps in Hawaii*. Unpublished report. Oahu, HI: Pacific Cooperative Studies Unit, Research Corp. of the University of Hawaii, Oahu Army Natural Resources Program.

Gillies, C.A., Leach, M.R., Coad, N.B., Theobald, S.W., Campbell, J., Herbert, T., Graham, P.J. and Pierce, R.J. (2003). 'Six years of intensive pest mammal control at Trounson Kauri Park, a Department of Conservation "mainland island", June 1996–July 2002'. *New Zealand Journal of Zoology* 30: 399–420.

Gillies, C.A. and Williams, D. (2013). *DOC tracking tunnel guide v2.5.2: using tracking tunnels to monitor rodents and mustelids*. Unpublished report. Hamilton, New Zealand: Science and Capability Group.

Glen, A.S., Atkinson, R., Campbell, K.J., Hagen, E., Holmes, N.D., Keitt, B.S., Parkes, J.P., Saunders, A., Sawyer, J. and Torres, H. (2013), 'Eradicating multiple invasive species on inhabited islands: The next big step in island restoration?'. *Biological Invasions* 15: 2589–2603.

Harper, G.A. and Bunbury, N. (2015). 'Invasive rats on tropical islands: Their population biology and impacts on native species'. *Global Ecology and Conservation* 3: 607–627.

Holmes, N.D., Griffiths, R., Pott, M., Alifano, A., Will, D., Wegmann, A.S. and Russell, J.C. (2015). 'Factors associated with rodent eradication failure'. *Biological Conservation* 185: 8–16.

Howald, G., Donlan, C.J., Galván, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

Macara, G.R. (2013). 'The Climate and Weather of Southland'. *NIWA Science and Technology Series* 63. Wellington: National Institute of Water and Atmospeheric Research.

Murphy, E., Sjoberg, T., Barun, A., Aylett, P., MacMorran, D. and Eason, C. (2014). 'Development of Re-setting Toxin Delivery Devices and Long-life Lures for Rats'. In: R.M. Timm and J.M. O'Brien (eds.) *Proceedings of the 26th Vertebrate Pest Conference,* pp. 396–399. Davis: University of California.

Nathan, H.W., Clout, M.N., MacKay, J.W.B., Murphy, E.C. and Russell, J.C. (2015). 'Experimental island invasion of house mice'. *Population Ecology* 57(2): 363–371.

Norbury, G.L., Pech, R.P., Byrom, A.E. and Innes, J. (2015). 'Density impact functions for terrestrial vertebrate pests and indigenous biota: guidelines for conservation managers'. *Biological Conservation* 191: 409–420.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Russell, J.C., Towns, D.R., Beaven, B.M., MacKay, J.W.B. and Clout, M.N. (2008), 'Testing island biosecurity systems for invasive rats'. *Wildlife Research* 35: 215–221.

Simberloff, D. (2011). 'Today Tiritiri Matangi, tomorrow the World! Are we aiming too low in invasives control?' In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 4–12. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

SIRCET. (2017) *SIRCET Rat Monitoring Results, 25th May 2017*, Unpublished report. Stewart Island, NZ: Stewart Island / Rakiura Community and Environment Trust.

Thorsen, M., Shorten, R., Lucking, R. and Lucking, V. (2000). 'Norway rats (*Rattus norvegicus*) on Frégate Island, Seychelles: The invasion, subsequent eradication attempts and implications for the island's fauna'. *Biological Conservation* 96: 133–138.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?'. *Biological Invasions* 8: 863–891.

Warburton, B. and Thomson, C. (2002). 'Comparison of three methods for maintaining possums at low density'. *Science for Conservation.* 189. Wellington, New Zealand: Department of Conservation.

Survival analysis of two endemic lizard species before, during and after a rat eradication attempt on Desecheo Island, Puerto Rico

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Abstract Rodent eradications are a key island restoration activity to counteract extinction and endangerment to native species. Despite the widespread use of brodifacoum as a rodenticide for island restoration, there has been little examination of its potential negative effects on native reptiles. Here we examined the survival of two endemic insular lizard populations before, during and after a brodifacoum-based rodent eradication using a mark-recapture study. We found no evidence of an effect from baiting in *Anolis desechensis* and evidence of a change in recapture rates after baiting for *Ameiva desechensis*. Effects of baiting on survival rates were not measurable due to a small sample size. Results suggest that brodifacoum did not result in population-level impacts during the three-week study period after brodifacoum exposure. For invasive species eradications using toxicants, potential risks to non-target species should be assessed against the expected benefits to native biota from the removal of threats posed by invasive mammals. We recommend continued studies that directly examine non-target risk to native reptile populations derived from toxicant baiting programs, particularly on tropical islands that are home for high numbers of endemic reptiles.

Keywords: brodifacoum, mark-recapture, non-target species, reptiles, rodent eradication

INTRODUCTION

Islands represent approximately 5% of the land area of the Earth, yet 61% of extinctions have been insular species, and 37% of species listed by the IUCN as critically endangered are confined to islands (Tershy, et al., 2015). Invasive species are a major driver of species extinctions on islands and remain a significant risk to threatened species (Bellard, et al., 2016; Doherty, et al., 2016). Invasive rats have been introduced to approximately 80% of the archipelagos of the world, and have wide-ranging negative impacts on native flora and fauna (Towns, et al., 2006). Techniques to eradicate invasive rodents from islands are available and the practice is increasing in scope, scale, and application (Howald, et al., 2007; Keitt, et al., 2011), with restoration benefits being accrued when eradication is achieved (Jones, et al., 2016). To date there have been over 650 eradication attempts of rats (*Rattus* spp.) on more than 500 islands worldwide (Russell & Holmes, 2015).

Successful rodent eradication from islands larger than 5 ha primarily relies on the use of anticoagulant rodenticide (Howald, et al., 2007). Second generation anticoagulants are the most commonly used toxicant in invasive rodent eradication programmes (Holmes, et al., 2015). When using toxicants for rodent eradication on islands, the risk to non-target native species is typically assessed. Efforts to reduce this risk during eradication operations commonly include application of bait when susceptible species are absent, temporary captive-holding of species during potential periods of exposure, and alternative delivery methods to reduce bait access (Howald, et al., 2007). While reptiles have been known to consume cereal-based rodent baits (Merton, 1987; Marshall & Jewell, 2007), they have typically been considered at lower risk (Hoare & Hare, 2006), in part because of decreased susceptibility due to differences in blood chemistry and physiology compared to mammals and birds (Merton, 1987; Hoare & Hare, 2006). Although evidence of population level impact to reptiles is sparse, observations from an increasing number of rodenticide-based eradications, plus targeted studies, have suggested the risk is low (Harper, et al., 2011). Nevertheless, additional studies are required to improve general knowledge of the risk of rodenticides to reptiles during rodent eradication operations.

During 2012, an eradication of *Rattus rattus* using rodenticide bait was attempted on Desecheo Island located approximately 21 km off the north-west coast of Puerto Rico. Black rats were introduced in the early 1900s and are considered an important threat on Desecheo, including impacts on native reptiles from direct predation and habitat modification via seed and seedling predation and soil nutrient changes (U.S. Fish and Wildlife Service, 2016). An additional potential threat from rats to native reptiles could include competition for space and food resources, consistent with rat impacts on reptiles elsewhere (Shiels, et al., 2014; Harper & Bunbury, 2015). Two years prior to the eradication operation, exposure of bait to the endemic Desecheo ameiva (*Ameiva desechensis*) and Desecheo anole (*Anolis desechensis*) was assessed through a placebo non-toxic bait biomarker study. The study found no evidence of ameivas (n=18 marked, n=5 recaptured) interacting with bait, but 21% of anoles recaptured were exposed (n=97 marked, n=20 recaptured) (Herrera & Bermúdez-Carambot, 2010). However, because these species occur only on Desecheo, and thus had high conservation value, the fate of both lizard species was followed during the application of toxic bait during the eradication operation. Here we report the results of a mark-recapture study to monitor the short-term survival of the ameivas and the anoles before, during and after the 2012 rodent eradication operation on Desecheo Island.

MATERIALS AND METHODS

Study area

Desecheo Island is a 117 ha hilly island located approximately 21 km off the north-west coast of the Commonwealth of Puerto Rico (18° 23’ N, 67° 29’ W; Fig. 1). It was declared a U.S. National Wildlife Refuge (NWR) in 1976 and is currently administered and managed by the U.S. Fish and Wildlife Service. Sub-tropical dry forest (i.e. woodland) is present primarily on the leeward slopes and valleys, and is dominated by the semi-deciduous almácigo tree (*Bursera simaruba*). The windward slopes and ridges also harbour cacti, shrubs and open grasslands. The annual

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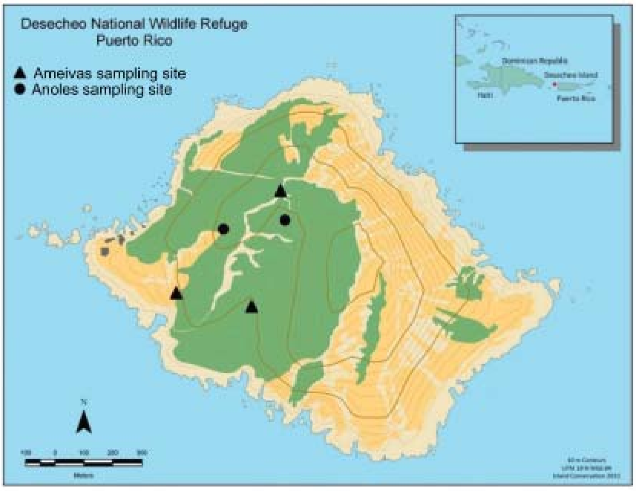

Fig. 1 Location of Desecheo Island and sampling sites for *Anolis desechensis* and *Ameiva desechensis* impact assessment during black rat eradication operations.

rainfall average is 828 mm (range 750–1039 mm; Morrison & Menzel, 1972) with a seasonal dry period between January and March, followed by a rainy season between July and November. The island supports five single-island endemic species (three lizards and two arachnids) as well as one of the three remaining populations of the threatened higo chumbo cactus (*Harrisia portoricensis*). Previous anthropogenic activities on the island included livestock grazing, military operations (e.g. bombing and gunnery range) and the introduction of invasive mammals: black rats (*Rattus rattus*), goats (*Capra hircus*), feral cats (*Felis catus)*, and rhesus macaques (*Macaca mulatta)*. The extirpation of nesting seabirds from Desecheo Island has been linked to the presence of these invasive mammals (U.S. Fish and Wildlife Service, 2016). The island is currently closed to the public due to the existence of unexploded ordnance.

Rat eradication

Aerial bait broadcast for rodent eradication was carried out on Desecheo between March 13 and 23, 2012. The bait used for the eradication was "Brodifacoum Conservation-25D" manufactured by Bell Laboratories in Madison, Wisconsin, USA. The bait was a 2 g extruded pellet, dyed green, and contained 25 ppm of the toxin brodifacoum. The bait broadcast was completed in two aerial applications separated by 10 days and with a ground application rate of 17 kg/ha for the first application and 9.1 kg/ha for the second application. There is no weather station on Desecheo Island and data were obtained from weather stations located in Rincon (13 miles from Desecheo) and Isabela (29 miles) and the Standard Precipitation Index (SPI) produced by Caribbean Regional Climate Center. January and March are usually a dry period but data from two weather stations and comparisons with 2008 and 2010 vegetation cover indicate that in 2012 Desecheo received greater than average rainfall.

Study species

The Desecheo ameiva (Fig. 2a) is a common lizard species found in coastal areas, including shoreline margins, in habitats of maximum solar exposure but frequently near some vegetation cover or shade (Evans, et al., 1991). Adult females tend to be smaller (SVL <90 mm) than males (average SVL 97 mm). Field surveys conducted in 2009 and 2010 estimated the island population at 7,469 individuals (95% CI 1,800–13,137) (McKown, et al., 2010). The Desecheo anole (Fig. 2b) is present throughout the island, but is most common in forested areas (e.g. valleys) and their margins (Evans et al., 1991). Average size for adult males is 57 mm (SVL) and for females 45 mm (SVL). Field surveys in 2009 and 2010 estimated the island population at 52,111 individuals (95% CI 31,464–72,758) (McKown et al., 2010).

Reptile monitoring

During the eradication, we implemented a reptile monitoring program between February and April 2012. We used a standard mark-recapture methodology (Jolly, 1965; Seber, 1965) over three discrete sampling periods of six days each, which coincided with bait application stages during the eradication. The first period began 21 days prior to the first bait application, the second between the first and second bait application, and the third began three days after the second bait application. The sampling sites were randomly located in five different locations within the woodland habitats in the Long and West Valleys (Fig. 1). Ameivas were sampled within one 100 × 10 m plot and

Fig. 2 *Ameiva desechensis* (a) and *Anolis desechensis* (b).

two 50 × 10 m plots. Anoles were sampled within two 100 × 10 m plots. Each plot was surveyed by two observers, each responsible for sampling one side (5 m) of a central transect through the plot. Each sampling day accounted for 8 hours of intensive searches for both species, and included the detection and capture of each observed individual. Individuals were captured using a pole and noose and by hand capture. Each anole was marked on the hind limb with a unique visible alphanumeric implant tag and each ameiva was marked with a unique combination of coloured glass beads sewed to the base of the tail (Fig 2a), and a unique combination of clipped toes (Censky, pers. comm. and modified from Fisher & Muth, 1989). Each individual was released at their capture location.

Statistical analyses

Survival of individuals was estimated using a mark-recapture model based on multiple capture histories within each sampling period (Cooch & White, 2015). We estimated the probability of recapture based on time and apparent survival to assess any potential impacts on either species as a result of the rodent bait application. We used MARK 5.0 (White & Burnham, 1999) to model factors influencing variation in survival. The Cormack-Jolly-Seber (CJS) model based on live animal recaptures in an open population (Lebreton, et al., 1992) was used to estimate the apparent survival (phi or ø). Models were constructed based on the recapture rates (*p*) and apparent survival (ø) remaining constant (.) or changing in time (*t*), and according to the bait dispersal events – before, during, and after (*asp*). The best performing model was selected using the Akaike Information Criteria (AIC) through the proportion test with Akaike weights (AICw; Burnham & Anderson, 2002). The assumptions of the CJS model were tested using TEST 2 and TEST 3 in the U-CARE program version 2.3 M 7.5 (Choquet, et al., 2005). To evaluate the fit of the set of models to the data, a Global TEST was conducted to calculate the variance inflation factor (ĉ).

RESULTS

A total of 452 anoles and 57 ameivas were captured and marked across 18 days of field sampling and 144 person-hours of sampling effort in the five study sites (Table 1). Although ameivas were detected less frequently across the study sites, they had a higher rate of recapture (35 recaptures, 61.4%) than anoles (92 recaptures, 20.4%; Table 1).

The best supported model for anoles explained the probability of recapture according to time and with apparent survival remaining constant (Table 2). For ameivas, the best supported model was the one in which the recapture probability varied across the sampling periods (i.e. bait application) and when apparent survival remained constant (Table 2). Both models indicated no changes in apparent survival along the three periods (asp) of bait applications. TEST 2 and TEST 3 showed no differences in the probability of recaptures and survival for the marked individuals (p>0.61). Global TEST indicated a sub-dispersion in the data (ĉ<1), thus no effect on the variance, therefore this parameter was not modified in the models (Cooch & White, 2015).

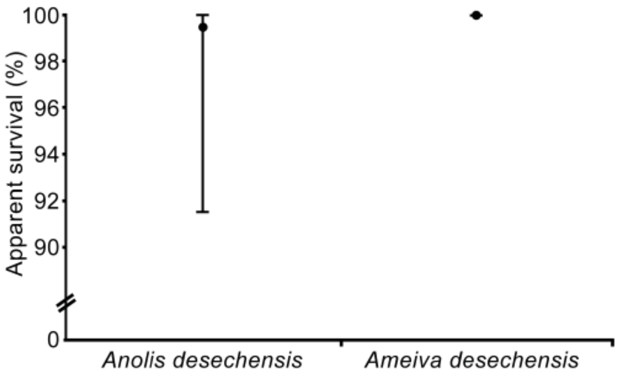

Fig. 3 Apparent survival percentage of *Anolis desechensis* and *Ameiva desechensis* during a black rat eradication on Desecheo Island (Error bars: 95% confidence intervals).

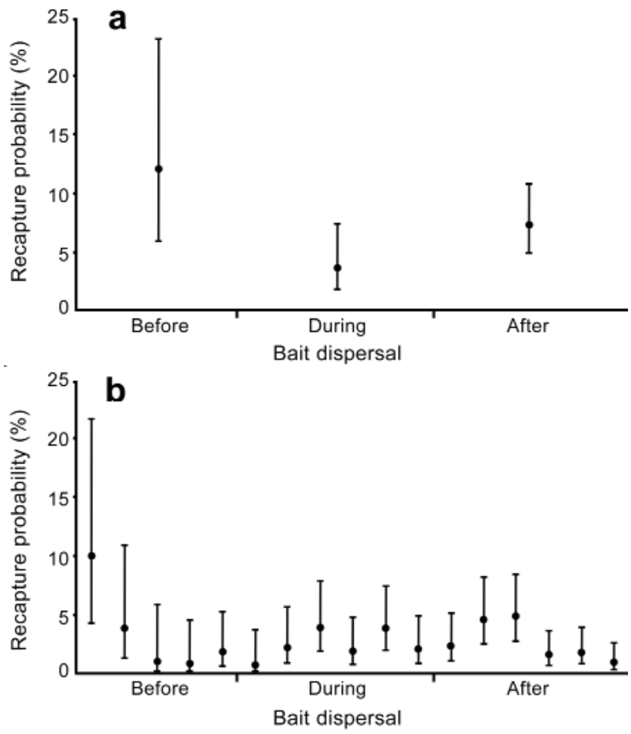

Fig. 4 Recapture probability for (a) *Anolis desechensis* and (b) *Ameiva desechensis* before, during and after bait dispersal for a black rat eradication on Desecheo Island (Error bars: 95% confidence intervals). We retain individual survey events in Figure 3b as these were found to be associated with recapture probability.

Apparent survival for both lizard species during the study period was estimated to be time-invariant and close to 100% (anoles: ø = 0.99, 95% CI = 0.91–0.99; ameivas: ø = 1, 95% CI = 1–1; Fig. 3). However, the precise apparent

Table 1 *Anolis desechensis* and *Ameiva desechensis* previously unmarked and accumulated recaptures (in parenthesis) according to sampling site and bait application stage during the black rat eradication on Desecheo Island.

Bait application stage	*Anolis desechensis*			*Ameiva desechensis*			
	Site1	Site 2	Total	Site1	Site2	Site3	Total
Before	89(2)	126(11)	215(13)	7(0)	12(3)	8(4)	27(7)
During	49(6)	46(37)	95(43)	8(3)	5(7)	3(4)	16(14)
After	75(24)	67(68)	142(92)	7(13)	3(13)	4(9)	14(35)

Table 2 Comparison of models to estimate the apparent survival (ø) and probability of recapture (*p*), according to the bait application stage (asp: before, during and after) and time (t) for *Anolis desechensis and Ameiva desechensis* during black rat eradication operations at Desecheo Island.

Model	AICc	ΔAICc	AICc weights	k	Deviance
Anolis desechensis					
ø(.) p(t)	888.04	0	0.788	18	249.89
ø(asp) p(t)	891.79	3.76	0.120	20	249.33
ø(.) p(.)	893.33	5.29	0.059	2	288.51
ø(asp) p(.)	895.79	7.76	0.016	4	286.92
ø(asp) p(asp)	896.42	8.39	0.012	6	283.46
ø(.) p(asp)	897.23	9.19	0.008	4	288.36
ø(t) p(.)	915.12	27.09	0	18	276.98
ø(t) p(asp)	915.68	27.64	0	20	273.22
ø(t) p(t)	918.92	30.89	0	33	247.56
***Ameiva desechensi*s**					
ø(.) p(asp)	268.91	0	0.615	1	163.39
ø(.) p(.)	270.44	1.52	0.287	0	169.26
ø(asp)p(asp	273.47	4.55	0.063	0	163.39
ø(asp) p(.)	274.67	5.76	0.034	0	169.15
ø(.) p(t)	287.42	18.51	<0.001	0	144.47
ø(asp) p(t)	294.05	25.13	0	0	144.47
ø(t) p(.)	311.42	42.50	0	0	168.46
ø(t) p(asp)	312.15	43.23	0	0	162.57
ø (t) p(t)	348.35	79.44	0	0	143.76

survival estimate for ameivas was not realistic due to the small sample size (Fig. 3). An effect of time across bait dispersal over the recapture probability was found in the ameiva, with a tendency to decrease during and after bait dispersal (Fig. 4a). In contrast, for the recapture probability of the anoles there was no pattern associated with bait dispersal, but this variation was related to survey events (Fig. 4b). No mortality was observed for either species.

DISCUSSION

We estimated the survival and recapture rates of two native reptile species during a black rat eradication on Desecheo Island, Puerto Rico. During our study, we found no significant change in apparent survival rates across the sampling periods in anoles or ameivas, indicating that the application of rodenticide bait did not result in any detectable mortality or negative effect on both populations. Furthermore, the recapture probabilities for anoles varied through time (between survey events), but were not dependent on bait application, suggesting that while anoles were exposed to rodent bait (23% of individuals), exposure did not impact survivorship within the sampling period. For ameivas, the placebo-bait biomarker study found no direct or indirect exposure of ameivas to rodent bait. For the current study, the precise apparent survival estimate for ameivas was influenced by the small sample size and was not considered statistically valid. The recapture probability estimate for the species decreased during bait application and then increased following the bait application, which may have been an artefact of increased human activity during the operation affecting movement of these animals.

Behavioural ecology, diet, and foraging habitat of lizards are important considerations in understanding potential pathways of exposure to rodenticides. Although we did not observe anoles or ameivas feeding directly on the placebo biomarker or toxic bait, other studies have shown direct consumption of bait by different reptile species (Merton, 1987; Merton, et al., 2002). Bait availability monitoring showed bait disappeared three days after the second bait application, thus removing a pathway of direct exposure (consumption) for ameivas and anoles. However, we anticipate that anoles were exposed to bait via indirect pathways through consumption of invertebrates feeding on bait. Few anole species are dietary specialists and most species, including the Desecheo anole, consume a wide variety of insects and fruit (Meier & Noble, 1991). The ameiva, a larger species than the anole, primarily forages on the ground where it could be easily exposed to bait through secondary pathways (e.g. ground-foraging beetles and ants that feed on bait).

Delayed response to toxicant impacts on reptiles has been previously reported. Telfair's skinks (*Leiolopisma telfairii*) on Round Island, Mauritius, showed an apparent increased mortality three to six weeks after a brodifacoum bait application (Merton, 1987) and Harper et al. (2011) estimated 4.5% mortality of the Galápagos marine iguana up to two months following rat eradication using brodifacoum. While toxicant as the cause of death was not confirmed during these events, a cautious approach suggests it be considered a risk. While our study was undertaken for approximately three weeks (22 days) after bait was dispersed, the impacts of the rodenticide could not be assessed beyond this timeframe.

This study focused on the survivorship of two reptile species because of the high conservation value of these single-island endemics. Rodenticide application risk assessments should also consider the role of lizards as prey items, and thus as potential toxin pathways to other native species. Food web models that include rodenticide introduction can inform risk assessments, including potential pathways and levels of exposure. Residue analyses can help confirm these assessments. Ultimately, risk assessments for rodent eradication operations using toxicants must evaluate the cost and benefit impacts of these efforts (i.e. negative impacts from using toxicants versus positive impacts from removing rats). Whereas reports of individual reptile mortality during rodenticide-based eradications are evident (Merton, 1987; Harper, et al., 2011) a greater body of evidence suggests that reptile populations benefit following rodent eradication (Jones, et al., 2016). Studies such as ours provide another case study to evaluate the value of island restoration efforts on reptiles. The combination of studies such as these can help managers make informed decisions about the potential negative impacts of rodenticides used during eradication operations versus the expected positive impact to native biota from the permanent removal of threats posed by invasive species.

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REFERENCES

Bellard, C., Cassey, P. and Blackburn, T.M. (2016). 'Alien species as a driver of recent extinctions'. *Biology Letters* 12: 20150623.

Burnham, K.P. and Anderson, D.R. (2002). *Model Selection and Multi Model Inference: A Practical Information-theoretic Approach*. New York: Springer-Verlag.

Choquet, R, Reboulet, A.M., Lebreton, J.D., Gimenez, O. and Pradel, R. (2005). *U-care 2.2 User´s Manual. CEFE, Montpellier, France*. <https://ftp.cefe.cnrs.fr/biom/Soft-CR/>.

Cooch, E. and White, G. (2015). *Program MARK: A Gentle Introduction. Colorado State University, Fort Collins*. < http://www.fwspubs.org/doi/suppl/10.3996/122012-JFWM-110R1/suppl_file/10.3996_122012-jfwm-110r1.s8.pdf?code=ufws-site>.

Doherty, T.S., Glen, A.S., Nimmo, D.G., Ritchie, E.G and Dickman, C.R. (2016). 'Invasive predators and global biodiversity loss'. *Proceedings of the National Academy of Sciences'* 113: 11261–11265.

Evans, M., Herbert, H.J. and Rohnke, K. (1991). 'Observations on the Status of the Herpetofauna of Desecheo Island National Wildlife Refuge, Puerto Rico'. In: J.A. Moreno (ed.) *Estatus y Distribución de los Reptiles y Anfibios de la Región de Puerto Rico*, pp. 34–36. San Juan: Departamento de Recursos Naturales de Puerto Rico.

Fisher, M. and Muth, A. (1989). 'A technique for permanently marking lizards'. *Herpetological Review* 20: 45–46.

Harper, G.A. and Bunbury, N. (2015). 'Invasive rats on tropical islands: Their population biology and impacts on native species'. *Global Ecology and Conservation* 3: 607–627.

Harper, G.A., Zabala, J. and Carrion, V. (2011). 'Monitoring of a population of Galapagos land iguanas (*Conolophus subcristatus*) during a rat eradication using brodifacoum'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 309–312. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Herrera, J.L. and Bermúdez-Carambot, E.N. (2010). *Survey of the Desecheo Dwarf Sphaero,* Sphaerodactylus levinsi *(Squamata: Gekkonidae) and a Mark-recapture Study with* Anolis desechensis *and* Ameiva desechensis*, Desecheo Island, Puerto Rico*. Internal report to Island Conservation. Unpublished.

Hoare, J.M. and Hare, K.M. (2006). 'The impact of brodifacoum on non-target wildlife: Gaps in knowledge'. *New Zealand Journal of Ecology* 30: 157–167.

Holmes, N.D., Griffiths, R., Pott, M., Alifano, A., Will, D., Wegmann, A.S. and Russell, J.C. (2015). 'Factors associated with rodent eradication failure'. *Biological Conservation* 185: 8–16.

Howald, G., Donlan, C.J, Galvan, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Jolly, G.M. (1965). 'Explicit estimates from capture-recapture data with both death and immigration-stochastic model'. *Biometrika* 52: 225–247.

Jones, H.P., Holmes, N.D., Butchart, S.H.M., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P.E., Cuthbert, R.J., Ebbert, S., Genovesi, P., 'Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P. J., Spatz, D.R., Towns, D.R. and Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences* 113: 4033–4038.

Keitt, B., Campbell, K., Saunders, A., Clout, M., Wang, Y., Heinz, R., Newton, K and Tershy, B. (2011). 'The Global Islands Invasive Wertebrate Eradication Database: A Tool to Improve and Facilitate Restoration of Island Ecosystems'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 74–77. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Lebreton, J.D., Burnham, K.P., Clobert, J. and Anderson, D.R. (1992). 'Modelling survival and testing biological hypotheses using marked animals: A unified approach with case studies'. *Ecological Monographs* 62: 67–118.

Marshall, J.E. and Jewell, T. (2007). 'Consumption of non-toxic baits by grand (*Oligosoma grande*) and Otago (*O. otagense*) skinks'. *DOC Research and Development Series* 272: 5–11.

McKown, M., Swinnerton, K. and Herrera-Giraldo, J.L. (2010). *Desecheo Reptile Monitoring Data.* Internal report to Island Conservation. Unpublished.

Meier, A.J. and Noble, R.E. (1991). 'Notes on the natural history of *Anolis desechensis'*. *Florida Field Naturalist* 19: 17–18.

Merton, D. (1987). 'Eradication of rabbits from Round Island, Mauritius: A conservation success story'. *Dodo* 24: 19–43.

Merton, D., Climo, G., Laboudallon, V., Robert, S. and Mander, C. (2002). 'Alien mammal eradication and quarantine on inhabited islands in the Seychelles'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 182–198. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Morrison, J.A and Menzel Jr, E.W. (1972). 'Adaptation of a free-ranging rhesus monkey group to division and transplantation'. *Wildlife Monographs* 31: 1–79.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Seber, G.A.F. (1965). 'A note on the multiple recapture census'. *Biometrika* 52: 249–259.

Shiels, A.B., Pitt, W.C., Sugihara, R.T. and Witmer, G.W. (2014). 'Biology and impacts of Pacific Island invasive species. 11. *Rattus rattus*, the black rat (Rodentia: Muridae)'. *Pacific Science* 68: 145–184.

Tershy, B.R., Shen, K.W., Newton, K.M., Holmes, N.D. and Croll, D.A. (2015). 'The importance of islands for the protection of biological and linguistic diversity'. *BioScience*: 10.1093/biosci/biv031.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?' *Biological Invasions* 8: 863–891.

U.S. Fish and Wildlife Service (2016). *Final Environmental Assessment: Desecheo National Wildlife Refuge Rat Eradication Project*. U.S. Department of the Interior Fish and Wildlife Service, San Juan. <https://www.fws.gov/caribbean/refuges/desecheo/>.

White, G.C. and Burnham, K.P. (1999). 'Program MARK: Survival estimation from populations of marked animals'. *Bird Study* 46: 120–138.

Long term rodent control in Rdum tal-Madonna yelkouan shearwater colony

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Abstract Rodent predation on eggs and chicks is one of the main threats to procellariiform species in the Mediterranean, where the black rat (*Rattus rattus*) and brown rat (*R. norvegicus*) have been present on many islands for centuries. The yelkouan shearwater (*Puffinus yelkouan*) is an endemic Mediterranean seabird species classified as vulnerable. Malta holds up to 10% of the global population; the largest colony, Rdum tal-Madonna (RM), protected as a Natura 2000 site, hosts around 500 breeding pairs. This colony has been monitored since its discovery in 1969. A very low reproductive success due to rat predation was noticed in the late 1990s to early 2000s. In 2007 a seasonal rodent control programme was established during the breeding season of yelkouan shearwater to reduce rat predation on eggs and chicks. Rodent control took place between 2007 and 2010 and was reviewed and continued from 2012 to 2017. Breeding success since 2007 has been higher than 80%. In two other colonies with rat presence and where rodent control did not take place, the breeding success in 2016 and 2017 was substantially lower than in the colony with the rodent control programme. The European storm-petrel (*Hydrobates pelagicus melitensis*) only breeds in rat-free areas like remote sea caves or islets around the Maltese islands. In 2014, the first breeding attempt by European storm-petrel was recorded on the Maltese mainland at RM with a chick fledging successfully for the first time in 2016. The ongoing LIFE Arcipelagu Garnija project is assessing rat predation in all Maltese yelkouan shearwater colonies in order to establish predator control in the most important yelkouan shearwater breeding sites in 2018.

Keywords: breeding success, chicks, eggs, littering, rats, seabirds

INTRODUCTION

Malta is a southern European archipelago in the Mediterranean Sea with three main islands: Malta, Gozo and Comino; and other important islets: Filfla, Saint Paul, Fungus Rock and Cominotto. Each island and islet harbours important colonies of seabirds. The archipelago lies 80 km from the south of Sicily (Italy), 284 km from the east of Tunisia and 333 km from the north of Libya. The islands cover over 315 km². Malta hosts internationally important breeding populations of procellariiforms: yelkouan shearwater (*Puffinus yelkouan*) (estimated 1370–2000 pairs, constituting up to 10% of the global population) (Metzger, et al., 2015), Scopoli's shearwater (*Calonectris diomedea*) (estimated 4,500 pairs, up to 5 % of the global population) and European storm-petrel (*Hydrobates pelagicus melitensis)* (estimated 5,000–8,000 pairs, around 50% of the Mediterranean population) (Sultana, et al., 2011).

The invasion of ecosystems by introduced species is one of the most significant sources of ecosystem change (Howald, et al., 2007) and biodiversity loss on islands (Martin, et al., 2000; Courchamp, et al., 2003). Introduction of alien rodents has been shown to have devastating effects on insular ecosystems and some rodent species can be important predators of nesting seabirds (Traveset, et al., 2009), especially procellariiforms (Imber, 1978). Rodent predation on eggs and chicks is one of the main dangers to this group of seabirds across the world (Booth, et al., 1996; Hobson, et al., 1999; Gaze, 2000; Imber, et al., 2000). Rats are associated with extinctions or declines of burrowing seabirds (Seto & Conant, 1996; Towns, et al., 2006). Rats have a severe impact on breeding success and are a major cause of seabird mortality in the world (Jones, et al., 2008; Pascal, et al., 2008).

Rats were introduced into the Mediterranean over 2000 years ago and have been present on many islands for centuries (Atkinson, 1985; Audoin-Rouzeau & Vigne, 1994; Martin, et al., 2000). Black rat (*Rattus rattus*) is the most devastating predator of seabirds in the Mediterranean (Igual, et al., 2006) and the main reason for breeding failure on some islands, for example Corsica (Thibault, 1995). Therefore, the persistence of native long-lived seabirds in the Mediterranean basin, despite the long-standing introduction of black rat on most islands, constitutes an amazing conservation paradox (Ruffino, et al., 2009).

Yelkouan shearwater is an endemic Mediterranean seabird belonging to the family Procellariidae. It is a long-lived species that lays a single egg each season in deep burrows. It has been classified as vulnerable since 2012 according to the IUCN (BirdLife International, 2016). The Maltese population of yelkouan shearwater has declined in recent years, mainly due to predation by rats, loss of breeding habitat, illegal hunting, fishing bycatch, disturbance and light and sound pollution (Sultana, et al., 2011).

The main colony in Malta situated in Rdum tal-Madonna (RM) holds around 500 breeding pairs (2 or 3% of the global population). It is a Natura 2000 site – part of the European network of protected areas. This colony is situated along 1 km of coralline limestone sea cliff. It has been monitored since its discovery in 1969 and it was noticed that the breeding success in the late 1990s to early 2000s was very low, largely due to rat predation, with very few chicks fledging (Sultana, et al., 2011). The best response to such a situation is almost always to control the alien population, either by frequently reducing their numbers, or better still, by eradicating the whole population (Courchamp, et al., 2003).

As the colony is located on the Maltese main island, eradication of rats was not feasible because it is not possible to isolate the area from rat populations found across the rest of the island. The population of rats benefits from the persistent availability of food close to the colony. Litter from recreational users in the area making barbecues and camping is compounded by the inefficient and inadequate waste disposal and collection system. Actions to increase awareness about littering between site users and authorities were carried out but no substantial improvement in the situation was observed.

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In 2007, a seasonal rodent control programme was established at the site to reduce rat predation. The control programme has now been active for 11 years from 2007 to 2017. In this paper we present the results of the rodent control programme on the breeding success of the yelkouan shearwater colony. We discuss the results and lessons learnt and their applicability to other locations.

MATERIALS AND METHODS

The colony site is surrounded by the ocean on three sides, making it an ideal site for rodent control. The methodology chosen for rodent control was seasonal control using rodenticide. The most frequent rodenticide distribution method used on small islands around the world is bait stations (Howald, et al., 2007) and other projects have shown that using a permanent bait-station system is an efficient methodology to control rats (Orueta, et al., 2005; Pascal, et al., 2008). Around 90 bait stations (PROTEXX TM) were distributed over 25 ha of RM on the top of the cliff plateau and the lower part of the cliffs where yelkouan shearwaters breed (Fig.1). The bait stations on top of the cliffs create a buffer to prevent rats accessing the colony. Bait stations were placed around areas of high rat presence, for example those areas subject to littering from campers. Rodent control took place around nesting sites between February and July during the yelkouan shearwater breeding season, when eggs and chicks are most vulnerable. The bait stations were baited one to three times per month, depending on rodent activity. Each bait station contained two blocks of anticoagulant rodenticide. Between 2007 and 2015 the rodenticide used was brodifacoum 0.005% and from 2016 it was bromadiolone 0.005% to reduce the risk of secondary poisoning. The bait blocks were threaded on to metal skewers that were clipped in place, to prevent them falling out of the stations even if they were shaken violently.

Every time the bait stations were checked, data were collected on the amount of rat sign (droppings and rat teeth marks in the wax bait blocks), non-target species sign like mice, shrews and insects taking the bait, and the number of bait blocks replaced. The area baited was checked for signs of dead rats and primary or secondary poisoning of non-target species. Rat presence was calculated as the number of bait stations with rat teeth marks on the bait divided by the total number of bait stations.

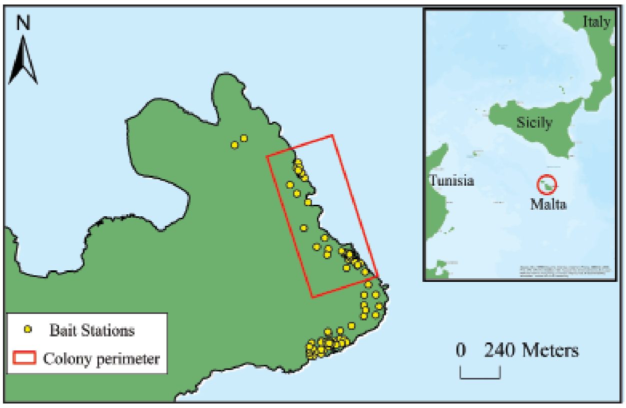

Fig. 1 Map of Rdum tal-Madonna colony in Malta where the Yelkouan shearwater colony is situated (rectangle) and the location of the bait stations (dots).

Table 1 Breeding success (% of chicks fledged per eggs laid) of yelkouan shearwater at Rdum tal-Madonna between 2007 and 2017.

Year	No. of nests	Breeding success
2007	6	83%
2008	12	92%
2009	11	91%
2012	16	94%
2013	32	88%
2014	25	88%
2016	24	88%
2017	38	84%

Table 2 Breeding success (% of chicks fledged per eggs laid) of yelkouan shearwater in 2016 and 2017 in Rdum tal-Madonna (rodent control) and St. Paul's Island and Majjistral (no rodent control).

Colony	Year	No. of nests	Breeding success
RM	2016	24	88%
St. Paul's Is	2016	9	67%
Majjistral	2016	12	33%
RM	2017	38	84%
St. Paul's Is	2017	9	11%
Majjistral	2017	11	55%

RESULTS

Rodent control took place between 2007 and 2010, after which it was reviewed and then continued from 2012 to 2017. After the first season of rodent control in 2007, the occurrence of eggs and chicks depredated by rats dropped dramatically and there have been few recorded signs of rat predation during the subsequent 11 years. Breeding success has been very high since rodent control started (Table 1), with a mean of 88 % (averaged over the eight years for which data are available).

In 2016 and 2017, the breeding success (chicks fledged per eggs laid) in RM (88% and 84%, respectively) was much higher than in two other colonies where rats were known to be present but no rat control took place, St. Paul's Island (67% and 11%, respectively) and Majjistral Park (33% and 55%, respectively) (Table 2).

In RM, rat activity varies throughout the yelkouan shearwater breeding season. Rats are regularly present from February until July. Rat activity is reduced after the first month of rat control in February, the peak of activity is in May and then a small upturn in June (Fig. 3). Rodent activity over the period 2012–2017 (data available for four years) shows a decrease in rat presence in recent years. No signs of secondary poisoning have been found in the study period.

After the first season of rat control in 2007, European storm-petrels were regularly seen in RM (Borg, et al., 2010). In 2014 the first breeding attempt was recorded and in 2016 and 2017 chicks fledged successfully. The data collected during 2014–2017 suggest European Storm-petrel is establishing a breeding colony in RM.

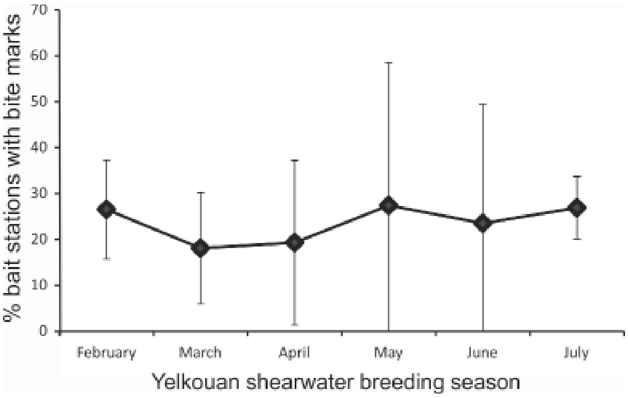

Fig. 2 Proportion (%; with mean and standard deviation) of bait stations with rodent bite marks throughout the yelkouan shearwater breeding season from February to July (for years 2012, 2015, 2016, 2017 combined).

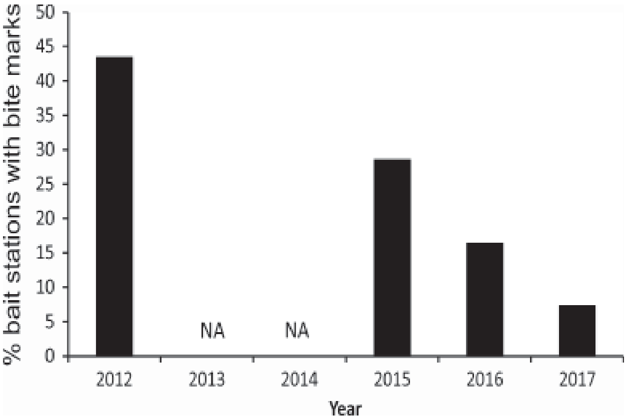

Fig.3 The percentage of bait stations with rodent bite marks by year; 2012, 2015, 2016 and 2017.

DISCUSSION

Seasonal rat control in seabird colonies where eradication is not feasible is an effective way to reduce rat predation and increase reproductive success (Imber, et al., 2000; Martin, et al., 2000; Jouventin, et al., 2003; Orueta, et al., 2005; Igual, et al., 2006; Pascal, et al., 2008). In many cases, the removal of the alien invasive species is followed by a fast and often great recovery of the damaged local populations (Courchamp, et al., 2003), even allowing new colonies of other species to become established, as has been seen at RM (Malta). However, only intensive and constant long-term poisoning will control rats satisfactorily (Jouventin, et al., 2003).

The increase in the reproductive success observed during recent years in the yelkouan shearwater colony in RM is correlated with the lower rat activity as a result of rodent control programme. Rat activity varies throughout the yelkouan shearwater breeding season. The peak of activity in May is related to the increase in ambient temperature and also to the start of camping activity in the area. The presence of campers increases littering (i.e. supplying food for rodents) which is the likely reason for the increase in the rat population around the colony. The general decrease of rat presence in 2016 and 2017 may be related to the very dry weather in these two years, but possibly also to increased public awareness about littering. On 30 April 2017, an intensive clean-up by more than 100 volunteers was organised in the area.

The rodent control programme showed its effectiveness at increasing the breeding success of yelkouan shearwater and allowed the establishment of a new European storm-petrel population. The main yelkouan shearwater colony locations are situated on the main islands of Malta and Gozo making the eradication of rats not possible. Rat eradication could only be feasible in the islands of Comino, Cominotto and Saint Paul that hold smaller colonies. Ongoing rodent control programmes are therefore needed in the main colonies to secure yelkouan shearwater populations in the archipelago and to improve their situation. Building on the lessons learnt and the success of the rodent control programme in RM, the current EU-Life Arċipelagu Garnija project LIFE14 NAT/MT/991 is assessing predation by rats in all Maltese yelkouan shearwater colonies in order to establish predator control in the most important sites in 2018 and secure the main colonies across Malta. During the study period, no evidence of secondary poisoning was found but, in any case, from 2016 the bait was changed from brodifacoum to bromadiolone that has less risk of secondary poisoning. Less toxic bait, such as first generation anticoagulants, are not available in Malta. In order to reduce the amount of anticoagulant used in the new rat control programmes, the current project is testing methodologies to replace or combine anticoagulant baiting with auto-reset mechanical traps and carrying out activities to increase awareness about littering among site users.

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REFERENCES

Atkinson, I.A.E. (1985). 'The spread of commensal species of *Rattus* to oceanic islands and their effects on island avifaunas'. In: P.J. Moors (ed.) *Conservation of Island Birds*, pp. 35–81. ICBP Technical Publication 3.

Audoin-Rouzeau F. and Vigne J.D. (1994). 'La colonisation de l'Europe par le rat noir *Rattus rattus*'. *Revue de Paléobiologia*13: 125–145.

BirdLife International (2016). Puffinus yelkouan. *The IUCN Red List of Threatened Species 2016*: e.T22698230A93672084. <http://dx.doi.org/10.2305/IUCN.UK.2016-3.RLTS.T22698230A93672084.en>. Downloaded on 13 February 2018.

Booth, A.M., Minot, E.O., Fordham, R.A. and Innes, J.G. (1996). 'Kiore (*Rattus exulans*) predation on the eggs of the little shearwater (*Puffinus assimilis haurakiensis*)'. *Notornis* 43: 147–153.

Borg, J.J., Raine, A. and Raine, H. (2010). 'Records in Malta of European storm-petrels *Hydrobates pelagicus melitensis* during the breeding season away from known breeding sites'. *Il-Merill* 32: 11–14.

Courchamp, F., Chapuis, J.L. and Pascal, M. (2003). 'Mammal invaders on islands: Impact, control and control impact'. *Biological Reviews* 78(3): 347–383.

Gaze, P. (2000). 'The response of a colony of sooty shearwater (*Puffinus griseus*) and flesh-footed shearwater (*P. carneipes*) to the cessation of harvesting and the eradication of Norway rats (*Rattus norvegicus*)'. *New Zealand Journal of* Zoology 27: 375–379.

Hobson, K.A., Drever, M.C. and Kaiser, G.W. (1999). 'Norway rats as predators of burrow-nesting seabirds: insights from stable isotope analyses'. *The Journal of Wildlife Management* 63: 14–25.

Howald, G., Donlan, C., Galván, J.P., Russell, J.C., Parkes, J., Samaniego, A.,Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Tershy, B. and Saunders, A. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21(5): 1258–1268.

Igual, J.M., Forero, M.G., Gomez, T., Orueta, J.F. and Oro, D. (2006). 'Rat control and breeding performance in Cory's shearwater (*Calonectris diomedea*): Effects of poisoning effort and habitat features'. *Animal Conservation* 9(1): 59–65.

Imber, M.J. (1978). 'The effects of rats on breeding success of petrels'. In: P.R. Dingwall, I.A.E. Atkinson and C. Hay (eds.) *The Ecology and Control of Rodents in New Zealand Nature Reserves,* pp. 67–71. Wellington: Department of Lands and Survey Information Series.

Imber, M., Harrison, M. and Harrison, J. (2000). 'Interactions between petrels, rats and rabbits on Whale Island, and effects of rat and rabbit eradication'. *New Zealand Journal of Ecology* 24: 153–160.

Jones, H.P., Tershy, B.R., Zavaleta, E.S., Croll, D.A., Keitt, B.S., Finkelstein, M.E. and Howald, G.R. (2008). 'Severity of the effects of invasive rats on seabirds: a global review'. *Conservation Biology* 22(1): 16–26.

Jouventin, P., Bried J. and Micol, T. (2003). 'Insular bird populations can be saved from rats: a long-term experimental study of white-chinned petrels *Procellaria aequinoctialis* on Île de la Possession (Crozet archipelago)'. *Polar Biology* 26: 371–378.

Martin, J.L., Thibault, J.C. and Bretagnolle, V. (2000). 'Black rats, island characteristics, and colonial nesting birds in the Mediterranean: consequences of an ancient introduction'. *Conservation Biology* 14: 1452–1466.

Metzger, B., Oppel, S., Carroll, M., Meirinho, A., Dias, M., Barbara, N. and Lago, P. (2015). *Malta Marine IBA Inventory Report. EU LIFE+Malta Seabird Project LIFE10 NAT/MT/090*. BirdLife Malta. [online] available at: <https://birdlifemalta.org/conservation/malta-seabirdproject/> Accessed 15 March 2018.

Orueta, J.F., Aranda, Y., Gómez, T., Tapia, G.G. and Sanchez-Marmol, L. (2005). 'Successful eradication of invasive rodents from a small island through pulsed baiting inside covered stations'. *Biological invasions* 7(1): 141–147.

Pascal, M., Lorvelec, O., Bretagnolle, V. and Culioli, J.M. (2008). 'Improving the breeding success of a colonial seabird: a cost-benefit comparison of the eradication and control of its rat predator'. *Endangered Species Research*4(3): 267–276.

Ruffino, L., Bourgeois, K., Vidal, E., Duhem, C., Paracuellos, M., Escribano, F., Sposimo, P., Baccetti N., Pascal, M. and Oro, D. (2009). 'Invasive rats and seabirds after 2,000 years of an unwanted coexistence on Mediterranean islands'. *Biological invasions* 11(7): 1631–1651.

Seto, N.W.H. and Conant, S. (1996). 'The effects of rat (*Rattus rattus*) predation on the reproductive success of the Bonin petrel (*Pterodroma hypoleuca*) on Midway Atoll'. *Colonial Waterbirds* 19 (2):171–185.

Sultana, J., Borg, J.J., Gauci, C. and Falzon, V. (2011). *The breeding birds of Malta*. Malta: Birdlife Malta.

Thibault, J.C. (1995). 'Effect of predation by the black rat *Rattus rattus* on the breeding success of Cory's Shearwater *Calonectris diomedea* in Corsica'. *Marine Ornithology* 23: 1–10.

Towns, D.R., Atkinson, I.A. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?'. *Biological Invasions* 8: 863–891.

Traveset, A., Nogales, M., Alcover, J.A., Delgado, J.D., López-Darias, M., Godoy, D., Igual, J.M. and Bover, P. (2009). 'A review on the effects of alien rodents in the Balearic (Western Mediterranean Sea) and Canary Islands (Eastern Atlantic Ocean)'. *Biological Invasions*11: 1653–1670.

Seasonal variation in movements and survival of invasive Pacific rats on sub-tropical Henderson Island: implications for eradication

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Abstract Invasive rodents are successful colonists of many ecosystems around the world, and can have very flexible foraging behaviours that lead to differences in spatial ranges and seasonal demography among individuals and islands. Understanding such spatial and temporal information is critical to plan rodent eradication operations, and a detailed examination of an island's rat population can expand our knowledge about possible variation in behaviour and demography of invasive rats in general. Here we investigated the movements and survival of Pacific rats (*Rattus exulans*) over five months on sub-tropical Henderson Island in the South Pacific Ocean four years after a failed eradication operation. We estimated movement distances, home range sizes and monthly survival using a spatially-explicit Cormack-Jolly-Seber model and examined how movement and survival varied over time. We captured and marked 810 rats and found a median maximum distance between capture locations of 39 ± 25 m (0–107 m) in a coastal coconut grove and 61 ± 127 m (0–1,023 m) on the inland coral plateau. Estimated home range radii of Pacific rats on the coral plateau varied between 'territorial' (median: 134 m; 95% credible interval 106–165 m) and 'roaming' rats (median: 778 m; 290–1,633 m). The proportion of rats belonging to the 'roaming' movement type varied from 1% in early June to 23% in October. There was no evidence to suggest that rats on Henderson in 2015 had home ranges that would limit their ability to encounter bait, making it unlikely that limited movement contributed to the eradication failure if the pattern we found in 2015 is consistent across years. We found a temporal pattern in monthly survival probability, with monthly survival probabilities of 0.352 (0.081–0.737) in late July and 0.950 (0.846–0.987) in late August. If seasonal variation in survival probability is indicative of resource limitations and consistent across years, an eradication operation in late July would likely have the greatest probability of success.

Keywords: home range, introduced species, island restoration, Pitcairn Islands, *Rattus exulans*

INTRODUCTION

Eradications are a powerful and frequently used management option to counter the native biodiversity loss caused by invasive species on islands (Jones, et al., 2016). Planning for an eradication requires a fundamental understanding of the ecology and movement characteristics of the target invasive species (Zavaleta, 2002; Keitt, et al., 2015). Among the most widespread invasive species on islands are three species of rat (*Rattus rattus*, *R. norvegicus*, *R. exulans*), which now occur on >80% of the world's island groups (Atkinson, 1985; Jones, 2010). Rat eradications have been successfully completed on hundreds of islands (Howald, et al., 2007), but eradications on tropical islands, where a lack of seasonal fluctuation in resource abundance allows rodents to reproduce throughout the year, still have a lower success probability than eradications on temperate islands (Holmes, et al., 2015; Keitt, et al., 2015). Detailed information on rat movements and demography from tropical islands should therefore benefit eradication planning on tropical islands (Keitt, et al., 2015).

Rodent eradications on islands larger than 100 ha are generally conducted by aerially distributing cereal-based toxic bait pellets across the island, and are only successful if every individual rodent has access to sufficient bait within its home range to consume a lethal dose of toxin (Cromarty, et al., 2002; Howald, et al., 2007; Broome, et al., 2014; Holmes, et al., 2015). Hence, a better understanding of the size of home ranges can inform the density at which bait pellets need to be dispersed on the ground. Movements of invasive rodents on islands vary by habitat, population density, food availability, individuals' age and sex (Bramley, 2014a; Ringler, et al., 2014; Harper, et al., 2015), but more information on the size of movements and their variation over time of year could contribute to eradication planning on islands.

Besides ensuring each individual has access to a sufficient quantity of bait, rodent eradications are also more likely to succeed if they are timed to coincide with a predictable period of rodent stress (e.g. mortality). On temperate islands, mortality occurs during a predictable seasonal shortage in resource availability during autumn or winter, and therefore provides a natural time window for an eradication operation when rodents are more likely to switch to palatable poison baits (Howald, et al., 2007; Russell & Ruffino, 2012). On tropical islands, with less-defined seasonality and irregular periods of resource limitation, there is still very little information on how the survival of rodents varies within a year (but see Tamarin & Malecha, 1971). Additional information on seasonal variation in survival of rodents on tropical islands can inform when an eradication operation would have the highest probability of success and therefore aid the planning of an eradication operation (Howald, et al., 2007; Holmes, et al., 2015; Keitt, et al., 2015).

Here we use data from a large spatial capture-recapture programme and conventional radio-tracking to investigate the movements of invasive Pacific rats (*R. exulans*) on an uninhabited sub-tropical island (Henderson) in the South Pacific. An eradication operation on this island in 2011 failed to kill all individuals. Among the reasons that can cause eradication failure, insufficient bait toxicity could be excluded due to follow-up experiments (Amos, et al., 2016). However, two further potential causes, namely that not all rats had access to bait and that the eradication was poorly timed and coincided with high survival, have not been investigated so far. Our study was designed to provide knowledge to better understand the 2011 eradication failure and improve the probability of success of a future eradication attempt. We estimate movement distances and home range sizes using mark-recapture and radio-tracking

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data and evaluate if the smallest rodent home ranges would contain a sufficient quantity of bait pellets based on bait distribution rates used during the eradication attempt in 2011. We further estimate survival of rats over a five-month period, examine temporal variation in their monthly survival probability, and assess whether the timing of the failed operation in 2011 was appropriate.

METHODS

Study area

Henderson Island (24º22′ S, 128º20′ W) is a flat, raised coral atoll of 4,309 ha in the sub-tropical Pacific Ocean with two distinct habitats – a central plateau roughly 25 m above sea level (4,290 ha), and a sandy beach area with a vegetated margin (hereafter referred to as 'embayment forest', 14 ha). Henderson Island has a sub-tropical climate with erratic rainfall patterns, and there are no permanent freshwater bodies on the island (Spencer, 1995; Weigelt, et al., 2013). The plateau substrate is fossilised coral with uniform, dense native vegetation consisting mostly of *Pandanus tectorius*, *Xylosma suaveolens* and *Psydrax odorata* (Waldren, et al., 1995). The beach and embayment forest areas have a sandy substrate with a mixed shrubby vegetation and small stands of introduced coconut (*Cocos nucifera*) (Paulay & Spencer, 1989; Waldren, et al., 1995).

Pacific rats were introduced to Henderson Island by Polynesians several hundred years ago (Steadman & Olson, 1985), and currently have adverse effects on native biodiversity on Henderson Island (Brooke, et al., 2010; Dawson, et al., 2015). In late August 2011, an operation using the aerial distribution of cereal-based pellets containing 20 µg/g of the toxin brodifacoum was carried out to eradicate all Pacific rats from Henderson Island. Although the baiting operation met best practice standards, had no spatial gaps in bait distribution, used bait pellets containing a sufficient amount of toxin (Torr & Brown, 2012), and used bait application densities well beyond those needed to overcome estimated hermit-crab consumption (Cuthbert, et al., 2012), the eradication operation was unsuccessful and 60-80 individual rats were predicted to have survived (Amos, et al., 2016). Rat populations recovered within 2–4 years (Bond, et al., 2019) and were at an unknown stage of expansion or fluctuation during 2013 and 2015.

Rat live trapping

To obtain a robust estimate of rat survival probability, and to document rat movements over five months, we implemented a spatial capture-mark-recapture programme in 2015. Rats were live-trapped on the plateau from 28 May to 16 October 2015 during seven primary sessions of 10 trapping nights each, with a window of 8–15 days with no trapping between primary sessions. This time frame was chosen because food availability for rats was assumed to be lower during the 'winter' months on Henderson than at other times of the year (Spencer, 1995; Brooke & Towns, 2008). In the embayment forest, rats were live-trapped between 1 August and 19 September 2015 during three primary sessions of 6–10 trapping nights each.

On the plateau we established a trap network placed along 3 km of cleared path (Fig. 1). Traps were arranged at distances from 3–20 m at 343 locations, with a different subset of trap locations used during each primary session due to gradual progression of trail clearance. In the embayment forest, we established a grid of 63 traps arranged in an oblique rectangular configuration (Fig. 1) with traps spaced 10 m apart. Traps were placed on the ground, marked with a unique number, and locations were recorded to within 5 m using a hand-held GPS device.

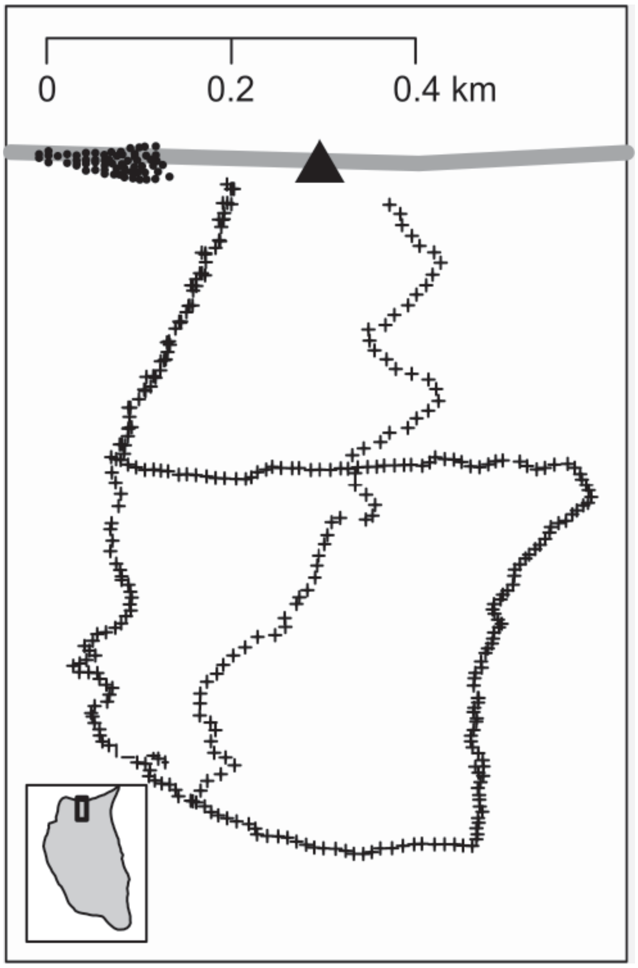

Fig. 1 Map of the trapping network used on Henderson Island in 2015. Black triangle is the research camp, the grey line is the upper margin of the beach, + indicate trap locations on the coral plateau, and black dots indicate trap locations in the embayment forest.

We used two different live trap types, a small metal box (7.6 × 8.9 × 22.9 cm, LFA Folding Trap, H. B. Sherman Traps Inc., Tallahassee, Florida, USA), and a metal cage (13 × 16 × 27 cm, Metal Rat Cage Trap, Key Industries, Auckland, New Zealand). Before the first primary session in each habitat, traps were deployed, but not opened, for approximately five days to allow rats to overcome neophobia (Yackel Adams, et al., 2011; Russell, et al., 2015). For each night in each ten-day trapping period, traps were baited with a small (1 × 1 cm) cube of fresh coconut between 1600–1800 h local time (UTC-8), and checked the following morning between 0800–1000 h.

Each captured rat was fitted with a uniquely numbered ear tag (size 1005-1; National Band & Tag Company, Newport, Kentucky, USA), and the rat was released next to the trap. We recorded the trap location for each capture, whether female rats showed signs of lactation or pregnancy, and whether traps were available to capture rats or had been de-activated (e.g., by crabs). Upon their first capture, rats were sexed by examination of external genitalia, weighed using a spring balance (± 1 g; Pesola AG, Schindellegi, Switzerland), and their body and tail lengths were measured to the nearest 1 mm using a metal ruler (Cunningham & Moors, 1996).

In November 2015, after the mark-recapture effort, we also conducted lethal snap-trapping along a subset of the locations of the live-trap locations on the plateau. This lethal trapping was designed to provide definitive age and sex classifications and body measurements for as many tagged rats as possible.

Radio-tracking

To provide an alternative estimate of movement range not dependent on the recapture of a rat, we radio-tracked rats that were captured on the plateau in July 2013 using the same small metal box traps as mentioned above. We fitted radio-collars (pipAg393, 2.6 g, Biotrack, Wareham, UK) attached to plastic collars with rubber tubing to each rat. After fitting the collar, rats were placed back in the trap and monitored for five minutes; adjustments were made to the collar if necessary before the rat was released at the site of capture. The capture location, sex, reproductive status (males with or without descended testes; females with or without a perforated vagina) and mass were recorded for all radio-tracked rats as described above.

After release, rats were located at least twice daily during daylight hours using a three-element Yagi antenna and Telonics TR-4 receivers with each radio-collar separated by frequency. Locations were either recorded by homing using a hand-held GPS device with an accuracy of <5 m or estimated through bisection by using distance and bearing from two observation points with an accuracy of ca. 25 m (Kenward, 2001).

Calculation of movement distances

We first calculated the straight-line distance between trap locations for subsequent captures of individual rats. These distances are a conservative estimate of rats' movement distances, because they assume an unrealistic direct line of travel from one trap to the next. We summed all distances between subsequent captures and divided the total travel distance calculated for each individual by the number of captures to provide an overall estimate of mean distance moved between two capture events that is not dependent on the number of captures (Püttker, et al., 2012). We also calculated the observed range length, defined as the maximum distance between any two capture locations for a given individual (Stickel, 1954; Lindsey, et al., 1973). We present results as median ± standard deviation and range.

Analysis of home range size and survival

To estimate rat survival while taking movements into account, we used a spatially-explicit Cormack-Jolly-Seber (CJS) model adapted from similar models (Gardner, et al., 2010; Raabe, et al., 2013; Royle, et al., 2016). We considered each primary session as a capture occasion and reduced binomial capture data from trapping nights to counts of each individual at each trap location during a given primary session because robust-design formulations of the spatial CJS model (Ergon & Gardner, 2014) did not converge. We removed all rats that were captured only once from the analysis, because these transients do not provide any information on movement or survival probability (Pradel, et al., 1997), and we draw no inferences from estimated capture probabilities. We also implemented a non-spatial CJS survival model following Russell & Ruffino (2012), to compare to the spatial model. This model yielded similar mean estimates and temporal variation in survival, suggesting the spatial model results are valid, but with much greater precision by not incorporating the large variance in rat movements (ESM Fig. S1). Understanding and incorporating rat movements is critical for distinguishing survival from movement in apparent survival models (Gilroy, et al., 2012; Schaub & Royle, 2014), especially for inferring potential factors in eradication failure, and we therefore present only the results of the spatial CJS model.

Our spatial CJS model assumed that rat home ranges were circular, but that the estimated centre of a rat's home range could vary spatially based on an individual-specific correlated random walk parameter (Royle, et al., 2016), which effectively allowed rats to shift their activity centre over time. We also assumed that capture probability of rats at a given trap followed a negative exponential function based on the distance of the rat's home range centre to the trap (Ergon & Gardner, 2014; Royle, et al., 2016), and that the shape of this capture probability function varied over time and among individuals. Because exploratory analysis of rat movements indicated that neither individual nor environmental covariates could adequately capture the variation in rat movement, we assumed that the shape of the capture probability function originated from two different statistical distributions: one distribution reflected 'territorial' rats and was specified as a normal distribution with a mean of $\sigma = 30$, which corresponds to a typical home range radius for insular rats (Bramley, 2014b; Ringler, et al., 2014; Harper, et al., 2015). The other distribution reflected 'roaming' rats with a uniform distribution between $\sigma = 60$ – 400, allowing a movement radius of 1,000 m, which has been recorded for Pacific rats in other studies (Wirtz, 1972; Lindsey, et al., 1973). For each individual rat, we allowed the model to select the home range radius parameter belonging to either the 'territorial' or 'roaming' movement type, and we report the proportion of males and females that were estimated to belong to each type.

We estimated rats' survival probability between primary sessions and assumed that survival varied over time. Because the interval among primary sessions was not constant, we calculated the interval as the time difference between the mid-point of subsequent primary sessions (range: 17–25 days) and converted survival probabilities to monthly survival probabilities to allow a direct comparison among different primary sessions. In a CJS model the probabilities of capture and survival are confounded for the last trapping occasion; to allow inference on survival probability up to our last live-trapping occasion in October 2015, we included data from a final additional session of kill trapping in November 2015 in the model (sensu Nathan, et al., 2015), and allowed for a different capture probability for that trapping period. Because rat survival may vary by sex and may depend on food availability (Russell & Ruffino, 2012; Ringler, et al., 2014), we included individual sex and the Normalised Difference Vegetation Index (NDVI) as covariates affecting survival probability. NDVI is a measure of vegetation 'greenness' derived from remote sensing imagery and can serve as a useful proxy for rat food availability (Pettorelli, et al., 2011; Pettorelli, et al., 2014). We downloaded NDVI for Henderson Island at a 250 m resolution from NASA Earth Data (https://daacmodis.ornl.gov/cgi-bin/MODIS/GLBVIZ_1_Glb/modis_subset_order_global_col5.pl), and averaged NDVI over 32 days centred on the mid-point of each survival period to reflect the food availability for rats during the period over which survival was estimated. We used diffuse priors for covariate effects on survival, but used informative priors for daily survival probabilities that were based on previous studies (Tamarin & Malecha, 1971; Moller & Craig, 1987; Roberts & Craig, 1990). Time-specific priors for daily survival probability were drawn from a random uniform distribution between 0.9 and 1.

We fitted the robust design CJS model in JAGS v 3.4.0 (Plummer, 2012) using the 'jagsUI' package (Kellner, 2016) called from R 3.2.5 (R Core Team, 2016). We ran three Markov chains each with 30,000 iterations, discarded the first 7,000 iterations as adaptation and burn-in, and tested for convergence using the Gelman-Rubin diagnostic (Brooks & Gelman, 1998) as well as visual representations of all parameters of interest. We report posterior mean estimates and 95% credible intervals for survival probability and the spatial shift of home range centres among primary capture sessions. Code to repeat the analysis can be downloaded from: https://github.com/steffenoppel/henderson/blob/master/Oppel_etal_SECR_ANALYSIS_and_DATA.zip.

To estimate a 'home range radius' from the shape of the spatial detection function, we assumed a circular exponential distribution for individual home ranges, and calculated an approximation of the home range radius that would encompass 95% of an individual's territory using the function 'circular.r' in R package secr 2.10.2 (Efford, 2016). We converted this estimate of home range radius to an estimate of home range size using standard geometry ($A = \pi r^2$). This estimate of space use, although not equivalent to a home range estimate obtained from telemetry, allowed us to compare the space use inferred from our spatial trapping approach to a similar metric estimated from radio-tracking to compare the conclusions from each approach.

To provide a comparable estimate of home range size from radio-tracking data, we first calculated the minimum convex polygon (MCP) for each tracked animal and then calculated the 95% kernel utilization distribution using the 'kernelUD' function of the 'adehabitatHR' package in R (Calenge, 2006) for all rats with >10 position fixes after capture. We parameterized our kernel density estimation model using a grid size of 1000, and a smoothing parameter of h = 10 m to avoid overestimation of home ranges due to large kernels around single locations.

Adequacy of cereal bait distribution during eradication attempt

To assess how many bait pellets would have been available to rats, we calculated the approximate number of bait pellets that would have been available in minimum home range sizes of rats during the eradication operation in 2011 based on mean bait application rates. In 2011, bait was distributed at 40–60 kg/ha in the embayment forest and 10 kg/ha on the plateau during the first of two bait applications. Given that a bait pellet weighs ca. 1.8 g, there were between 22,000 and 33,000 pellets/ha available in the embayment forest, and 5,500 pellets/ha on the plateau. For each of the home range estimates from radio-tracking and spatial re-capture, we multiplied the estimated size of the minimum home range area by the density of pellets to infer how many bait pellets would have been accessible to individual rats.

RESULTS

Rat movement

We captured and marked a total of 810 rats, of which 580 were recaptured at least once, yielding a total of 4,920 capture events at 396 unique trap locations. On the plateau, we captured 727 individuals of which 524 were recaptured at least once; in the embayment forest we captured 86 individuals of which 56 were recaptured at least once; only three individuals were captured in both habitats.

The median movement distance between subsequent captures was 17 ± 19 m (range: 0–153 m) in the embayment forest and 23 ± 70 m (0–970 m) on the plateau (Table 1). The median maximum distance between subsequent capture locations averaged across all individuals was 31 ± 23 m in the embayment forest and 54 ± 105 m on the plateau. The observed range length was 39 ± 25 m (0–107 m) in the embayment forest and 61 ± 127 m (0–1,023 m) on the plateau. The total minimum movement distance of individuals summed across all their capture events was 83 ± 100 m (range: 0–387 m) in the embayment forest and 140 ± 617 m (0–8,022 m) on the plateau; however, due to the unequal trapping effort in both time and space these basic movement distances are not directly comparable between the two habitats. Males showed generally longer and more variable movements than females in both habitats, but this effect was more pronounced on the plateau where much longer movements could be recorded by the larger trap network (Table 1). There was very little difference among females that were recorded with or without signs of current reproduction (Table 1). Of the rats recaptured at least once, 8.4% were only captured in one trap location. With the exception of one lactating female which was captured nine times in the same trap location, all rats that were captured >5 times moved between at least two different trap locations.

Table 1 Median and standard deviation (sd) straight-line movement distances (m) and observed range lengths of Pacific rats between live capture events during a spatial mark–recapture study on Henderson Island in May–October 2015. Note that the trapping effort in the two habitats had a different spatial and temporal extent (see Fig. 1 for spatial extent of trap locations). 'breed' females were classified if they had obvious signs of lactation or pregnancy.

Parameter	Embayment forest						Coral plateau					
	males		non-breed females		breed females		males		non-breed females		breed females	
	median	sd	median	sd	median	sd	median	sd	median	sd	median	sd
n individuals	32		13		20		262		201		171	
n captures	171		49		77		2010		1195		608	
mean distance between subsequent captures (m)	17.5	17.9	20.7	19.6	13.6	21.5	27.2	80.4	20.9	53.0	21.6	57.0
maximum distance between subsequent captures (m)	36.7	23.3	31.7	23.2	18.4	21.9	60.7	117.6	49.8	91.1	46.3	85.8
observed range length (m)	43.3	25.2	35.4	23.2	18.4	21.9	70.4	144.9	55.5	106.2	46.3	93.0
total minimum distance travelled (m)	93.6	105.1	108.4	102.2	23.7	56.7	172.0	793.2	115.7	367.4	85.0	204.6

Seasonal variation in survival and space use

Based on the capture and recapture of 540 individual rats on the plateau (including recapture in snap traps in November), we found seasonal variation in monthly survival probability (Fig. 2), but no evidence that survival was influenced by sex (β = -0.15; 95% credible interval -0.43–0.12) or NDVI (β = 0.44; -0.87–1.73). In June and early July, the median monthly survival probabilities of Pacific rats on the plateau were 0.794 (0.306–0.967) and 0.781 (0.471–0.933), respectively, but dropped to 0.353 (0.081–0.737) and 0.636 (0.488–0.763) in late July and early August, respectively (Fig. 2). Remaining survivors had very high survival in late August (0.950; 0.846–0.986) and September (Fig. 2), despite persisting low NDVI (Fig. S2). Similar estimates were obtained from 60 individual rats in the embayment forest, with median monthly survival probabilities of 0.361 (0.054–0.907) in early August and 0.864 (0.466–0.995) in September.

The survival estimates had very low precision due to the potential for confounding emigration, because during the times of lower mean survival probability, a larger number of rats appeared to exhibit longer movements. Rat movements were captured by two frequency distributions (Fig. 3), with the majority of rats (79.1%) belonging to a ‘territorial’ type that exhibited home range radii between 100 and 200 m, and a smaller proportion (19.9% of males, 22.0% of females) belonging to a ‘roaming’ type with highly variable and occasionally very long-distance movements (Fig. 3). The proportion of captured rats belonging to the roaming type increased from 0.8% in June to 13.8% in late July (Table 2). In the embayment forest, we estimated only marginally smaller home range radii as on the plateau in early August (Table 2).

Besides large movements around a central point in their territory, our model also indicated that, for rats that were captured in two subsequent primary sessions, the central point of their activity shifted by a median of 50 m (5–290 m) between early and late August, and by a median of 92 m (4–378 m) between September and October (Fig. 4).

Home range sizes estimated from telemetry

In 2013, we successfully tracked 19 rats (9 females, 10 males) between 1 July and 24 August with body mass ranging from 29 to 107 g (median: 71 g, SD: 32 g). The median 50% utilization distribution (the core home range) was 0.095 ± 0.08 ha (range 0.05–0.30 ha), and the 95% utilization distribution (UD) was 0.55 ± 0.37 ha (range 0.21–1.58 ha). The minimum convex polygon home range was more variable with a median of 0.36 ± 0.86 ha (range 0.003–2.99 ha). Rats used vegetation in the canopy or sub-canopy during less than 20% of re-locations. There was no relationship between the number of days a rat was tracked (range: 7–54 days) and the size of its home range (MCP: p = 0.11; 95% UD: p = 0.31). Thus, the estimates derived from radio-tracking suggested much smaller rat home range areas than those derived from spatially-explicit mark-recapture models, which ranged from 2.88 to 931.6 ha for territorial rats on the plateau, and from 0.11 to 53.6 ha in the embayment forest, assuming that these rats used a circular home range.

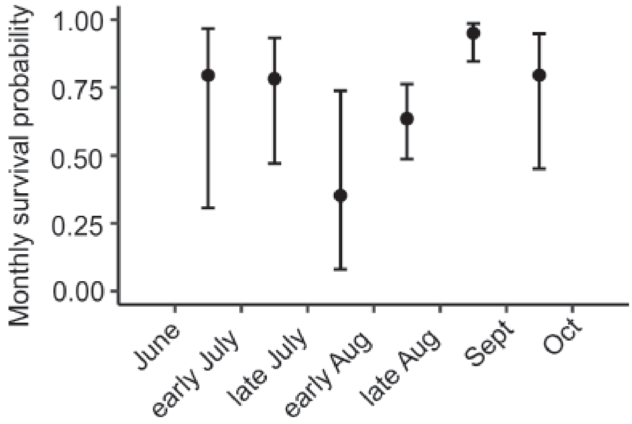

Fig. 2 Mean (95% credible interval) monthly survival probability of Pacific rats on Henderson Island between seven primary trapping sessions over five months in 2015 estimated from a spatial Cormack-Jolly-Seber model. Note that survival probability is scaled over a 30-day period due to unequal time intervals between primary trapping sessions.

Adequacy of cereal bait distribution during the 2011 eradication attempt

The lowest confidence limit for an estimated home range for any season based on our spatial capture data was 2.88 ha on the plateau and 0.11 ha in the embayment forest.

Table 2 Home range radius (m) of two different behavioural types of Pacific rats on the coral plateau and in the embayment forest of Henderson Island between June and October 2015, estimated from a spatial mark–recapture model. Median estimated home range radius and lower (lcl) and upper (ucl) 95% credible limits are given in m. ‘prop’ indicates the proportion of captured rats in a 10-day trapping session that belonged to one of the behavioural types. Roaming rats could not be detected in the embayment forest.

		Residential rats				Roaming rats			
		prop	median	lcl	ucl	prop	median	lcl	ucl
Plateau	June	0.99	135	107	162	0.01	399	290	584
	early July	0.91	132	103	161	0.09	776	223	1,659
	late July	0.86	133	104	162	0.14	866	279	1,725
	early Aug	0.89	135	107	165	0.11	1,038	307	1,767
	late Aug	0.93	138	110	167	0.07	1,229	619	1,774
	Sept	0.88	137	110	171	0.12	688	150	1,579
	Oct	0.77	132	102	171	0.23	724	293	1,568
Embayment forest	early Aug		96	37	228				
	late Aug		137	36	377				
	Sept		142	34	382				

Home ranges of this size would result in 15,988 toxic bait pellets being available within a rat's home range on the plateau, and 2,456 in the embayment forest. Based on radio-tracking, where the smallest 95% UD was 0.21 ha, 1,175 pellets would have been available in a rat's home range on the plateau, and 4,700 pellets in the embayment forest.

DISCUSSION

We demonstrated that invasive Pacific rats on Henderson Island exhibited substantial individual and temporal variation in their movement and survival over a five-month period. We found no evidence to suggest that rats had home ranges that would have limited their ability to encounter bait if bait was distributed with a density similar to the 2011 eradication attempt. Indeed, the movements and home range estimates that we obtained were considerably higher than those of any other published study on the same species (Table 3), including populations that have been eradicated (Bramley, 2014b). The timing of the failed eradication operation in mid/late August 2011 also appears to have been at a time of the year where we recorded naturally low survival in 2015, and the seasonal timing of the operation was likely appropriate if conditions in 2011 followed a similar phenology as in 2015 (Fig. S2).

Monthly survival probability of Pacific rats varies between 0.40 and 0.72 (Tamarin & Malecha, 1971; Moller & Craig, 1987; Bunn & Craig, 1989), with an expected life span around 8–10 months (Harrison, 1956;

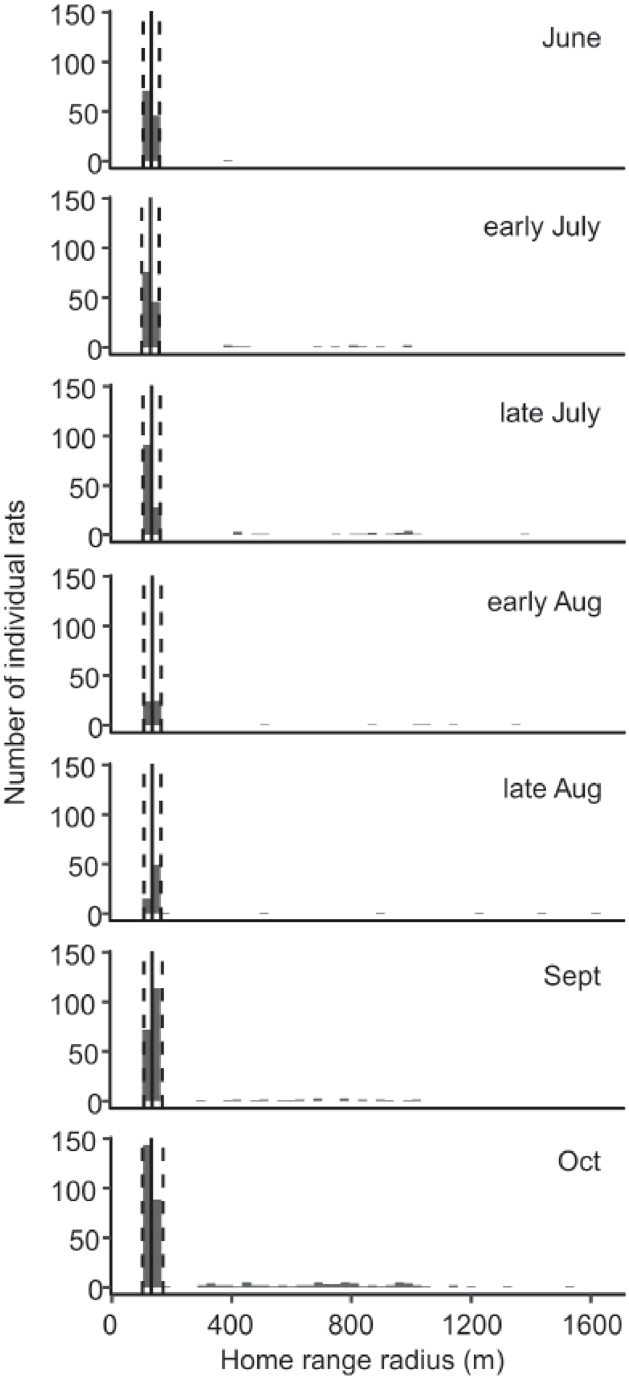

Fig. 3 Histogram of the number of individual Pacific rats having a home range of a radius estimated from a spatial Cormack-Jolly-Seber based on mark–recapture data from the coral plateau on Henderson Island during seven primary trapping sessions in 2015. Vertical lines indicate the population mean (solid) and 95% credible interval (dashed) home range radius.

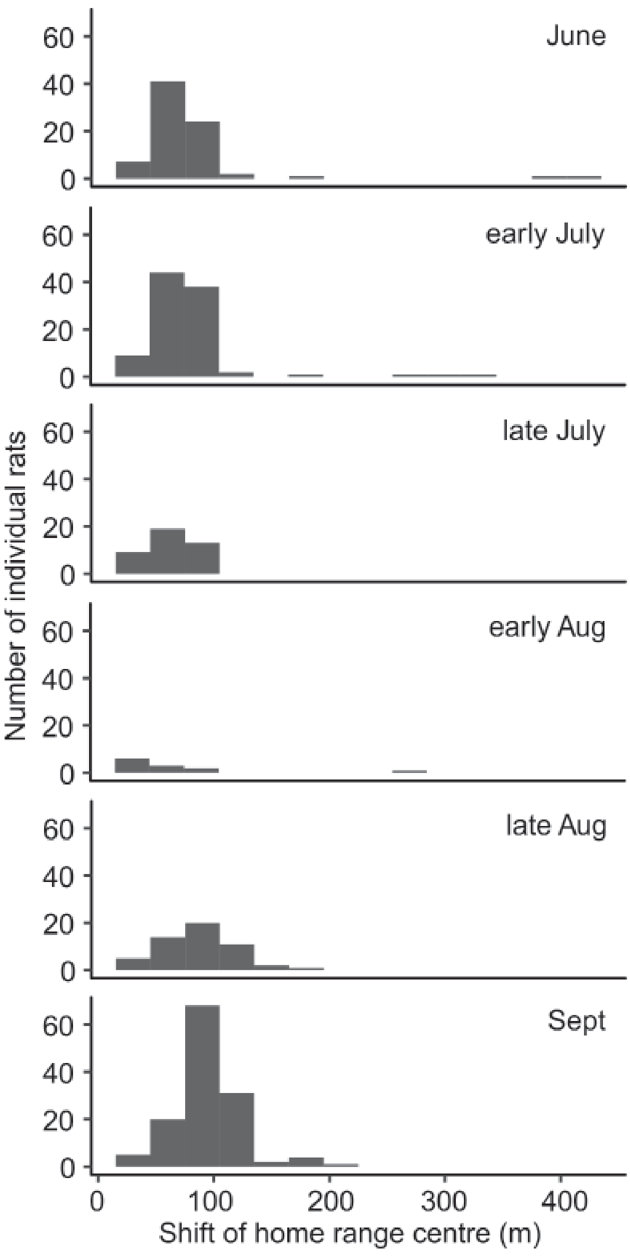

Fig. 4 Frequency of displacement distances of activity centres of male and female Pacific rats on the interior coral plateau of Henderson Island between seven primary trapping sessions over five months in 2015 estimated from a spatial Cormack-Jolly-Seber model.

Bourliere, 1959). We estimated broadly similar median monthly survival probabilities of 0.36–0.90 on Henderson Island. However, previous estimates were mostly based on raw recapture rates and did not account for recapture probabilities, and our slightly higher estimates of survival in June, early July, and late August may be due to our model accounting for low recapture probability. A higher survival probability of Pacific rats on Henderson Island might also be expected given the absence of larger competitors (*R. rattus* or *R. norvegicus*).

There was temporal fluctuation in survival probability of Pacific rats in other tropical (Tamarin & Malecha, 1971) and temperate island populations (Moller & Craig, 1987; Bunn & Craig, 1989), and we found similar short-term variability in survival on Henderson Island. We currently do not understand what may have caused the temporal decline in survival probabilities in July and August, and whether such a reduction occurs predictably every year in response to regular environmental events. As a sub-tropical island, Henderson Island experiences only moderate fluctuations in temperature and day length, which are unlikely to lead to the same predictable population fluctuations as observed on temperate islands (Russell & Holmes, 2015). The changes in both survival and movement within our five-month study period on Henderson may have reflected a period of resource shortage from late July to September that may have induced higher mortality and emigration as a larger proportion of rats belonged to the 'roaming' movement type. Assuming that the reduced survival that we observed in 2015 was caused by resource limitation (e.g. Russell & Ruffino, 2012), and that fluctuations in resource availability and survival are similar among years (Fig. S2), an operational timing in July or early August may maximise the chances of eradication success.

Our spatial mark-recapture data on the plateau, where traps were up to 1.5 km apart, revealed many long movements by rats. These movements matched or exceeded the previously estimated maximum travel distance of 1,097 m or home range estimate of 3 ha for Pacific rats (Lindsey, et al., 1973; Nass, 1977; Lindsey, et al., 1999; Clapperton, 2006; Scheffler, et al., 2012), and were similar to movements typically found in the much larger Norway rat (*R. norvegicus*) (Clapperton, 2006; Bramley, 2014b). Despite some long movements that we recorded, the extrapolated 'home range areas' from our spatial capture data are possibly biased high, because these extrapolations are based on the assumption that rats occupy a circular home range, which may not be the case (Nass, 1977; Lindsey, et al., 1999; Clapperton, 2006). In particular, our trails may have affected rat movement by providing highly nutritious and palatable coconut bait in traps that is otherwise not available on the plateau. However, our trails were characterised by an absence of vegetation between 30 to 250 cm above ground, and probably did not materially affect the movement ability of rats on the ground. Nonetheless, the maximum estimates of home range area that we provide must be considered with caution, as the areas actually exploited by rats may be significantly smaller than the assumed circular radius range.

Based on our estimates of movement behaviour from radio-tracking in 2013 and spatial mark-recapture in 2015, individual rats would have theoretically encountered hundreds to thousands of bait pellets in their typical home range, which would likely be sufficient for them to ingest a lethal dose even if crab consumption gradually reduced bait density over time (Cuthbert, et al., 2012). We therefore consider it unlikely that the eradication failed because individual rats did not have access to a sufficient quantity of toxic bait, but uncertainty remains with respect to certain life stages (e.g. nursing female rats and freshly weaned pups): the number of rats surviving the 2011 operation was very small, constituting <0.2% of the estimated rat population (Amos, et al., 2016). An eradication operation may fail if only a very small number of rats exhibit no movement and would therefore not encounter a sufficient quantity of bait. Of the 810 rats that we captured in 2015, 28% were never recaptured, and of those that were

Table 3 Summary of home range size (ha) estimates of Pacific rats (*Rattus exulans*) on subtropical and tropical islands derived from either radio tracking (TR) or spatial capture–mark–recapture (CMR); type of estimate refers to minimum convex polygon (MCP) or spatially-explicit capture recapture (SECR) and indicates what measure of uncertainty (standard deviation, SD; range) is provided with the estimate.

Location	Tracking method	Sex	*n*	Home range (ha)	Type of estimate	Reference
Hilo, HI, USA	TR	F	28	0.06 (0.01–0.18)	MCP (range)	(Nass, 1977)
Green Island, Kure Atoll, HI, USA	CMR	F	40	0.08 (0.01–0.48)	Mean minimum (range)	(Wirtz, 1972)
Kapiti Island, NZ	TR	M	6	0.14 ± 0.04	MCP (mean ± SD)	(Bramley, 2014b)
Green Island, Kure Atoll, HI, USA	CMR	M	19-40	0.17 (0.01–0.73)	Mean minimum (range)	(Wirtz, 1972)
Hilo, HI, USA	TR	M	29	0.18 (0.01-1.21)	MCP (range)	(Nass, 1977)
Kapiti Island, NZ	TR	F	5	0.18 ± 0.05	MCP (mean ± SD)	(Bramley, 2014b)
Henderson Island, Pitcairn Islands	TR	F+M	19	0.32 ± 0.38	MCP (mean ± SD)	This study
Henderson Island, Pitcairn Islands	CMR	F+M	541	0.11 –931.6	SECR (range)	This study
Hilo, HI, USA	TR	F+M	26	1.73	Circle with radius mean distance from burrow	(Lindsey, et al., 1973)
Hilo, HI, USA	TR	F+M	3	3	MCP (mean ± SD)	(Lindsey, et al., 1999)
Kahanahaiki, HI, USA	TR	Unk	1	1.8	95% kernel	(Shiels, 2010)

recaptured at least once, 8% were only captured in a single location. Because we did not record any movement for a greater proportion of rats than the estimated surviving population in 2011, it is theoretically possible that there are some individuals that move very little or move very little for a short period of time during which bait is available on the ground. Unfortunately, the probability of detecting a non-moving phenotype that exists with a prevalence of <0.2% in the population is virtually zero for any practically feasible sample size.

In summary, the rat eradication attempt on Henderson Island in 2011 failed to kill all individuals, and our work provides new knowledge to evaluate the potential causes of this failure. An eradication failure can occur if (i) not all individuals had access to sufficient bait; (ii) not all individuals died despite consuming bait; or (iii) not all individuals consumed a lethal dose of bait despite having access (Holmes, et al., 2015). We have shown that the timing of the operation was appropriate and that it is unlikely that rats did not have access to sufficient bait. Previous work confirmed that rats remain susceptible to brodifacoum, suggesting that toxicological resistance is an unlikely cause of the 2011 eradication failure (Amos, et al., 2016). A combination of factors leading to high alternative food availability and a small number of rats preferring natural food sources and disregarding bait may have resulted in the failure of the eradication attempt in 2011, and further research is required to examine whether that risk can be reduced for a new eradication attempt.

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REFERENCES

Amos, W., Nichols, H.J., Churchyard, T. and Brooke, M. de L. (2016). 'Rat eradication comes within a whisker! A case study from the South Pacific'. *Royal Society Open Science* 3: 160110.

Atkinson, I.A.E. (1985). 'The Spread of Commensal Species of *Rattus* to Oceanic Islands and their Effects on Island Avifaunas.'. In*:* P.J. Moors (ed.) *Conservation of Island Birds.* pp. 35–81. Cambridge, United Kingdom: International Council for Bird Preservation.

Bond, A.L., Churchyard, T., Cuthbert, R.J., Donaldson, A., Duffield, N., Forrest, A., Havery, S., Lavers, J.L., MacKinnon, L., McClelland, G.T.W., Kelly, J., Proud, T., Skinner, A., Torr, N., Vickery, J.A. and Oppel, S. (2019). 'Recovery of introduced Pacific rats (*Rattus exulans*) following a failed eradication attempt on Subtropical Henderson Island, South Pacific Ocean'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 167–174. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Bourliere, F. (1959). 'Lifespans of Mammalian and Bird Populations in Nature'. In*:* G.E.W. Wolstenholme and M. O'Connor (eds.) *The Lifespan of Animals*, pp. 90–102. London: J & A Churchill.

Bramley, G.N. (2014a). 'Habitat use by kiore (*Rattus exulans*) and Norway rats (*R. norvegicus*) on Kapiti Island, New Zealand'. *New Zealand Journal of Ecology* 38: 64–75.

Bramley, G.N. (2014b). 'Home ranges and interactions of kiore (*Rattus exulans*) and Norway rats (*R. norvegicus*) on Kapiti Island, New Zealand'. *New Zealand Journal of Ecology* 38: 328–334.

Brooke, M. de L. and Towns, D. (2008). *A Feasibility Study for the Eeradication of Kiore Rattus exulans from Henderson Island.* Royal Society for the Protection of Birds, Sandy, UK.

Brooke, M. de L., O'Connell, T.C., Wingate, D., Madeiros, J., Hilton, G.M. and Ratcliffe, N. (2010). 'Potential for rat predation to cause decline of the globally threatened Henderson petrel *Pterodroma atrata*: Evidence from the field, stable isotopes and population modelling'. *Endangered Species Research* 11: 47–59.

Brooks, S.P. and Gelman, A. (1998). 'General methods for monitoring convergence of iterative simulations'. *Journal of Computational and Graphical Statistics* 7: 434–455.

Broome, K., Cox, A., Golding, C., Cromarty, P., Bell, P. and McClelland, P. (2014). *Rat Eradication Using Aerial Baiting. Current Agreed Best Practice Used in New Zealand.* Internal document DOC-29396. New Zealand Department of Conservation, Wellington, New Zealand.

Bunn, T.J. and Craig, J.L. (1989). 'Population cycles of *Rattus exulans*: Population changes, diet, and food availability'. *New Zealand Journal of Zoology* 16: 409–418.

Calenge, C. (2006). 'The package "adehabitat" for the R software: A tool for the analysis of space and habitat use by animals'. *Ecological Modelling* 197: 516–519.

Clapperton, B.K. (2006). *A Review of the Current Knowledge of Rodent Behaviour in Relation to Control Devices.* Wellington, NZ: Department of Conservation.

Cromarty, P.L., Broome, K.G., Cox, A., Empson, R.A., Hutchinson, W.M. and McFadden, I. (2002). 'Eradication planning for invasive alien animal species on islands – the approach developed by the New Zealand Department of Conservation'. In*:* C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 85–91. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Cunningham, D.M. and Moors, P.J. (1996). *Guide to the Identification and Collection of New Zealand Rodents, 2nd edition,* Wellington, NZ: Department of Conservation.

Cuthbert, R.J., Brooke, M. de L. and Torr, N. (2012). 'Overcoming hermit-crab interference during rodent-baiting operations: A case study from Henderson Island, South Pacific'. *Wildlife Research* 39: 70–77.

Dawson, J., Oppel, S., Cuthbert, R.J., Holmes, N., Bird, J.P., Butchart, S.H.M., Spatz, D.R. and Tershy, B. (2015). 'Prioritizing islands for the eradication of invasive vertebrates in the United Kingdom overseas territories'. *Conservation Biology* 29: 143–153.

Efford, M. (2016). 'secr – Spatially Explicit Capture–recapture in R. R Package Version 2.10.4'. <https://CRAN.R-project.org/package=secr>.

Ergon, T. and Gardner, B. (2014). 'Separating mortality and emigration: Modelling space use, dispersal and survival with robust-design spatial capture–recapture data'. *Methods in Ecology and Evolution* 5: 1327–1336.

Gardner, B., Reppucci, J., Lucherini, M. and Royle, J.A. (2010). 'Spatially explicit inference for open populations: Estimating demographic parameters from camera-trap studies'. *Ecology* 91: 3376–3383.

Gilroy, J.J., Virzi, T., Boulton, R.L. and Lockwood, J.L. (2012). 'A new approach to the "apparent survival" problem: Estimating true survival rates from mark–recapture studies'. *Ecology* 93: 1509–1516.

Harper, G.A., van Dinther, M., Russell, J.C. and Bunbury, N. (2015). 'The response of black rats (*Rattus rattus*) to evergreen and seasonally arid habitats: Informing eradication planning on a tropical island'. *Biological Conservation* 185: 66–74.

Harrison, J. (1956). 'Survival rates of Malayan rats'. *Bulletin of the Raffles Museum* 27: 5–26.

Holmes, N.D., Griffiths, R., Pott, M., Alifano, A., Will, D., Wegmann, A. and Russell, J.C. (2015). 'Factors associated with rodent eradication failure'. *Biological Conservation* 185: 8–16.

Howald, G., Donlan, C.J., Galvan, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y.W., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Jones, H.P. (2010). 'Seabird islands take mere decades to recover following rat eradication'. *Ecological Applications* 20: 2075–2080.

Jones, H.P., Holmes, N.D., Butchart, S.H.M., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P.E., Cuthbert, R.J., Ebbert, S., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R. and Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences* 113: 4033–4038.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

Kellner, K.F. (2016). 'jagsUI: A Wrapper Around rjags to Streamline JAGS Analyses. R Package Version 1.4.2'. <https://CRAN.R-project.org/package=jagsUI>.

Kenward, R. (2001). *A Manual for Wildlife Radio Tagging,* London: Academic Press.

Lindsey, G.D., Nass, R.D., Hood, G.A. and Hirata, D.N. (1973). 'Movement patterns of Polynesian rats (*Rattus exulans*) in sugarcane'. *Pacific Science* 27: 239–246.

Lindsey, G.D., Mosher, S.M., Fancy, S.G. and Smucker, T.D. (1999). 'Population structure and movements of introduced rats in an Hawaiian rainforest'. *Pacific Conservation Biology* 5: 94–102.

Moller, H. and Craig, J.L. (1987). 'The population ecology of *Rattus exulans* on Tiritiri Matangi Island, and a model of comparative population dynamics in New Zealand'. *New Zealand Journal of Zoology* 14: 305–328.

Nass, R.D. (1977). 'Movements and home ranges of Polynesian rats in Hawaiian sugarcane'. *Pacific Science* 31: 135–142.

Nathan, H.W., Clout, M.N., MacKay, J.W.B., Murphy, E.C. and Russell, J.C. (2015). 'Experimental island invasion of house mice'. *Population Ecology* 57: 363–371.

Paulay, G. and Spencer, T. (1989). 'Vegetation of Henderson Island'. *Atoll Research Bulletin* 328: 1–20.

Pettorelli, N., Ryan, S., Mueller, T., Bunnefeld, N., Jędrzejewska, B., Lima, M. and Kausrud, K. (2011). ‚The normalized difference vegetation index (NDVI): unforeseen successes in animal ecology'. *Climate Research* 46: 15–27.

Pettorelli, N., Laurance, W.F., O'Brien, T.G., Wegmann, M., Nagendra, H. and Turner, W. (2014). 'Satellite remote sensing for applied ecologists: opportunities and challenges'. *Journal of Applied Ecology* 51: 839–848.

Plummer, M. (2012). ‚JAGS Version 3.3.0'. <http://sourceforge.net/projects/mcmc-jags/files/Manuals/>.

Pradel, R., Hines, J.E., Lebreton, J.D. and Nichols, J.D. (1997). 'Capture–recapture survival models taking account of transients'. *Biometrics* 53: 60–72.

Püttker, T., de Barros, C.d.S., Martins, T.K., Sommer, S. and Pardini, R. (2012). 'Suitability of distance metrics as indexes of home-range size in tropical rodent species'. *Journal of Mammalogy* 93: 115–123.

R Core Team (2016). *R: A Language and Environment for Statistical Computing*. Vienna, Austria: R Foundation for Statistical Computing.

Raabe, J.K., Gardner, B. and Hightower, J.E. (2013). 'A spatial capture–recapture model to estimate fish survival and location from linear continuous monitoring arrays'. *Canadian Journal of Fisheries and Aquatic Sciences* 71: 120–130.

Ringler, D., Russell, J., Jaeger, A., Pinet, P., Bastien, M. and Le Corre, M. (2014). 'Invasive rat space use on tropical islands: Implications for bait broadcast'. *Basic and Applied Ecology* 15: 179–186.

Roberts, M. and Craig, J. (1990). 'The demography of kiore, *Rattus exulans* in three habitats'. *New Zealand Journal of Zoology* 17: 43–53.

Royle, J.A., Fuller, A.K. and Sutherland, C. (2016). 'Spatial capture-recapture models allowing Markovian transience or dispersal'. *Population Ecology* 58: 53–62.

Russell, J.C. and Ruffino, L. (2012). 'The influence of spatio-temporal resource fluctuations on insular rat population dynamics'. *Proceedings of the Royal Society B: Biological Sciences* 279: 767–774.

Russell, J.C., Caut, S., Anderson, S.H. and Lee, M. (2015). 'Invasive rat interactions and over-invasion on a coral atoll'. *Biological Conservation* 185: 59–65.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Schaub, M. and Royle, J.A. (2014). 'Estimating true instead of apparent survival using spatial Cormack–Jolly–Seber models'. *Methods in Ecology and Evolution* 5: 1316–1326.

Scheffler, P.Y., Foote, D., Forbes Perry, C., Schlappa, K. and Stone, C.P. (2012). *Population Dynamics of Introduced Rodents in Hawai'i Volcanoes National Park 1986–1990.* Honolulu, HI: Pacific Cooperative Studies Unit, University of Hawai'i.

Shiels, A.B. (2010). 'Ecology and Impacts of Introduced Rodents (*Rattus* spp. and *Mus musculus*) in the Hawaiian Islands'. PhD thesis. Honolulu, HI: University of Hawai'i.

Spencer, T. (1995). 'The Pitcairn Islands, south Pacific Ocean: Plate tectonic and climatic contexts'. *Biological Journal of the Linnean Society* 56: 13–42.

Steadman, D.W. and Olson, S.L. (1985). 'Bird remains from an archaeological site on Henderson Island, South Pacific: Man-caused extinctions on an "uninhabited" island'. *Proceedings of the National Academy of Sciences* 82: 6191–6195.

Stickel, L.F. (1954). 'A comparison of certain methods of measuring ranges of small mammals'. *Journal of Mammalogy* 35: 1–15.

Tamarin, R.H. and Malecha, S.R. (1971). 'The population biology of Hawaiian rodents: Demographic parameters'. *Ecology* 52: 383–394.

Torr, N. and Brown, D. (2012). *Henderson Island restoration project post-operational report.* Sandy, UK: Royal Society for the Protection of Birds.

Waldren, S., Florence, J. and Chepstow-Lusty, A.J. (1995). 'A comparison of the vegetation communities from the islands of the Pitcairn group'. *Biological Journal of the Linnean Society* 56: 121–144.

Weigelt, P., Jetz, W. and Kreft, H. (2013). 'Bioclimatic and physical characterization of the world's islands'. *Proceedings of the National Academy of Sciences* 110: 15307–15312.

Wirtz, W.O.I. (1972). 'Population ecology of the Polynesian rat, *Rattus exulans*, on Kure Atoll, Hawaii'. *Pacific Science* 26: 433–464.

Yackel Adams, A.A., Stanford, J.W., Wiewel, A.S. and Rodda, G.H. (2011). 'Modelling detectability of kiore (*Rattus exulans*) on Aguiguan, Mariana Islands, to inform possible eradication and monitoring efforts'. *New Zealand Journal of Ecology* 35: 145–152.

Zavaleta, E.S. (2002). 'It's often better to eradicate, but can we eradicate better?'. In*:* C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 393–404. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Assessing the critical role that land crabs play in tropical island rodent eradications and ecological restoration

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Abstract Invasive rodent eradications are one of the most effective conservation interventions to restore island ecosystems. However, achievements in the tropics are lagging behind those in temperate regions. Land crab interference in bait uptake has been identified as one of the main causes of rodent eradication failure on tropical islands, but the issue of effective mitigation of bait loss due to land crab consumption is poorly understood. For example, there are over 100 species of land crab and each may behave differently. We reviewed the available literature to answer: (1) which crab species are the most problematic? (2) what mitigation measures have been effective? and (3) how do invasive rodents impact land crab communities? We analysed a systematic dataset from six tropical islands to test two hypotheses: (a) bait uptake is highest when burrowing (Brachyura) land crabs are present; and (b) small land crabs (including juveniles of the larger species) are highly vulnerable to rodent predation. We found that large species (e.g. genera *Cardisoma, Johngarthia* and *Birgus*) are the most problematic during rodent eradications. Effective mitigation measures to prevent bait loss include using higher bait application rates and conducting eradications during the driest months. Land crab communities tend to go through significant changes after rodent removal. From our analyses, we confirmed pre-eradication data are valuable for eradication planning, as seasonality and type of crab can influence outcomes. Post-eradication data confirmed small crab species (<60 mm) are highly vulnerable to rodent predation. More effort should be invested into monitoring land crabs in tropical latitudes, particularly to determine any biogeographic or taxon trends in land crab interference. Land crabs are key for the restoration of the islands, as they shape ecosystems through their role as ecosystem engineers, hence they are excellent indicators of ecosystem recovery. Our results will contribute to the better planning of future rodent eradications on tropical islands where land crabs are significant bait competitors.

Keywords: *Birgus, Cardisoma, Coenobita, Gecarcinus*, impacts, monitoring, *Mus, Rattus*

INTRODUCTION

Islands are some of the most important repositories for biodiversity, with 15–20% of terrestrial biodiversity held on only 5.3% of the world's land area (Weigelt, et al., 2013). Tropical islands are particularly important due to their high levels of endemism (Myers, et al., 2000). Island species are also highly vulnerable to anthropogenic impacts, of which invasive alien species (IAS) introductions are often the most severe (Russell, et al., 2017), causing 86% of island endemic species extinctions (Bellard, et al., 2016). Moreover, IAS also interrupt ecosystem functioning through predation of, and competition with, other biotic components (Athens, et al., 2002; Towns, et al., 2006; Hilton & Cuthbert, 2010).

Over the past 50 years, eradication of IAS has been increasing (Towns, et al., 2013; Jones, et al., 2016), with the removal of invasive rodents proving highly effective in targeted species recovery and island ecosystem restoration (Le Corre, et al., 2015; Croll, et al., 2016). Over 90% of rat eradication attempts have been successful, with increasingly larger islands being effectively targeted (Holmes, et al., 2015). However, the rate of eradication failure on tropical islands has been 2–2.5 times higher than on temperate islands (Russell & Holmes, 2015). This discrepancy is due to several contributing factors (Holmes, et al., 2015). Probably the most significant are the benign climate facilitating rodent reproduction (Harper & Bunbury, 2015), and bait competition from abundant land crabs (Wegmann, 2008; Griffiths, et al., 2011).

Land crabs comprise over a hundred species in three broad groups, burrowing crabs, hermit crabs and coconut crabs, although the latter single species (*Birgus latro*) is technically a hermit crab. As the largest invertebrates on islands, particularly coral atolls, land crabs are often the apex land predator (Burggren & McMahon, 1988), and can attain high population densities and occupy the niches of vertebrates on small oceanic islands. As such, they act as allogenic ecosystem engineers (Green, et al., 2008; Paulay & Starmer, 2011) through their significant influence on forest structure, plant species composition, soil formation and nutrient transfer and cycling (Green, et al., 1999; Sherman, 2002; Gutiérrez & Jones, 2006; Gutiérrez, et al., 2006; Sherman, 2006; Green, et al., 2008; Lindquist, et al., 2009). As keystone consumers (Paine, 1966), the removal of or reduction in crab abundance through the introduction of IAS can trigger a trophic cascade of effects, leading to 'meltdown' in island ecosystems in the worst cases (O'Dowd, et al., 2003; Pitman, et al., 2005; Nigro, et al., 2017). Moreover, as smaller crab species in particular are vulnerable to predation by rodents (St Clair, 2011; Samaniego-Herrera & Bedolla-Guzmán, 2012) and invasive rodents are found on >80% of island groups (Atkinson, 1985), an improved understanding of the interaction between rodents and land crabs is urgently required. However, land crabs have rarely been monitored before and after rodent eradications (but see Nigro, et al., 2017), and basic tools such as inventories are lacking for most tropical islands where rodent eradications are being planned.

The Pacific Invasives Initiative (PII) commissioned the first review on land crab interference in rodent eradications about 10 years ago (Wegmann, 2008) and many lessons have been learnt since. To improve the justification and implementation of rodent eradications on tropical islands, we conducted literature reviews on two main topics: the role of land crabs in invasive rodent eradications and the vulnerability of land crabs to rodent invasion. A case study from six tropical islands is presented, demonstrating the utility of monitoring land crabs both pre- and post-rodent eradications. Based on our previous observations of land crabs across islands, we expected (a) bait uptake to be highest on the islands where large burrowing species were abundant, and (b) population abundance of small burrowing species to increase over time after rodent eradications.

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METHODS

Land crabs and rodent eradications

Following Burggren & McMahon (1988), we consider land crabs to be crabs that show significant behavioural, morphological, physiological, or biochemical adaptations permitting extended activity out of water. This includes a few families of the diverse infraorders Anomura (hermit crabs) and Brachyura (burrowing crabs), yet there are over a hundred species that can be considered land crabs. Land crab distribution ranges from tropical to subtropical areas, hence the scope of this paper focuses on islands located between ~25° north and south of the equator. We also focus on the two most common rodent eradication methods: aerial and hand broadcast of bait directly onto the ground (Howald, et al., 2007; DIISE, 2016).

The islands included in the review are a subset from the Database of Island Invasive Species Eradications (DIISE, 2016). These were selected based on the following criteria: 1) location: between latitudes ~25° north and south of the equator, 2) target IAS taxa: Muridae, 3) whole island eradications, 4) toxicant used: 2nd generation anticoagulant, 5) main bait delivery method: hand or aerial broadcast, 6) quality of data: good or satisfactory, the latter were updated to good, and 7) eradication status: known or 'to be confirmed', the latter were updated to failed or successful. Islands without land crabs such as the Galapagos Islands and Western Australia islands were excluded.

For each island, we collated the following additional data: bait rates used during the rodent eradications, island type (savanna, tropical seasonal forest or tropical rainforest), presence/absence status and abundance for each land crab group (hermit, coconut, burrowing), land crab group identified as the main bait competitor and timing of the eradication (dry or wet season). This information was collated through review of project documents (i.e. feasibility studies, operational plans, post-operation reports and scientific papers). We also sought inputs from project managers when we required further clarification/confirmation or information was missing from the documents available. Given the scarcity of scholarly information on land crabs, and the lack of a single source with the basic biology and current taxonomy for all land crabs (as most crab species are marine), we conducted an additional literature review to compile such information.

A 2-way ANOVA test for unbalanced designs was used to evaluate the variations in bait rates in relation to island type and main bait competitors. Data were log-transformed to achieve normality. All analyses were performed in R 3.4.

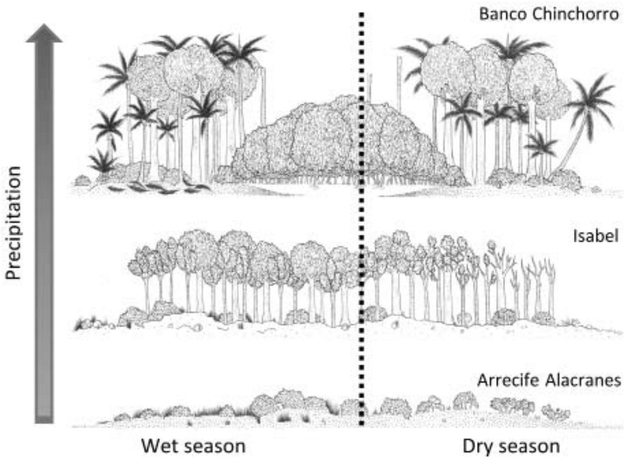

Fig. 1 Dominant vegetation on the islands where the case study took place.

Rodent impacts on land crabs

Some of the information on the impacts of rodents on land crabs was collated from the project documents mentioned above. In addition, we also searched the Web of Science, Scopus and Google Scholar for published literature using keywords: [island OR atoll OR cay OR archipelago] AND [rodent OR rat OR rattus OR mus] AND ["land crab" OR invertebrate]. We collated information on impacts through review of the resulting publications and relevant references listed in these.

Case study: Mexican tropical islands

Study sites

As part of a wider restoration programme led by Grupo de Ecología y Conservación de Islas (GECI) (Samaniego-Herrera, et al., 2011), bait uptake and land crab monitoring was conducted on six Mexican tropical islands. The islands, three in the Gulf of Mexico, one in the Mexican Pacific, and two in the Caribbean Sea fall into the three categories of tropical island ecosystems described by Russell and Holmes (2015): savanna, tropical seasonal forest and tropical rainforest, respectively (Table 1; Fig. 1). The aims of the monitoring were to inform the specific rodent eradication plans by assessing the potential interference of each land crab community, and to compare such communities before and after the removal of the invasive rodents. Invasive rodents (Table 1) were successfully eradicated from all islands either by hand or aerial broadcast of bait (Samaniego-Herrera, et al., 2014; Samaniego-Herrera, et al., 2018), following international best practices (Keitt, et al., 2015).

Bait uptake

Two types of bait were used: placebo bait for pre-eradication assessments and toxic bait for the actual rodent eradications. The toxic bait consisted of 2 g cereal bait pellets containing 25 ppm brodifacoum (second generation anticoagulant), manufactured by Bell Labs. The placebo bait, also from Bell Labs, was identical but non-toxic. Total bait uptake (i.e. by the target and non-target species) was measured before and during each eradication operation. Pre-eradication, bait uptake was monitored to help decide application rates for the eradication. During eradications, the monitoring took place to (a) validate the intended bait rate, by estimating bait density on the ground immediately after the bait drops, (b) assess the daily uptake rate, by repeating measurements every 24 hours, and (c) investigate the relationship of bait uptake rate, rodent abundance, and land crab diversity and abundance, by combining results from different islands.

In all cases, bait uptake was measured daily for 6–10 consecutive days in a systematic way, starting on the same day of bait broadcast. For all pre-eradication studies bait was broadcast by hand, whereas for the eradications either aerial or hand broadcast was used (Table 1). Bait uptake was measured in fixed circular plots as described by Pott, et al. (2015). A subset of the resulting dataset was included in the meta-analysis by Pott, et al. (2015), which showed the utility of bait availability studies. However, there are three major differences with the present study. Firstly, Pott and colleagues only used a subset of the Mexican dataset due to the limited data available for the other islands (e.g. data from only two of the 6–10 days available were analysed). Secondly, the results presented here derived from standardised monitoring methodologies. Lastly, our focus is to investigate the role of land crabs in the overall bait uptake in more detail, distinguishing for crab type (hermit and burrowing).

Table 1 General description of the six Mexican islands where land crabs were monitored before and after the successful rodent eradications.

Archipelago Island	Area (ha)	Ecosystem type	Dominant vegetation[1]	Species eradicated (year)[2]	Eradication method and bait rate (total kg/ha)	Main bait competitors (seasonal fluctuation)
Arrecife Alacranes						
Pájaros	3	Savanna	Shrubs and grasses	*Mus musculus* (2011)	Hand broadcast 17 kg/ha	Hermit crabs (low fluctuation)
Pérez	13	Savanna	Shrubs	*Rattus rattus* (2011)	Hand broadcast 17 kg/ha	Hermit crabs (low fluctuation)
Muertos	15	Savanna	Shrubs	*Mus musculus* (2011)	Hand broadcast 17 kg/ha	Small burrowing crabs (low fluctuation)
Isabel	82	Tropical seasonal forest	Deciduous forest	*Rattus rattus* (2009)	Aerial broadcast 20 kg/ha	Large burrowing crabs[3] (high fluctuation)
Banco Chinchorro						
Cayo Norte Mayor	30	Tropical rainforest	Mangroves & evergreen forest	*Rattus rattus* (2012)	Aerial broadcast 42 kg/ha	Large burrowing crabs (moderate fluctuation)
Cayo Centro	539	Tropical rainforest	Mangroves & evergreen forest	*Rattus rattus* (2015)	Aerial broadcast 60 kg/ha	Large burrowing crabs (moderate fluctuation)

[1]See Fig. 1.
[2]Always end of dry season.
[3]There was virtually no crab interference during the rat eradication; bait lasted for weeks.
Sources: Samaniego-Herrera, et al., 2013; Samaniego-Herrera, et al., 2014; Samaniego-Herrera, et al., 2018.

Land crab recovery

On all islands, land crab activity was monitored twice a year, at the end of each dry and wet season, both before (2–3 years) and after (1–5 years) each rodent eradication. Every season, several (6–18) fixed plots (25 m × 2 m) were used to estimate crab density; the exception was on Isabel Island, where two 300 m × 6 m plots were used. Plots were walked for 3–5 consecutive nights. One person with a headlamp walked in the middle of the plot, starting one hour after sunset. In order to walk all plots within 90 minutes (i.e. the peak activity period), several observers participated each night on some islands. The number of land crabs, by species, was recorded. Minimum training is required to carry out this task given the morphological differences of the species present.

Data analysis

Bait uptake trends (always measured in the dry season) were investigated using a linear mixed model (R software package nlme) for bait availability, where the density of bait (kg/ha), using the difference of target density minus measured density as the response variable (i.e. comparing rates of decline rather than actual densities), was dependent on time (days) and interactions of time with fixed effect covariates (i.e. covariates which would affect bait availability). These fixed effects included whether the study was conducted prior to or during the eradication (distinguishing between first and second bait application), whether rats or mice were the target species and how abundant they were (according to local mark-recapture studies by Samaniego-Herrera (2014)), and the abundance of both types of land crabs: hermit and burrowing (low – high, based on the monitoring in this study, therefore standardised). Although each island used different bait application rates, we were specifically interested in the rates of decline in bait availability. Inter-island differences were accounted for by including island as a random effect in our model. Diagnostic plots were visually checked for violations of model assumptions.

For land crab activity, an index of density was estimated as the number of nocturnal surface-active crabs per hectare. First, we used 2-way ANOVA to test the difference in density between seasons (dry and wet) and islands only for the pre-eradication periods (i.e. avoiding potential confounding effects caused by the eradications), as obvious fluctuations were occurring at least on some islands. In order to compare trends during favourable periods (hence closer to real density, as inactive crabs typically bury themselves), and given that the lower land crab activity during the dry season was confirmed for some islands, further analysis comparing pre- and post-eradication density used data from wet seasons only. Differences in density among island types (savanna, seasonal, rainforest) and periods (pre- and post-eradication) were tested with linear models. Data were log-transformed to achieve normality. All analyses were performed in R 3.4.

RESULTS

Land crabs and rodent eradications

The resulting database contains 108 eradication attempts spread over 101 tropical islands (Appendix 1; detailed spreadsheet: www.pacificinvasivesinitiative.org/). On some islands, there were two eradication attempts targeting a single rodent species or there were two rodent species being targeted by a single eradication attempt. Island sizes range from 0.1 ha to 4,310 ha (median = 10 ha). Most attempts (86.1%) targeted only rats (*Rattus exulans*, *R. norvegicus, R. rattus* or *R. tanezumi*), 2.8% only mice (*Mus musculus*) and 11.1% targeted both rats and mice.

Eighty-nine (82.4%) of the eradication attempts were successful and 19 (17.6%) failed (Table 2). Land crab interference was reported as important in 56.2% of the successful attempts and in 100% of the failed ones. However, for the latter, in addition to land crab interference other potential factors that may have contributed to the failure were also reported. Examples of such factors are gaps in bait coverage, which in turn can increase in area as land crabs take bait at the edges.

Over 82% of the eradication attempts used higher bait rates ($\bar{x} = 25.7$ kg/ha, range: 3–163 kg/ha) compared to those typically used on temperate islands (12 kg/ha; Broome, et al., 2014). In all cases, the justification for using higher bait rates was high rodent abundances (either estimated or assumed) and land crab presence, although abundance of either was rarely quantified. On some islands, additional factors such as high abundance of small invertebrates (e.g. ants and cockroaches, also bait consumers) were also mentioned.

Land crab abundances were reported as having been estimated either through measurements (21.5%) or observations (55.9%) during the planning phase; the rest (22.6%) of the cases did not try to estimate land crab abundance. On most islands, land crabs have been identified to the genus level. Through our research on current taxonomy, we identified 165 species of land crabs in 52 genera and 15 families, of which seven genera have been reported as important bait consumers (Table 3). For most islands (90.7%), only three or fewer land crab species were reported to be present.

Considering all islands, the 2-way ANOVA test revealed significant differences in total bait rates used depending on which type of land crab was the main source of interference (F = 11.33, $p<0.001$) and on the interaction between crab type and island type (F = 3.65, $p<0.001$), whereas island type was marginally significant (F = 3.02, $p = 0.05$). Higher bait rates (17–163 kg/ha) were used when burrowing crabs were the main bait competitors (n= 8), followed by hermit crabs (3–83.3 kg/ha; n= 55) and cases with 'no interference' (8–33.2 kg/ha; n= 39), i.e. low crab abundance (Fig. 2). When considering only successful attempts, the patterns remained the same.

Rodents impacts on land crabs

Accounts of insular land crab populations being negatively impacted by invasive rodents included 15 populations of ten species across nine countries and overseas territories (Table 4). These impacts are mainly in the form of population suppression and ecological extirpation. Pascal, et al. (2004) first suggested possible rat predation on *Gecarcinus ruricola* on Hardy Islet, after they found numerous crab carapaces in rat middens and the index of crab abundance increased after the rat eradication. Samaniego-Herrera & Bedolla-Guzmán (2012) and Samaniego-Herrera (2014) confirmed invasive rats can indeed cause ecological extirpations of land crabs; they documented how *G. quadratus*, a small burrowing crab, shifted from being extremely rare before the *R. rattus* eradication to being the most abundant species four years post-eradication. Evidence of rodent predation on land crabs as well as the dramatic recovery responses following rat eradication is growing (Harper & Bunbury, 2015; Samaniego-Herrera, et al., 2017). For example, Nigro, et al. (2017) showed that the recovery of two carnivorous species (*Geograpsus* spp.), the smallest of the Palmyra atoll crabs, led to a dramatic widening of the crab trophic niche following the rat eradication on Palmyra atoll, which is altering the ecology of the atoll presumably towards a more natural state. Likewise, Russell, et al. (2015) showed land crab trophic position differed depending on what invasive rat species is present.

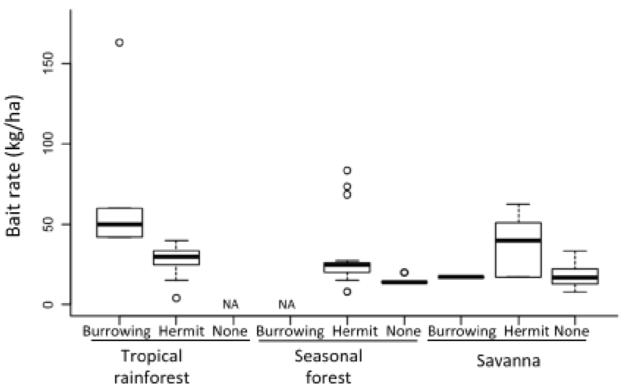

Fig. 2 Total bait used (median, IQR, range and outliers) in successful rodent eradications on tropical islands with land crabs, by type of dominant land crab and type of island. None = land crabs were present at low densities therefore interference was minimum.

Table 2 Success rate of hand and aerial rodent eradication projects, per target species, on tropical islands with land crabs.

Target species	Failed		Successful	
	%	n	%	n
Mus musculus	28.6	2	71.4	5
Rattus exulans	17.4	12	82.6	57
Rattus norvegicus	0	0	100	3
Rattus rattus	18.5	5	81.5	22
Rattus tanezumi	0	0	100	2
TOTAL	17.6	19	82.4	89

Case study: Mexican tropical islands

Bait uptake

Linear mixed models were constructed, including period (trials or during eradication), rodent species (*R. rattus* or *M. musculus*), rodent abundance (high or low) and land crab abundance (high or low) per crab type (hermit or burrowing) as covariates. The model with the greatest support (49.8%) revealed a complex relationship between the difference in bait density from target density (response variable) and all variables tested and some interactions (Table 5). Essentially, the rate of decline of bait density depends on (a) days since broadcast, declining over time, (b) the type of crab, declining faster with burrowing crabs, (c) the density of burrowing crabs, declining with high density, (d) the density of hermit crabs, declining with high density, (e) study type, declining faster during trials, (f) the broadcast, declining faster after first broadcast, (g) the density of rats, declining with high density, and (h) the density of mice, declining with high density. Fig. 3 illustrates the variability in bait density among plots and islands after the rodent eradications, although in all cases (trials and eradications) bait was still readily available after the recommended four nights (Keitt, et al., 2015). The faster decline in available bait when burrowing crabs are abundant is a novel result.

Land crab recovery

For the pre-eradication period, the 2-way ANOVA revealed significant differences in land crab density depending on island type (highest on rainforest islands)

Table 3 Current taxonomy of the 52 genera and 165 species of land crabs with island records.

Infraorder	Family	Genus[1]	No. species[1]	Documented bait consumer? [2]	Documented rodent vulnerability?[3]
Anomura	Coenobitidae	*Birgus*	1	Yes	Yes
		Coenobita	17	Yes	
	Diogenidae	*Clibanarius*	4		
		Calcinus	1		
	Porcellanidae	*Petrolisthes*	1		
Brachyura	Eriphiidae	*Eriphia*	2		
	Gecarcinunidae	*Barytelphusa*	1		
	Parathelphusidae	*Adeleana*	1		
		Austrothelphusa	1		
		Geelvinkia	1		
		Holthuisana	2		
		Rouxana	3		
		Terrathelphusa	1		
		Thelphusula	2		
	Gecarcinidae	*Cardisoma*	4	Yes	
		Discoplax	7	Yes	
		Epigrapsus	3		
		Gecarcinus	4	Yes	Yes
		Gecarcoidea	3	Yes	
		Johngarthia	5	Yes	Yes
	Grapsidae	*Geograpsus*	5	Yes	Yes
	Sesarmidae	*Aratus*	1		
		Armases	5		
		Chiromantes	4		
		Episesarma	1		
		Geosesarma	24		
		Karstama	3		
		Labuanium	2		
		Metasesarma	1		
		Metopaulinas	1		
		Neosarmatium	1		
		Parasesarma	1		
		Sesarma	8		
		Sesarmoides	1		
		Sesarmops	1		
	Mictyridae	*Mictyris*	1		
	Ocypodidae	*Afruca*	1		
		Austruca	1		
		Cranuca	1		
		Gelasimus	2		
		Leptuca	12		
		Minuca	6		
		Ocypode	6	Yes	Yes
		Tabuca	1		
		Uca	1		Yes
		Ucides	2		
	Geryonidae	*Carcinus*	1		
	Potamidae	*Cerberusa*	1		
		Potamon	2		
	Potamonautidae	*Madagapotamon*	1		
		Malagasya	1		
	Pseudothelphusidae	*Guinotia*	2		

[1]According to Ng, et al. (2008), McLaughlin, et al. (2010) and Shih, et al. (2016).
[2]According to this review.
[3] See Table 4.

Table 4 Documented land crab species negatively impacted by invasive rodents on tropical islands.

Species	Mean size (mm)	Invasive rodent	Rodent impact	Island/archipelago	Reference
Birgus latro	640	*Rattus rattus*	Population suppression	Tetiaroa Atoll, Society Islands, French Polynesia	Genet & Gaspar, pers. comm. 2017
Gecarcinus lateralis	37.7	*Rattus rattus*	Ecological extirpation	Pérez Island, Arrecife Alacranes, Mexico	Samaniego-Herrera, et al., 2017
		Rattus rattus	Ecological extirpation	Banco Chinchorro, Mexico	This study
		Rattus rattus	Ecological extirpation	Half Moon Caye, Belize	Samaniego-Herrera, et al., 2015
Gecarcinus quadratus	38.5	*Rattus rattus*	Ecological extirpation	Isabel Island, Mexico	Samaniego-Herrera & Bedolla-Guzmán, 2012
Gecarcinus ruricola	69	*Rattus rattus*	Population suppression	Hardy Island, Martinique	Lorvelec & Pascal, 2005
Geograpsus crinipes	46.4	*Rattus rattus*	Ecological extirpation	Palmyra Atoll, Tropical Pacific	Nigro, et al., 2017
		Rattus sp.	Ecological extirpation	Mañagaha Island, Northern Mariana Is	Paulay & Starmer, 2011
Geograpsus grayi	55.3	*Rattus norvegicus, R. exulans*	Ecological extirpation	Raoul Island, New Zealand	Bellingham, et al., 2010
		Rattus rattus	Ecological extirpation	Palmyra Atoll, Tropical Pacific	Nigro, et al., 2017
		Rattus sp.	Ecological extirpation	Mañagaha Island, Northern Mariana Is	Paulay & Starmer, 2011
Johngarthia planata[1]	92.5	*Rattus rattus*	Population suppression	Clipperton Island, eastern Pacific	Pitman, et al., 2005
Ocypode kuhlii	50	*Rattus norvegicus, R. exulans*	Ecological extirpation	Raoul Island, New Zealand	Bellingham, et al., 2010
Ocypode quadrata	45	*Mus musculus, Rattus rattus*	Population suppression	Alacranes Islands, Mexico	This study
Uca spp.	25–40	*Rattus exulans, R. tanezumi*	Population suppression	Wake Atoll, Tropical Pacific	Carlton & Hodder, 2003

[1]Referred to as *Gecarcinus planatus* before 2008.

($F = 28.01$, $p < 0.001$) and season (higher in wet season) ($F = 15.05$, $p < 0.001$). However, Tukey comparisons confirmed that the increase during wet seasons was significant only on two islands (Isabel and Muertos).

Given the differences between seasons on some islands, trends in land crab abundance between pre and post eradication periods was evaluated only for wet seasons. The linear model confirmed that land crab densities (pooling species) are significantly higher ($R^2 = 0.19$, $F = 30.27$, $p < 0.001$) post-eradication (1–5 years later) on all islands, except on Cayo Norte and Pérez where the increase was not significant. On Isabel Island, the smallest burrowing crab (*G. quadratus*) showed a substantial trend of increase over a period of five years after the rat eradication (Fig. 4).

DISCUSSION

Land crabs and rodent eradications

Interference by land crabs remains poorly understood, documented and managed, despite its significant impact on rodent eradication operations on tropical islands. It appears that inconsistent application of recommended practices (Wegmann, 2008; PII, 2011; Keitt, et al., 2015) continues across projects (Broome, 2011), e.g. a lack of estimating land crab densities and undertaking consumption trials to inform baiting rates. Without determining how significant the land crab problem is on an island and how it fluctuates with seasons, managers tend to automatically apply a mitigation strategy of high bait application rates.

However, the use of high bait rates increases the cost of operations, adds logistical complexity and, if unnecessarily high, potentially increases the risk to non-target vertebrates (Pitt, et al., 2015) (as invertebrates are not susceptible to anticoagulants (Broome, et al., 2012)). The highest total bait rate used in a rodent eradication to date was 186 kg/ha on Palmyra Atoll (Wegmann, et al., 2012). This high bait rate (spread over two applications) was determined and approved on the basis of sound field studies (Wegmann, et al., 2008; Wegmann, et al., 2011) as it had been demonstrated that the diverse and abundant land crab community represented a significant bait 'sink', warranting drastic mitigation measures. Importantly, Palmyra has an exceptionally wet climate and this eradication case is an outlier, as the second highest bait rate used to date is 94.2 kg/ha for the eradication of *R. exulans* on two Gambier

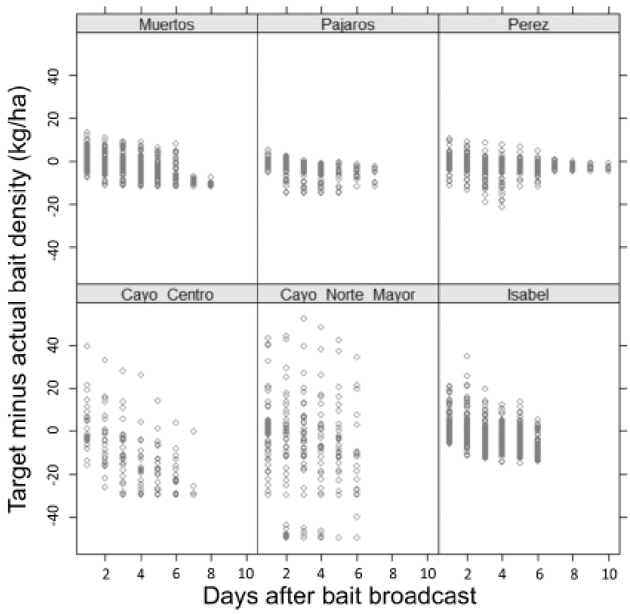

Fig. 3 Bait availability days after hand or aerial broadcast during six rodent eradications on Mexican islands.

islets (Kamaka and Makaroa)(Appendix 1). In addition, the largest rat eradication on a rainforest island (539 ha Cayo Centro, Banco Chinchorro), where large land crabs are abundant, was successful using 60 kg/ha over two applications (Samaniego-Herrera, et al., 2018). Moreover, there have been several instances where managers reported that the bait rates used were higher than required based on their observations of bait availability 5–15 days post bait application, which also coincided with 'fewer land crabs than expected' (Steve Cranwell, Araceli Samaniego, Elenoa Seniloli, pers. comm.). Studies (Samaniego-Herrera, et al., 2014; Samaniego-Herrera, et al., 2018) have shown that temporal land crab interference can vary substantially even on rainforest islands, the lowest peaks of activity being over the driest months. Thus, timing the eradication operation to coincide with the driest conditions of the year is recommended, particularly on islands with high abundance of burrowing crabs. Unquestionably, land crab activity is only one of the many factors that must be taken into account while planning an eradication (PII, 2011), so this has to be done in tandem with the other components, for example, minimising operational risks and the potential lethality to non-target species. Land crab interference with bait stations was outside the scope of this paper, but it is certainly a problem (Wegmann, 2008).

In addition to land crab abundance, land crab species composition also affects bait uptake. Burrowing land crabs, in particular large species (e.g. genera *Cardisoma*, *Johngarthia* and *Discoplax*), are generally the most problematic, although coconut crabs (*B. latro*) can be as troublesome even though they are usually in low to medium abundances. This has been reflected in the tendency to use higher bait rates on islands with abundant burrowing crabs, and confirmed by our case study where bait availability was quantified. Hermit crabs appear to be more widespread globally, so they may cause less interference on individual islands but affect more islands overall. Note that burrowing crabs, although larger, are more elusive than hermits due to their propensity to burrow or seek shelter under rocks or leaf litter during the day (Bliss, et al., 1978). Species richness of land crabs per island is likely to be generally underestimated. On the few islands for which comprehensive inventories exist (e.g. Christmas Island, Seychelles or French Polynesia), a list of 5–12 species of land crabs is common (Orchard, 2012), which is higher when compared to the three or fewer species usually recorded by eradication managers.

At present, bait uptake trials (e.g. Pott, et al., 2015) are the best way to predict bait rates required for rodent eradication. However, if climatic conditions are not very similar during implementation, the 'land crab scenario' could be very different and significantly affect the bait consumption rate. In order to better predict the potential variability of bait uptake in the presence of certain land crab communities, future monitoring of climatic conditions and land crab communities on a suite of islands is required. Note that carnivorous and intertidal crabs may also consume bait (confirmed for ghost crabs, A. Barnaud pers. comm.), but due to their generally low abundance (probably due to vulnerability to rats) and/or limited distribution (coastal), they tend to be neglected. The implications for failed eradication attempts are important. Estimations of bait uptake rates may no longer be true if a second eradication attempt occurs within a few years of the first one, i.e. when crab populations are more abundant because rats haven't had the time to fully recover and therefore haven't again supressed the land crabs.

Behaviours as well as consumption capabilities vary widely among groups and species, hence the importance of identifying species or at least type of crab (hermit, coconut, burrowing; Fig. 5). Hermit crabs are small and slow eaters and walkers compared to the average burrowing crab. They can take only one piece of bait at a time and they do not cache food. In contrast, most burrowing crabs are able to take up to three pieces of bait at a time (depending on crab size/species) and walk quickly to their burrow where they cache the bait (G. Harper & A. Samaniego pers. obs.). How much bait they can accumulate, how fast, and how long it takes for it to be eaten has not been determined.

Land crab activity is regulated by a combination of air and soil surface temperature, relative humidity, the intensity of insolation (solar radiation) and the availability of protective cover, be it leaf litter, suitable cavities, or soil for burrowing, which is further influenced by the soil compaction (Bliss, et al., 1978; Green, 1997; Brook, et al., 2009). The optimum temperature for land crab activity appears to be about 30°C, with virtually no activity below 18°C (Bliss, et al., 1978). Hence, in order to mitigate the desiccating effects of the high temperatures that crabs require to be active, their activity is largely restricted to periods and locations with high humidity. To reduce interference by land crabs, on 'wet' subtropical islands cooler months should be targeted for eradications.

The thermoregulatory abilities of hermit crabs (*Coenobita* spp.) are low, which is less of a problem where there is little temperature variation, as often occurs on wet tropical islands, but on arid islands they are essentially nocturnal (Achituv & Ziskind, 1985). Wind strength will also affect activity through increasing desiccation (Barnes, 1997). For burrowing crabs (e.g. *Gecarcinus* spp.) on seasonally arid islands, activity is dictated by relative humidity, with little or no activity below 77% RH, through to high activity above 95% RH (Bliss, et al., 1978; Green, 1997; Capistrán-Barradas, et al., 2003). Burrowing crabs probably occupy burrows to reduce their water loss. Often, unseasonal rain or even showers will initiate short periods of activity, but if conditions are very dry land crabs can remain underground for several months (Bliss, et al., 1978).

The effects of humidity and insolation on land crab activity strongly suggest nocturnal land crab monitoring will detect the highest crab activity and species diversity, particularly if all habitats are sampled and the season is taken into account. Land crabs are long-lived species, so monitoring tends to indicate how many land crabs are active at that time, not how many are actually present.

Table 5 Significant parameters in relation to bait availability after rodent eradications, according to linear mixed models for six rodent eradications on Mexican tropical islands.

	Value	Std Error	DF	t-value	p-value
Day	-1.456	0.275	2768	-5.287	> 0.001
Period	4.204	1.014	2768	4.142	> 0.001
Bait application	-4.658	1.056	2768	-4.407	> 0.001
Day × Burrowing	-2.343	0.262	2768	-8.914	> 0.001
Day × Hermit	-1.135	0.338	2768	-3.357	> 0.001
Day × Period	-0.896	0.254	2768	-3.521	> 0.001
Day × Bait application	1.115	0.275	2768	4.050	> 0.001
Hermit × Bait application	6.912	1.610	2768	4.291	> 0.001
Day × Rodent	-0.019	0.005	2768	-3.725	> 0.001
Day × Burrowing × Bait application	1.672	0.431	2768	3.875	> 0.001

Rodent impacts on land crabs

Except for New Zealand, little has been documented regarding impacts of invasive rodents on island invertebrates in general (St Clair, 2011). Furthermore, few land crab accounts exist, due to the limited research conducted in the tropics (Brook, et al., 2009) and the low proportion of rodent eradications carried out in this region (Howald, et al., 2007; Holmes, et al., 2015). Our list of land crab species impacted by rodents is most likely to be severely under-reported as Rodentia comprise over a quarter of terrestrial mammal species known to forage on intertidal food sources. Burrowing crabs make up a substantial proportion of this and the number of species is highly likely to be an underestimation (Carlton & Hodder, 2003).

Adding to the impacts of invasive rodent predation on a wide range of plants and animals (Carlton & Hodder, 2003; St Clair, 2011; Sunde, 2012; Harper & Bunbury, 2015), the suppression of native ecosystem engineers, such as land crabs, by rats and other invasive species such as ants, cats and dogs, could have significant and enduring consequences on relatively simple island ecosystems (Carlton & Hodder, 2003; O'Dowd, et al., 2003). Land crabs are often the largest invertebrates on tropical islands, and particularly atolls, and will occupy the niches of vertebrates on small oceanic islands (Burggren & McMahon, 1988). They are highly integrated in the ecosystem energetics of tropical islands, as they control recruitment and species composition of seedlings on the forest floor (Green, et al., 1999; Lindquist & Carroll, 2004). They also regulate nutrient dynamics through substantial leaf litter consumption (Kellman & Delfosse, 1993; Capistrán-Barradas, et al., 2003; Gutiérrez & Jones, 2006; Gutiérrez, et al., 2006). Hence, they may govern the growth and productivity of tropical forests (Lindquist, et al., 2009) and sustain diversity at large scales (Young, et al., 2013; Nigro, et al., 2017). Moreover, plant composition will be influenced by soil structure, which is affected by land crab activity through inland transfer of marine debris and shells, removal of algae from rock surfaces subsequently deposited as faeces, and by increasing leaf litter breakdown through deposition underground in burrows. Given the critical role land crabs play in island ecosystems, it is recommended that these are included as outcome indicators and monitored post-eradication.

CONCLUSION

Land crabs are a diverse group. For management purposes, it is useful to distinguish three groups: hermit crabs, coconut crabs and burrowing crabs, noting the latter vary widely in size. The ecology and climatic tolerances of each group is different, as is their capacity as bait consumers. Assessing species richness and abundance of land crabs should be a priority in the planning phase of rodent eradication projects in the tropics, so that their interference can be efficiently managed during eradications. Similarly, changes to the land crab community should be measured

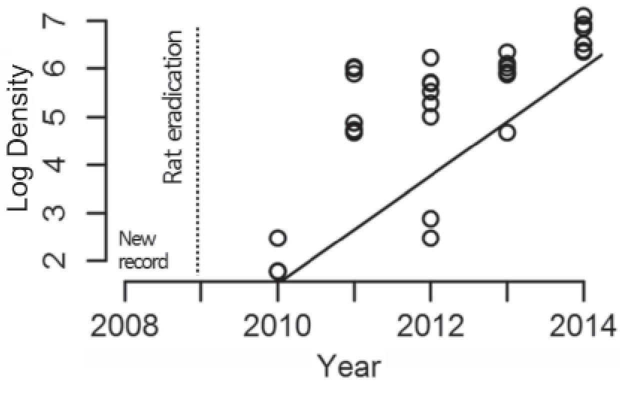

Fig. 4 Population increase of *Gecarcinus quadratus* on Isabel Island, Mexico after the ship rat eradication in 2009. The single record in 2008 was a new record for the island.

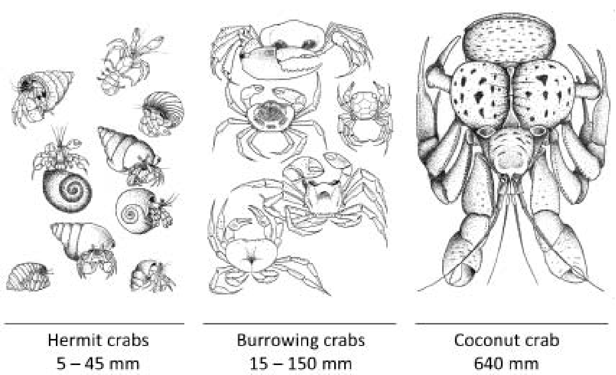

Fig. 5 Types of land crabs in relation to potential interference with rodent eradications on tropical islands. Sizes refer to the range of mean size across species.

post-eradication as indicators of ecosystem recovery. Once enough data are gathered regarding bait consumption in a standardised manner, we will improve our ability to predict appropriate bait rates for the eradication of invasive rodents on different types of tropical islands, as is currently done for temperate islands. Where compatible with other factors (e.g. non-target species and human activities), rodent eradications on tropical islands should be timed for the driest conditions or alternatively on 'wet' subtropical islands, the coolest months.

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REFERENCES

Achituv, Y. and Ziskind, M. (1985). 'Adaptation of *Coenobita scaevola* (Forskål, 1775) (Crustacea, Anomura) to terrestrial life in desert-bordered shore line'. *Marine Ecology Progress Series* 25: 189–197.

Athens, J.S., Tuggle, H.D., Welch, D.J. and Ward, J.V. (2002). 'Avifaunal extinctions, vegetation change, and Polynesian impacts in prehistoric Hawai'i'. *Archaeology in Oceania* 37(2): 57.

Atkinson, I.A.E. (1985). 'The spread of commensal species of *Rattus* to oceanic islands and their effect on island avifaunas'. In*:* P.J. Moors (ed.) *Conservation of Island Birds*. pp. 35–81. Cambridge: International Council for Bird Preservation.

Barnes, D.K. (1997). 'Ecology of tropical hermit crabs at Quirimba Island, Mozambique: Distribution, abundance and activity'. *Marine Ecology Progress Series* 154: 133–142.

Bellard, C., Cassey, P. and Blackburn, T.M. (2016). 'Alien species as a driver of recent extinctions'. *Biology Letters* 12(2): 20150623.

Bliss, D.E., Van Montfrans, J., Van Montfrans, M. and Boyer, J.R. (1978). 'Behavior and growth of the land crab *Gecarcinus lateralis* (Freminville) in southern Florida'. *Bulletin of the American Museum of Natural History* 160(11): 111–152.

Brook, S., Grant, A. and Bell, D. (2009). 'Can land crabs be used as a rapid ecosystem evaluation tool? A test using distribution and abundance of several genera from the Seychelles'. *Acta Oecologica* 35(5): 711–719.

Broome, K. (2011). 'Transferring rodent eradication success from temperate regions into the tropical Pacific – What do we need to learn?'. *PII News.* Auckland: Pacific Invasives Initiative.

Broome, K.G., Fairweather, A.A.C. and Fisher, P. (2012). *Brodifacoum Pesticide Information Review. Version 2012/1*. Hamilton, New Zealand: New Zealand Department of Conservation.

Broome, K.G., Cox, A., Golding, C., Cromarty, P., Bell, P. and McClelland, P. (2014). *Rat Eradication Using Aerial Baiting: Current Agreed Best Practice Used in New Zealand (Version 3.0)*. Wellington, New Zealand: New Zealand Department of Conservation.

Burggren, W.W. and McMahon, B.R. (1988). *Biology of the Land Crabs,* Cambridge, Cambridge University Press.

Capistrán-Barradas, A., Defeo, O. and Moreno-Casasola, P. (2003). 'Density and population structure of the red land crab *Gecarcinus lateralis* in a tropical semi-deciduous forest in Veracruz, Mexico'. *Interciencia* 28(6): 323–327.

Carlton, J.T. and Hodder, J. (2003). 'Maritime mammals: Terrestrial mammals as consumers in marine intertidal communities'. *Marine Ecology Progress Series* 256: 271–286.

Croll, D.A., Newton, K.M., McKown, M., Holmes, N., Williams, J.C., Young, H.S., Buckelew, S., Wolf, C.A., Howald, G., Bock, M.F., Curl, J.A. and Tershy, B.R. (2016). 'Passive recovery of an island bird community after rodent eradication'. *Biological Invasions* 18(3): 703–715.

DIISE (2016*). The Database of Island Invasive Species Eradications*, Developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand. <http://diise.islandconservation.org>.

Green, P.T. (1997). 'Red crabs in rain forest on Christmas Island, Indian Ocean: Activity patterns, density and biomass'. *Journal of Tropical Ecology* 13(1): 17–38.

Green, P.T., Lake, P.S. and O'Dowd, D.J. (1999). 'Monopolization of litter processing by a dominant land crab on a tropical oceanic island'. *Oecologia* 119(3): 435–444.

Green, P.T., O'Dowd, D.J. and Lake, P.S. (2008). 'Recruitment dynamics in a rainforest seedling community: Context-independent impact of a keystone consumer'. *Oecologia* 156(2): 373–385.

Griffiths, R., Miller, A. and Climo, G. (2011). 'Addressing the impact of land crabs on rodent eradications on islands'. *Pacific Conservation Biology* 17(4): 347–353.

Gutiérrez, J.L. and Jones, C.G. (2006). 'Physical ecosystem engineers as agents of biogeochemical heterogeneity'. *Bioscience* 56(3): 227–236.

Gutiérrez, J.L., Jones, C.G., Groffman, P.M., Findlay, S.E.G., Iribarne, O.O., Ribeiro, P.D. and Bruschetti, C.M. (2006). 'The contribution of crab burrow excavation to carbon availability in surficial salt-marsh sediments'. *Ecosystems* 9(4): 647–658.

Harper, G.A. and Bunbury, N. (2015). 'Invasive rats on tropical islands: Their population biology and impacts on native species'. *Global Ecology and Conservation* 3: 607–627.

Hilton, G.M. and Cuthbert, R.J. (2010). 'The catastrophic impact of invasive mammalian predators on birds of the UK Overseas Territories: A review and synthesis'. *Ibis* 152(3): 443–458.

Holmes, N.D., Griffiths, R., Pott, M., Alifano, A., Will, D., Wegmann, A.S. and Russell, J.C. (2015). 'Factors associated with rodent eradication failure'. *Biological Conservation* 185: 8–16.

Howald, G., Donlan, C.J., Galván, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21(5): 1258–1268.

Jones, H.P., Holmes, N.D., Butchart, S.H.M., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P.E., Cuthbert, R.J., Ebbert, S., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R. and Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences* 113(15): 4033–4038.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

Kellman, M. and Delfosse, B. (1993). 'Effect of the red land crab (*Gecarcinus lateralis*) on leaf litter in a tropical dry forest in Veracruz, Mexico'. *Journal of Tropical Ecology* 9(1): 55–65.

Le Corre, M., Danckwerts, D.K., Ringler, D., Bastien, M., Orlowski, S., Morey Rubio, C., Pinaud, D. and Micol, T. (2015). 'Seabird recovery and vegetation dynamics after Norway rat eradication at Tromelin Island, western Indian Ocean'. *Biological Conservation* 185: 85–94.

Lindquist, E.S. and Carroll, C.R. (2004). 'Differential seed and seedling predation by crabs: Impacts on tropical coastal forest composition'. *Oecologia* 141(4): 661–671.

Lindquist, E.S., Krauss, K.W., Green, P.T., O'Dowd, D.J., Sherman, P.M. and Smith, T.J. (2009). 'Land crabs as key drivers in tropical coastal forest recruitment'. *Biological Reviews* 84(2): 203–223.

McLaughlin, P.A., Komai, T., Lemaitre, R. and Rahayu, D.L. (2010). 'Annotated checklist of anomuran decapod crustaceans of the world (exclusive of the Kiwaoidea and families Chirostylidae and Galatheidae of the Galatheoidea) Part I – Lithodoidea, Lomisoidea and Paguroidea'. *The Raffles Bulletin of Zoology* 23(1): 131–137.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G.A.B. and Kent, J. (2000). 'Biodiversity hotspots for conservation priorities'. *Nature* 403(6772): 853–858.

Ng, P.K.L., Guinot, D. and Davie, P.J.F. (2008). 'Systema Brachyurorum: Part 1. An annotated checklist of extant brachyuran crabs of the world'. *The Raffles Bulletin of Zoology* 17: 1–286.

Nigro, K.M., Hathaway, S.A., Wegmann, A.S., Miller-ter Kuile, A., Fisher, R.N. and Young, H.S. (2017). 'Stable isotope analysis as an early monitoring tool for community-scale effects of rat eradication'. *Restoration Ecology* 10.1111/rec.12511.

O'Dowd, D.J., Green, P.T. and Lake, P.S. (2003). 'Invasional 'meltdown' on an oceanic island'. *Ecology Letters* 6(9): 812–817.

Orchard, M. (2012). *Crabs of Christmas Island*, Christmas Island Natural History Association.

Paine, R.T. (1966). 'Food web complexity and species diversity'. *American Naturalist* 100(910): 65–75.

Pascal, M., Brithmer, R., Lorvelec, O. and Venumiere, N. (2004). 'Conséquences sur l'avifaune nicheuse de la réserve naturelle des Îlets De Sainte-Anne (Martinique) de la récente invasion du rat noir (*Rattus rattus*), etablies à l'issue d'une tentative d'éradication'. *Revue d'écologie* 59(1–2): 309–318.

Paulay, G. and Starmer, J. (2011). 'Evolution, insular restriction, and extinction of oceanic land crabs, exemplified by the loss of an endemic *Geograpsus* in the Hawaiian Islands'. *PLOS ONE* 6(5): e19916.

PII. (2011). *Resource Kit for Rodent and Cat Eradication.* Pacific Invasives Initiative. <http://pacificinvasivesinitiative.org/rce/index.html>. [Accessed January 2017].

Pitman, R.L., Ballance, L.T. and Bost, C. (2005). 'Clipperton Island: Pig sty, rat hole and booby prize'. *Marine Ornithology* 33: 193–194.

Pitt, W.C., Berentsen, A.R., Shiels, A.B., Volker, S.F., Eisemann, J.D., Wegmann, A.S. and Howald, G.R. (2015). 'Non-target species mortality and the measurement of brodifacoum rodenticide residues after a rat (*Rattus rattus*) eradication on Palmyra Atoll, tropical Pacific'. *Biological Conservation* 185: 36–46.

Pott, M., Wegmann, A.S., Griffiths, R., Samaniego-Herrera, A., Cuthbert, R.J., Brooke, M.d.L., Pitt, W.C., Berentsen, A.R., Holmes, N.D., Howald, G.R., Ramos-Rendón, K. and Russell, J.C. (2015). 'Improving the odds: Assessing bait availability before rodent eradications to aid in selecting bait application rates'. *Biological Conservation* 185: 27–35.

Russell, J.C., Caut, S., Anderson, S.H. and Lee, M. (2015). 'Invasive rat interactions and over-invasion on a coral atoll'. *Biological Conservation* 185: 59–65.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Russell, J.C., Meyer, J.-Y., Holmes, N.D. and Pagad, S. (2017). 'Invasive alien species on islands: Impacts, distribution, interactions and management'. *Environmental Conservation* 10.1017/S0376892917000297.

Samaniego-Herrera, A. (2014). '*Ecology and Impacts of Invasive Rodents on Tropical Islands, in Relation to Eradication Operations'.* PhD thesis. Auckland: The University of Auckland.

Samaniego-Herrera, A., Aguirre-Muñoz, A., Rodríguez-Malagón, M., González-Gómez, R., Torres-García, F., Méndez-Sánchez, F., Félix-Lizárraga, M. and Latofski-Robles, M. (2011). 'Rodent eradications on Mexican islands: Advances and challenges'. In*:* C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 350–355. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Samaniego-Herrera, A. and Bedolla-Guzmán, Y. (2012). 'Land crabs (Decapoda, Brachyura, Gecarcinidae) on Isabel Island, Mexico, including a new record, and its relation to the removal of invasive rats'. *Crustaceana* 85(8): 1007–1011.

Samaniego-Herrera, A., Anderson, D.P., Parkes, J.P. and Aguirre-Muñoz, A. (2013). 'Rapid assessment of rat eradication after aerial baiting'. *Journal of Applied Ecology* 50(6): 1415–1421.

Samaniego-Herrera, A., Russell, J.C., Choquenot, D., Aguirre-Muñoz, A. and Clout, M. (2014). 'Invasive rodents on tropical islands: eradication recommendations from Mexico'. In*:* R.M. Timm and J.M. O'Brien (eds.) *Proceedings of the 26th Vertebrate Pest Conference*, pp. 43–50. Davis: University of California.

Samaniego-Herrera, A., Clout, M.N., Aguirre-Muñoz, A. and Russell, J.C. (2017). 'Rodent eradications as ecosystem experiments: A case study from the Mexican tropics'. *Biological Invasions* 19(6): 1761–1779.

Samaniego-Herrera, A., Aguirre-Muñoz, A., Bedolla-Guzmán, Y., Cárdenas-Tapia, A., Félix-Lizárraga, M., Méndez-Sánchez, F., Reina-Ponce, O., Rojas-Mayoral, E. and Torres-García, F. (2018). 'Eradicating invasive rodents from wet and dry tropical islands in Mexico'. *Oryx* 52: 559–570.

Sherman, P.M. (2002). 'Effects of land crabs on seedling densities and distributions in a mainland neotropical rain forest'. *Journal of Tropical Ecology* 18(1): 67–89.

Sherman, P.M. (2006). 'Influence of land crabs *Gecarcinus quadratus* (Gecarcinidae) on distributions of organic carbon and roots in a Costa Rican rain orest'. *Revista de Biologia Tropical* 54(1): 149–161.

Shih, H.-T., Ng, P.K., Davie, P.J., Schubart, C.D., Türkay, M., Naderloo, R., Jones, D. and Liu, M.-Y. (2016). 'Systematics of the family Ocypodidae Rafinesque, 1815 (Crustacea: Brachyura), based on phylogenetic relationships, with a reorganization of subfamily rankings and a review of the taxonomic status of *Uca* Leach, 1814, sensu lato and its subgenera'. *The Raffles Bulletin of Zoology* 64: 139–175.

St Clair, J.J.H. (2011). 'The impacts of invasive rodents on island invertebrates'. *Biological Conservation* 144(1): 68–81.

Sunde, F.G. (2012). '*The Effects of Rats on Rocky Intertidal Ecosystems in Northeastern New Zealand'.* MSc thesis. Auckland: University of Auckland.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?' *Biological Invasions* 8(4): 863–891.

Towns, D.R., West, C.J. and Broome, K.G. (2013). 'Purposes, outcomes and challenges of eradicating invasive mammals from New Zealand islands: An historical perspective'. *Wildlife Research* 40(2): 94–107.

Wegmann, A.S. (2008). *Land Crab Interference with Eradication Projects: Phase I – Compendium of Available Information*. Auckland: Pacific Invasives Initiative.

Wegmann, A.S., Helm, J., Jacobs, B., Samaniego, A., Smith, W., Drake, D., Fisher, R., Hathaway, S., Henry, A., Smith, J. and McKown, M. (2008). *Palmyra Atoll Rat Eradication: Biomarker Validation of an Effective Bait Application Rate*. Technical Field Report. Santa Cruz, California: Island Conservation.

Wegmann, A.S., Buckelew, S., Howald, G., Helm, J. and Swinnerton, K. (2011). 'Rat eradication campaigns on tropical islands: Novel challenges and possible solutions'. In*:* C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 239-243. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Wegmann, A.S., Flint, E., White, S., Fox, M., Howald, G., McClelland, P., Alifano, A. and Griffiths, R. (2012). 'Pushing the envelope in paradise: A novel approach to rat eradication at Palmyra Atoll'. In*:* R.M. Timm (ed.) *25th Vertebrate Pest Conference*, pp. 48–53. Davis: University of California.

Weigelt, P., Jetz, W. and Kreft, H. (2013). 'Bioclimatic and physical characterization of the world's islands'. *Proceedings of the National Academy of Sciences* 110(38): 15307–15312.

Young, H.S., McCauley, D.J., Guevara, R. and Dirzo, R. (2013). 'Consumer preference for seeds and seedlings of rare species impacts tree diversity at multiple scales'. *Oecologia* 172(3): 857–867.

Appendix 1. Details of rodent eradications on tropical islands with land crabs, up to 2015, conducted either by aerial broadcast (Aerial) or hand broadcast (Hand; Hand*= bait piles on the ground on a grid) of bait. Target species: Mm= *Mus musculus*, Re= *Rattus exulans*, Rn= *Rattus norvegicus*, Rr= *Rattus rattus*, Rt= *Rattus tanezumi*; Eradication result: F= failure, S= success; Bait type: 20R= Pestoff 20R, 25D= Bell-25D, 25W= Bell-25W; Crab genera: Bi= *Birgus*, Ca= *Cardisoma*, Co= *Coenobita*, Ge= *Gecarcinus*, Gd= *Gecarcoidea*, Gg= *Geograpsus*, Jo= *Johngartia*, Oc= *Ocypode*. [1]During the eradication; [2]Observed by project managers throughout the project.

Country/ territory	Island	Area (ha)	Target species	Erad. year	Erad. season	Erad. result	Erad. method	Total bait rate (kg/ ha)	Bait type	Island type	Main interference	Burrowing crab density[1]	Hermit crab density[1]	Coconut crab density[1]	Land crab genera[2]
Bahamas	Allen Cay	6.9	Mm	2012	Dry	S	Hand	40	25D	Savanna	Hermit	Low	Low	None	Ca, Co, Ge
Cook Islands	Anchorage	12.8	Re	2012	Wet	S	Hand	20	20R	Seasonal	Hermit	Medium	Medium	Medium	Bi, Ca, Co, Oc
Cook Islands	Motu Kena	1.2	Re	2012	Wet	S	Hand	20	20R	Seasonal	Hermit	Medium	Medium	Low	Bi, Ca, Co, Oc
Cook Islands	Motu Kena-iti	0.7	Re	2012	Wet	S	Hand	20	20R	Seasonal	Hermit	Low	Medium	Medium	Bi, Ca, Co, Oc
Cook Islands	Motu Tou	14.7	Re	2012	Wet	F	Hand	20	20R	Seasonal	Hermit	Medium	Medium	High	Bi, Ca, Co, Oc
Fiji	Mabualau	3.2	Re	2008	Dry	S	Hand	26	20R	Seasonal	Hermit	High	High	None	Ca, Co
Fiji	Nukubasaga	18.0	Re	2008	Dry	S	Aerial	25	20R	Seasonal	Hermit	Low	Medium	Low	Bi, Co
Fiji	Nukupureti	3.0	Re	2008	Dry	S	Aerial	25	20R	Seasonal	Hermit	Low	Medium	None	Co
Fiji	Nukusemanu	1.6	Re	2008	Dry	S	Aerial	25	20R	Seasonal	Hermit	Low	Medium	None	Co
Fiji	Qelelevu	147.0	Re	2008	Dry	S	Aerial	25	20R	Seasonal	Hermit	Low	Medium	High	Bi, Co
Fiji	Tauraria	49.3	Re	2008	Dry	S	Aerial	25	20R	Seasonal	Hermit	Low	Medium	Low	Bi, Co
Fiji	Tuinibeka	2.9	Re	2008	Dry	S	Aerial	25	20R	Seasonal	Hermit	Low	Medium	Low	Bi, Co
Fiji	Vatu-i-Ra	2.0	Re	2006	Dry	S	Hand	26	20R	Seasonal	Hermit	Low	Medium	None	Co
Fiji	Vetauua	35.0	Re	2008	Dry	S	Aerial	25	20R	Seasonal	Hermit	Low	Medium	High	Bi, Co
French Polynesia	Hiuveru	3.2	Rr	2008	Wet	F	Hand	15	20R	Seasonal	Hermit	Low	Low	None	Co
French Polynesia	Hiveu	4.7	Rr	2008	Wet	F	Hand	15	20R	Seasonal	Hermit	Low	Low	None	Co
French Polynesia	Kamaka	47.6	Re	2015	Dry	F	Aerial	94.2	25W	Rainforest	Hermit	Low	Medium	None	Co, Gg
French Polynesia	Makapu	11.2	Re	2003	Dry	S	Hand	4	20R	Rainforest	Hermit	Low	Medium	None	Co
French Polynesia	Makaroa	16.4	Re	2015	Dry	S	Aerial	94.2	25W	Rainforest	Hermit	Low	Medium	None	Co, Gg
French Polynesia	Mekiro	11.5	Re	2003	Dry	F	Hand	4	20R	Rainforest	Hermit	Low	Medium	None	Co
French Polynesia	Motu-o-ari	4.5	Re	2003	Dry	F	Hand	3	20R	Rainforest	Hermit	Low	High	None	Co
French Polynesia	Teauaone	8.8	Re	2003	Dry	F	Hand	3	20R	Rainforest	Hermit	Low	High	None	Co
French Polynesia	Temoe	430.8	Re	2015	Dry	S	Aerial	87.6	25W	Seasonal	Hermit	None	High	None	Co
French Polynesia	Tenarunga	424.0	Re	2015	NA	S	Aerial	72.6	25W	Seasonal	Hermit	None	Medium	None	Co
French Polynesia	Tenarunga	424.0	Rr	2015	NA	S	Aerial	72.6	25W	Seasonal	Hermit	None	Medium	None	Co
French Polynesia	Tepapuri	26.0	Re	2003	Dry	F	Hand	3	20R	Rainforest	Hermit	Low	High	None	Co
French Polynesia	Tiarao	4.2	Rr	2008	Wet	F	Hand	15	20R	Seasonal	Hermit	Low	Low	None	Co, Ge

Appendix 1 (continued) Details of rodent eradications on tropical islands with land crabs, up to 2015, conducted either by aerial broadcast (Aerial) or hand broadcast (Hand; Hand*= bait piles on the ground on a grid) of bait. Target species: Mm= *Mus musculus*, Re= *Rattus exulans*, Rn= *Rattus norvegicus*, Rr= *Rattus rattus*, Rt= *Rattus tanezumi*; Eradication result: F= failure, S= success; Bait type: 20R= Pestoff 20R, 25D= Bell-25D, 25W= Bell-25W; Crab genera: Bi= *Birgus*, Ca= *Cardisoma*, Co= *Coenobita*, Ge= *Gecarcinus*, Gd= *Gecarcoidea*, Gg= *Geograpsus*, Jo= *Johngartia*, Oc= *Ocypode*. [1]During the eradication; [2]Observed by project managers throughout the project.

Country/ territory	Island	Area (ha)	Target species	Erad. year	Erad. season	Erad. result	Erad. method	Total bait rate (kg/ ha)	Bait type	Island type	Main interference	Burrowing crab density[1]	Hermit crab density[1]	Coconut crab density[1]	Land crab genera[2]
French Polynesia	Toreauta	5.3	Rr	2011	Dry	S	Hand	27.3	25W	Seasonal	Hermit	Low	Low	Low	Bi, Co, Gg
French Polynesia	Vahanga	380.0	Re	2015	NA	S	Aerial	72.4	25W	Seasonal	Hermit	Low	Medium	Low	Co, Ca?
Kiribati	Big Ambo	1.4	Re	2009	Dry	S	Hand	14.3	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	Big Fred/Tonga	3.5	Re	2009	Dry	S	Hand	22.8	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	Big Nimroona	6.5	Re	2009	Dry	S	Hand	19.2	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	Big Tibo	3.8	Re	2012	Dry	S	Hand	12.5	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	Birnie	49.4	Re	2011	Dry	S	Aerial	51	20R	Savanna	Hermit	Low	High	None	Ca, Co
Kiribati	Drum	6.1	Re	2009	Dry	S	Hand	22.15	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	E isle	0.8	Re	2009	Dry	S	Hand	13.3	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	East Drum	1.0	Re	2009	Dry	S	Hand	15	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	Enderbury	608.0	Re	2011	Dry	F	Aerial	38.4	20R	Savanna	Hermit	Low	High	None	Bi, Ca, Co, Gg
Kiribati	Isles Lagoon 13	1.2	Re	2009	Dry	S	Hand	16.6	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	Isles Lagoon 16	4.1	Re	2009	Dry	S	Hand	13.9	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	Isles Lagoon 2	1.4	Re	2009	Dry	S	Hand	17.1	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	Isles Lagoon 21	1.2	Re	2009	Dry	S	Hand	16.6	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	Isles Lagoon 22	1.5	Re	2009	Dry	S	Hand	14.6	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	Isles Lagoon 23	0.1	Re	2009	Dry	S	Hand	10	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	Isles Lagoon 3	0.5	Re	2009	Dry	S	Hand	20	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	Isles Lagoon 4	1.4	Re	2009	Dry	S	Hand	10	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	Isles Lagoon 5	0.3	Re	2009	Dry	S	Hand	20	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	McKean	27.0	Rt	2008	Dry	S	Hand	62.8	20R	Savanna	Hermit	Low	High	None	Ca, Co
Kiribati	North Drum	2.5	Re	2009	Dry	S	Hand	24	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	NW Fred/Tonga	1.3	Re	2009	Dry	S	Hand	26.9	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	NW Nimroona	0.6	Re	2009	Dry	S	Hand	33.2	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	NW Tibo	0.8	Re	2009	Dry	S	Hand	12.5	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	SE Fred/Tonga	0.8	Re	2009	Dry	S	Hand	22.5	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	SW Islet Koil	0.1	Re	2009	Dry	S	Hand	8	20R	Savanna	None	Low	None	None	Ca?, Oc?
Kiribati	SW motu Koil	3.0	Re	2009	Dry	S	Hand	10	20R	Savanna	None	Low	None	None	Ca?, Oc?

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Country/ territory	Island	Area (ha)	Target species	Erad. year	Erad. season	Erad. result	Erad. method	Total bait rate (kg/ ha)	Bait type	Island type	Main interference	Burrowing crab density[1]	Hermit crab density[1]	Coconut crab density[1]	Land crab genera[2]
Kiribati	SW Nimroona	3.9	Re	2009	Dry	S	Hand	22.6	20R	Savanna	None	Low	None	None	Ca?, Oc?
Mauritius	Flat	249.6	Rr	1998	Dry	S	Hand	15	20R	Seasonal	Hermit	Low	High	None	Ca, Co
Mauritius	Flat	249.6	Mm	1998	Dry	S	Hand	15	20R	Seasonal	Hermit	Low	High	None	Ca, Co
Mauritius	Gabriel	40.5	Rr	1995	Dry	S	Hand	20	20R	Seasonal	Hermit	High	High	None	Ca, Co
Mauritius	Gunner's Quoin	65.0	Rn	1995	Dry	S	Hand	15	20R	Seasonal	Hermit	Low	Low	None	Ca, Co
Mexico	Cayo Centro	539.0	Rr	2015	Dry	S	Aerial	60	25W	Rainforest	Burrowing	Medium	Low	None	Ca, Co, Ge
Mexico	Cayo Norte Mayor	29.0	Rr	2012	Dry	S	Aerial	42	25W	Rainforest	Burrowing	Medium	Low	None	Ca, Co, Ge
Mexico	Cayo Norte Menor	15.0	Rr	2012	Dry	S	Aerial	42	25W	Rainforest	Burrowing	Medium	Low	None	Ca, Co, Ge
Mexico	Isabel	82.0	Rr	2009	Dry	S	Aerial	20	25D	Seasonal	Hermit	Low	Low	None	Co, Ge, Jo
Mexico	Muertos	15.0	Mm	2011	Dry	S	Hand	17	25D	Savanna	Burrowing	Medium	Low	None	Co, Ge
Mexico	Pájaros	3.0	Mm	2011	Dry	S	Hand	17	25D	Savanna	Hermit	Low	High	None	Co, Ge
Mexico	Pérez	14.0	Rr	2011	Dry	S	Hand	17	25D	Savanna	Hermit	Low	High	None	Co, Ge
Micrionesia	Dekehtik	2.6	Re	2007	Dry	S	Hand	50	25W	Rainforest	Burrowing	High	High	Low	Bi, Ca, Co
Micrionesia	Pein Mal	2.2	Rr	2007	Dry	S	Hand	50	25W	Rainforest	Burrowing	High	High	Low	Bi, Ca, Co
New Caledonia	Double	6.0	Re	2008	Dry	S	Hand	20	20R	Seasonal	None	None	Low	None	Co
New Caledonia	G'I	5.0	Re	1998	Dry	S	Hand*	14	20R	Seasonal	None	Low	Low	None	Co, Ge?
New Caledonia	Laregnere	0.5	Re	1998	Dry	S	Hand*	14	20R	Seasonal	None	Low	Low	None	Co, Ge?
New Caledonia	Mato	5.0	Rr	1998	Dry	S	Hand*	14	20R	Seasonal	None	Low	Low	None	Co, Ge?
New Caledonia	Ndo	17.2	Re	1998	Dry	S	Hand*	14	20R	Seasonal	None	Low	Low	None	Co, Ge?
New Caledonia	Nge	7.0	Re	1998	Dry	S	Hand*	14	20R	Seasonal	None	Low	Low	None	Co, Ge?
New Caledonia	Redika	7.0	Re	1998	Dry	S	Hand*	14	20R	Seasonal	None	Low	Low	None	Co, Ge?
New Caledonia	Signal	6.0	Re	1998	Dry	S	Hand*	14	20R	Seasonal	None	Low	Low	None	Co, Ge?
New Caledonia	Table	11.5	Rr	2008	Dry	S	Hand	20	20R	Seasonal	None	None	Low	None	Co
New Caledonia	Tiam'bouene	17.0	Re	2008	Dry	S	Hand	20	20R	Seasonal	None	None	Low	None	Co
New Caledonia	Uatermbi	1.0	Re	1998	Dry	S	Hand*	14	20R	Seasonal	None	Low	Low	None	Co, Ge?
New Caledonia	Uatio	5.0	Re	1998	Dry	S	Hand*	14	20R	Seasonal	None	Low	Low	None	Co, Ge?
New Caledonia	Uie	2.0	Re	1998	Dry	S	Hand*	14	20R	Seasonal	None	Low	Low	None	Co, Ge?
New Caledonia	Uo	3.0	Re	1998	Dry	S	Hand*	14	20R	Seasonal	None	Low	Low	None	Co, Ge?

Appendix 1 (continued) Details of rodent eradications on tropical islands with land crabs, up to 2015, conducted either by aerial broadcast (Aerial) or hand broadcast (Hand; Hand*= bait piles on the ground on a grid) of bait. Target species: Mm= *Mus musculus*, Re= *Rattus exulans*, Rn= *Rattus norvegicus*, Rr= *Rattus rattus*, Rt= *Rattus tanezumi*; Eradication result: F= failure, S= success; Bait type: 20R= Pestoff 20R, 25D= Bell-25D, 25W= Bell-25W; Crab genera: Bi= *Birgus*, Ca= *Cardisoma*, Co= *Coenobita*, Ge= *Gecarcinus*, Gd= *Gecarcoidea*, Gg= *Geograpsus*, Jo= *Johngartia*, Oc= *Ocypode*. [1]During the eradication; [2]Observed by project managers throughout the project.

Country/ territory	Island	Area (ha)	Target species	Erad. year	Erad. season	Erad. result	Erad. method	Total bait rate (kg/ha)	Bait type	Island type	Main interference	Burrowing crab density[1]	Hermit crab density[1]	Coconut crab density[1]	Land crab genera[2]
New Caledonia	Vua	5.0	Re	1998	Dry	S	Hand*	14	20R	Seasonal	None	Low	Low	None	Co, Ge?
Palau	Fanna	35.0	Re	2009	Dry	F	Hand	50	20R	Rainforest	Burrowing	High	Low	Low	Bi, Co, Gd
Palau	Kayangel	112.0	Re	2011	Wet	F	Hand	25	20R	Rainforest	Hermit	Low	Low	None	Bi, Ca, Co
Pitcairn	Ducie	75.0	Re	1997	Dry	S	Hand	8	20R	Seasonal	Hermit	Low	High	None	Co
Pitcairn	Henderson	4,310.0	Re	2011	Dry	F	Aerial	16	20R	Seasonal	Hermit	Low	Medium	Low	Bi, Co
Pitcairn	Oeno	66.0	Re	1997	Dry	S	Hand	8	20R	Seasonal	Hermit	Low	High	Low	Co
Pitcairn	Pitcairn	476.1	Re	1998	Dry	F	Hand	8	20R	Seasonal	Hermit	Low	Low	Low	Co
Puerto Rico	Desecheo	116.0	Rr	2012	Dry	F	Aerial	26	25D	Savanna	Hermit	Low	High	None	Co, Ge
Seychelles	Conception	69.0	Rn	2007	Dry	S	Aerial	26.7	20R	Rainforest	Hermit	Low	Low	None	Co
Seychelles	Curieuse	289.0	Rr	2000	Dry	S	Aerial	23	20R	Rainforest	Hermit	Low	Medium	None	Co, Jo?
Seychelles	Curieuse	289.0	Mm	2000	Dry	F	Aerial	23	20R	Rainforest	Hermit	Low	Medium	None	Co, Jo?
Seychelles	Denis	133.0	Rr	2000	Dry	F	Aerial	23.6	20R	Rainforest	Hermit	Low	High	None	Co, Jo?
Seychelles	Denis	133.0	Mm	2000	Dry	F	Aerial	23.6	20R	Rainforest	Hermit	Low	High	None	Co, Jo?
Seychelles	Fregate	219.0	Mm	2000	Dry	S	Aerial	35	20R	Rainforest	Hermit	Low	Medium	None	Co, Jo?
Seychelles	Fregate	219.0	Rn	2000	Dry	S	Aerial	35	20R	Rainforest	Hermit	Low	Medium	None	Co, Jo?
Seychelles	Grande Ile	143.0	Rr	2007	Dry	S	Aerial	29.7	20R	Rainforest	Hermit	Low	Medium	None	Co, Jo?
Seychelles	Grande Polyte	21.0	Rr	2007	Dry	S	Aerial	29.7	20R	Rainforest	Hermit	Low	Medium	None	Co, Jo?
Seychelles	Grande Soeur	105.2	Rr	2010	Dry	S	Aerial	35.9	20R	Rainforest	Hermit	Low	Low	None	Co, Jo?
Seychelles	Ile aux Rats	1.0	Rr	2005	Wet	S	Hand	15	20R	Rainforest	Hermit	Low	Low	None	Co, Jo?
Seychelles	North	201.0	Rr	2003	Dry	S	Aerial	31	20R	Rainforest	Hermit	Low	Medium	None	Co, Jo?
Seychelles	North	201.0	Rr	2005	Dry	S	Aerial	39.9	20R	Rainforest	Hermit	Low	Low	None	Co, Jo?
Seychelles	Petit Polyte	1.0	Rr	2007	Dry	S	Aerial	29.7	20R	Rainforest	Hermit	Low	Medium	None	Co, Jo?
Seychelles	Petite Soeur	48.0	Rr	2010	Dry	S	Aerial	32.1	20R	Rainforest	Hermit	Low	Low	None	Co, Jo?
US Islands	Palmyra	234.9	Rr	2011	Dry	S	Aerial	163	25D	Rainforest	Burrowing	High	High	Medium	Bi, Ca, Co
US Islands	Wake	696.0	Re	2012	Dry	F	Aerial	36	25D	Seasonal	Hermit	Low	Low	None	Ca, Co
US Islands	Wake	696.0	Rt	2012	Dry	S	Aerial	36	25D	Seasonal	Hermit	Low	Low	None	Ca, Co

Trail cameras are a key monitoring tool for determining target and non-target bait-take during rodent removal operations: evidence from Desecheo Island rat eradication

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Abstract Efforts to remove invasive rodents (e.g. *Rattus* spp. and *Mus musculus*) from islands often use toxicant-laced baits containing the anticoagulants brodifacoum or diphacinone. Rodenticide baits are generally delivered through aerial- or hand-broadcast, or in bait stations. These baits are not rodent-specific and are subject to non-target consumption or secondary exposure (e.g. an individual preying upon another individual that has consumed bait). During rodenticide applications, it is generally unknown which animals are visiting and consuming bait; and to quantify this, we recommend using trail cameras (e.g. Reconyx™ motion-activated infra-red) positioned to monitor individual bait pellets. To demonstrate the importance and effectiveness of using trail cameras during such operations, we report results of target (*Rattus rattus*, black rat) and non-target (native land crab, lizard, insect) bait-interactions after an aerial-broadcast of Brodifacoum-25D Conservation to eradicate rats from Desecheo Island, Puerto Rico. During the first five days following bait application, trail cameras (n = 15) revealed that there were 40 incidences of animals contacting bait pellets: 50% rat, 32% hermit crab, 13% *Ameiva* lizard, and 5% insect. Trail cameras provide temporal and spatial information regarding the effectiveness of rodent removal, and the last rat pictured by trail cameras on Desecheo was six days after bait application began. Trail cameras revealed 30 incidences of animals contacting bait pellets 6–20 days after bait application began: 47% hermit crab, 37% *Ameiva* lizard, 13% insect, and 3% black crab. Despite viewing ~69,000 images from trail cameras, lizards were never pictured consuming bait on Desecheo; therefore, any brodifacoum exposure to Desecheo lizards likely occurred via secondary pathways (e.g. consumption of contaminated insects). Scaling up, we estimate that > 75% of the total bait distributed on Desecheo was not consumed by rats. Trail cameras help inform the hazards of rodenticide use and can be easily incorporated into rodent removal operations.

Keywords: aerial rodenticide broadcast, best practice methods, brodifacoum anticoagulant, land crabs, motion-sensing cameras, *Rattus rattus*, risk assessment, tropical dry forest

INTRODUCTION

Invasive species, particularly rodents, are among the greatest threats to native biodiversity on islands. The breadth of flora and fauna that have been extirpated, or are currently threatened, by invasive rats (*Rattus* spp.) and house mice (*Mus musculus*) is extensive (*see* Towns, et al., 2006; St Clair, 2011; Shiels, et al., 2014). The most common method to suppress invasive rodent populations, or eradicate them from islands, is by using toxicant-laced baits such as those containing the anticoagulants brodifacoum, bromadiolone, or diphacinone (Howald, et al., 2007; Duron, et al., 2017). These rodenticide baits are not rodent-specific and are subject to non-target exposure through their direct consumption of the bait (i.e. primary exposure) or by an individual preying upon another individual that has consumed bait directly (i.e. secondary exposure). Until there is a rodent-specific toxicant developed that can be effectively delivered to target rodent species, non-target species that co-habit treatment areas where rodenticides are used may be at risk to exposure and possibly death. Therefore, there is a level of risk involved when using anticoagulant rodenticides that is relevant to livestock managers and pet owners in domestic settings, and to conservationists attempting to protect native species from the negative effects of rodents in natural areas (Hoare & Hare, 2006).

Existing methods for rodenticide risk assessments suggest implementation of non-toxic bait-uptake trials with biomarker-laced bait, and rodenticide residue analysis of native fauna, both of which can be expensive and may require harvesting individuals including those that are threatened or rare (Pott, et al, 2015). Bait uptake trials with biomarkers are important to determine the level of non-target exposure to bait, and subsequently help determine the bait application rates needed at the site. However, such trials are not always used for island-wide rodent eradication attempts (Pott, et al, 2015) and rarely used for rodent suppression projects (Duron, et al., 2017), perhaps in part because such trials are not a requirement for use of the rodenticide product, and they necessitate considerable effort associated with the capture and sampling of the target and non-target animal community. Although expensive and requiring the harvest of native animals, rodenticide residue studies revealed that residues of the used toxicant establish throughout most of the biological food web and often result in some non-target animal mortalities (e.g. Pitt, et al., 2015). The general acceptance of risk associated with rodenticide use is based on the premise that benefits to native wildlife outweigh the costs (i.e. native wildlife populations increase despite losing a few native individuals from toxicant exposure). A recent example in Alaska reviewed by Croll, et al. (2016) demonstrates that the short-term loss of some individuals of native birds following a rat eradication using brodifacoum has been overwhelmed by large increases in types and abundances of native seabirds over the long term.

The use of trail cameras (i.e. motion-triggered infra-red cameras) is an underutilised method to assess risk to non-target animals associated with rodenticide use. Trail cameras are a means of continuously monitoring rodenticide bait for animal interactions without having to be physically present for such observations. Human observations of animals visiting the bait during rodenticide applications are rare, due to the inability to watch more than a few bait pellets at once and the great likelihood of missing certain animals because of their unique behaviours during foraging (e.g. being secretive, nocturnal, or confined

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to particular habitats). Trail cameras can be placed across a variety of habitats, installed to monitor bait for long periods (days to months), and reliably record diurnal and nocturnal visitation while not substantially altering behaviours (some animals can hear or see cameras/functions; Meek, et al., 2014) or harming resident animals (Swan, et al., 2004). When monitoring bait exposure to wildlife, trail cameras may be less expensive than other methods that require capturing or harvesting animals, and do not require animal use permits or animal sampling. Furthermore, the nearly real-time evidence of bait consumption by target and non-target species documented by trail cameras provides the operational staff confidence that the target rodents are consuming the bait, and allows for adjustments to any subsequent rodenticide bait applications or non-target mitigation strategies, if needed.

We propose that trail cameras provide critical information regarding target bait acceptance, effectiveness, and primary non-target bait exposure during rodent removal campaigns, and therefore future rodent removal campaigns should consider employing this tool. To demonstrate how trail cameras can be used effectively to meet such goals, we report the results of a field study associated with a rat eradication project on Desecheo Island, Puerto Rico, where bait take by target (*R. rattus*) and non-target animals (native crab, lizard, insect) were assessed after the aerial-broadcast of Brodifacoum-25D Conservation bait (3 g pellets, 0.0025% brodifacoum). We used trail cameras to assess the proportion of bait that rats and non-target species interacted with, including how much they removed or consumed, during each of the bait applications. We were also interested in documenting the spatial and temporal changes in bait interactions, including when rats were no longer observed visiting baits. We expected rats to be early primary consumers of the bait, and their observation would quickly decline one to two weeks after the first bait application. Because of the high densities of hermit crabs (*Coenobita clypeatus*) on many parts of the island, we expected that their role in bait consumption and removal would be formidable and consistent between applications; yet, we expected much less bait removal and consumption from other non-targets, such as the three endemic lizard species that have mostly insectivorous life-histories, and the few forest birds and seabirds on the island.

METHODS

Study site and animals

Desecheo (18°23′14″N, 67°28′19″W) is a small (1.2 km^2 or 117 ha) island approximately 21 km from the western shore of the main island of Puerto Rico. The terrain is rugged with karst limestone as parent material, and the peak elevation is 218 m. Vegetation is *Bursera simaruba*-dominated forest, shrubland, and grassland. Annual rainfall averages 1020 mm (Seiders, et al., 1972). The island is a U.S. Fish and Wildlife Service (USFWS) National Wildlife Refuge. *Rattus rattus* is abundant on Desecheo, and was first reported in 1912 (Wetmore, 1918). The negative impacts of *R. rattus* to natural areas and native species on tropical islands are well known (Towns, et al., 2006; St Clair, 2011; Shiels & Drake, 2011; Pender, et al., 2013; Shiels, et al., 2013; Shiels, et al., 2014); rats on Desecheo have been observed eating juvenile lizards and suspected of consuming other native species (Draft EA, 2015). Desecheo has three endemic lizards (anole: *Anolis desechensis*, gecko: *Sphaerodactylus levinsi*, ameiva ground lizard: *Ameiva desechensis*) that may be vulnerable to rats. Although non-native goats (*Capra hircus*) and non-native rhesus monkeys (*Macaca mulatta*) were once common to the island, they have been functionally eradicated (Hanson, et al., 2019). Prior to military actions and rhesus monkeys being introduced to the island, Desecheo had one of the largest nesting colonies of brown boobies (*Sula leucogaster*) in Puerto Rico.

Bait application

In March/April 2016 (the dry season), USFWS and Island Conservation (IC) conducted the bait application operation on Desecheo using Brodifacoum-25D Conservation (25 ppm brodifacoum in ~3 g pellets), under a supplemental label specific to the 2016 eradication effort (Will, et al., 2019). Bait was applied aerially at 30–45 kg/ha (depending upon habitat; see Fig. 1) for each of two applications (18 March and 9 April) in 2016. The 2016 rat eradication attempt used application rates two to three times greater of Brodifacoum 25-D Conservation than those used in the unsuccessful 2012 eradication attempt.

Experimental design

There were 11 sites on Desecheo established for monitoring (Table 1; Fig. 1). These sites were chosen to occupy the different habitats and bait application regions (e.g. deflector, coastal overlap, valleys, cliff; Fig. 1) in areas accessible (often near established trails) on the western half of the island; the steeply sloped terrain and cliffs were avoided for safety and logistical concerns. In total, we established four sites in the 'interior' on ridges or slopes, two sites in 'valley floor/bottoms', one 'cliff' site, two sites in the 'deflector' zone, which was immediately inland of the water's edge and high tide line, and two sites in the 'coastal overlap', which was the most inland portion of the deflector zone and the adjacent inland zone (i.e. interior or valley floor/bottom). To consistently describe the habitat at each site, slope and vegetation were described by a single person (A. Shiels) measuring three variables at each of the 11 sites (Table 1).

At each of the 11 sites, we established a single 150 m transect that had flags marking each 10 m along the transect. Transects were established with meter tapes in a straight line that roughly paralleled walking trails. Once within at least 150 m of a targeted habitat (i.e. interior, valley floor/

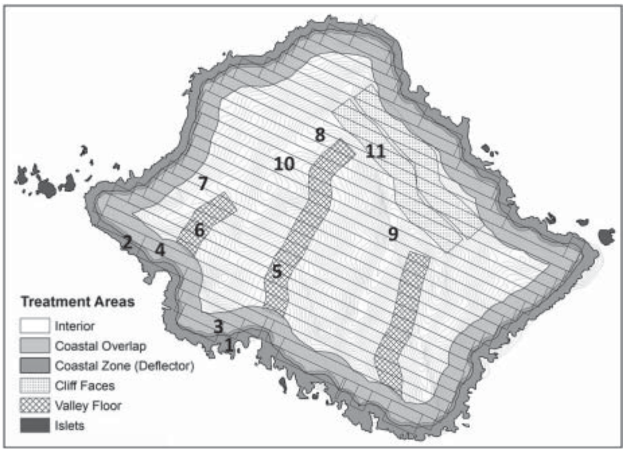

Fig. 1 Map of Desecheo Island, Puerto Rico, outlining the different treatment zones for bait application. The entire island received two applications of Brodifacoum 25D: Conservation rodenticide bait in 2016 (18 March and 9 April). Bait application rates were 30 kg/ha for both applications for all parts of the island except the coastal overlap (#3, #4), cliff faces (#11), and valley floors (#5, #6), which each received a total of 45 kg/ha during both applications. For orientation, there are three main valleys on the island, where (left to right, or west to east) West Valley (containing #6) is the smallest and most western (also where camp was set up at the base), Long Valley is the middle valley (containing #5), and East Valley is the eastern valley. See Table 1 for details of each site.

Table 1 Bait application rates (mean +/- SE bait pellets per m²) and ground cover vegetation (0–1 m height) measured on the ground in 1 m² plots (n = 15 for each site) along 150 m transects on Desecheo Island, Puerto Rico. Target application rates were either 30 kg/ha (equivalent to 1 bait pellet per m²), or 45 kg/ha (equivalent to 1.5 bait pellets per m², and listed in bold), as each pellet weighed 3.06 ± 0.09 g (n = 49).

Site No (see Fig. 1)	Site	Habitat description	Average & (Maximum) Canopy Height (m)	Slope (%)	Application 1 (Pellets/m²) (March 18, 2016)	Application 2 (Pellets/m²) (April 9, 2016)
1	Deflector #1 (coastline of Long Valley [L.V.])	Coastal; rocky with herb/grass	0.2 ± 0.1 (2.5 ± 0.4)	2.4 ± 0.8	1.6 ± 0.4	1.6 ± 0.3
2	Deflector #2 (coastline of West Valley [W.V.])	Coastal; sand with little to no vegetation	0.1 ± 0.0 (0.7 ± 0.3)	4.4 ± 2.2	0.6 ± 0.2	1.6 ± 0.4
3	Coastal Overlap #1 (50–80 m inland of high tide line, L.V.)	Mixed shrubland with herbs, grass, small trees	1.3 ± 0.2 (4.0 ± 0.1)	7.3 ± 1.5	**0.9 ± 0.2**	**1.2 ± 0.3**
4	Coastal Overlap #2 (50–80 m inland of high tide line, W.V.)	Thick grassland and scattered shrubs	0.7 ± 0.1 (3.0 ± 0.2)	4.4 ± 1.5	**1.8 ± 0.2**	**0.7 ± 0.3**
5	Valley Bottom #1 (L.V.)	Forest	3.3 ± 0.1 (7.0 ± 1.1)	15.4 ± 1.9	**0.8 ± 0.2**	**2.1± 0.4**
6	Valley Bottom #2 (W.V.)	Forest	3.5 ± 0.2 (9.3 ± 0.7)	18.4 ± 2.4	**1.4 ± 0.3**	**2.1 ± 0.4**
7	Ridge/Slope #1 (West Ridge of W.V.)	Forest edge and open shrubland	2.6 ± 0.3 (6.9 ± 0.7)	10.4 ± 2.4	1.2 ± 0.3	1.7 ± 0.3
8	Ridge/Slope #2 (Head-slope of L.V.)	Forest	3.1 ± 0.3 (7.8 ± 0.7)	8.0 ± 3.2	1.2 ± 0.3	1.3 ± 0.3
9	Ridge/Slope #3 (Ridge and slope of island peak)	Forest edge and open shrubland	2.3 ± 0.2 (5.4 ± 0.5)	28.1 ± 3.3	1.1 ± 0.2	0.9 ± 0.2
10	Ridge/Slope #4 (Slope of L.V. northwest wall)	Forest	4.2 ± 0.2 (10.4 ± 0.9)	19.6 ± 6.0	0.5 ± 0.2	0.9 ± 0.3
11	Cliff (northeast cliff and windward slope)	Windswept shrubland with herbs and grass	0.8 ± 0.1 (2.9 ± 0.2)	14.3 ± 4.8	**0.7 ± 0.2**	**1.7 ± 0.2**

bottoms, cliff, deflector, coastal overlap), the start of a transect was randomly established by blindly throwing an object over one's shoulder while standing on the walking trail and then beginning the transect from where the object landed. The 10 m interval flagging marked the location of the 1 m² plots for which we monitored bait pellets (15 1 m² plots per transect; 165 total plots for each application at all 11 sites).

A total of 15 trail cameras (12 Reconyx HyperFire models HC500 and HC600, and three Browning Model No: BTC-6HD) were placed to monitor bait pellets to help identify animals visiting and consuming the pellets. Each of the 11 sites always had at least one plot with a trail camera monitoring baits, and some sites had up to three cameras positioned at randomly assigned plots. Only one camera was placed per plot, and each camera was secured to the lower 30–70 cm of a tree or rock. Within 15–120 minutes of the helicopter applying bait to the site, two bait pellets were gathered from the surrounding 2 m² of a respective plot and the trail camera was aimed at the two bait pellets that were placed side-by-side, 40–90 cm away from the camera. A pin-flag was placed next to the two bait pellets in each plot so their presence could be monitored with subsequent visits. All other bait pellets in a 1 m diameter around the pin flag that marked the two target pellets were removed from the area so as not to confuse the observer monitoring pellets. The cameras were set to be triggered by motion, but also were programmed to take a picture each hour (on the hour), and sometimes more frequently (15 or 30 min) at set intervals to help account for periods where bait disappeared or was visited without an animal triggering the camera (e.g. insects rarely trigger these cameras). Once a Reconyx camera was triggered by motion, it would take 10 consecutive pictures over 20 seconds; Browning cameras would take one picture each time triggered.

Cameras were serviced (batteries and SD cards changed, checked for functioning) as needed, and if both bait pellets were removed from a plot with a motion-camera, the camera would be moved to another plot within the site, where bait pellets were still present. Upon activating the cameras on the day of each bait application, the baits and cameras were checked daily for at least seven consecutive days, which was the duration that field staff was on the island; the bait pellets and cameras were also checked at day 20 after the first application because that day preceded the second (and final) application and field staff had returned to the island.

For our analysis, we scored the number of incidences where an animal was observed contacting the bait (i.e. touching, eating, removing). An incidence ended when the animal left the camera's field of view, or when a series of pictures produced by one triggering event ended. The trail cameras monitored for 27 continuous days, which began the first day of application one and ended seven days after application two. Results were presented in three time-periods: 1) application one until the date rats were last observed contacting bait (i.e. day five), 2) days 6–20 post-application one, and 3) the first seven days following application two.

RESULTS

From the 15 cameras deployed, ~38,000 pictures were taken between application one and two (i.e. 20 days of continuous monitoring). We reviewed each picture from all 11 sites, and found 2,686 pictures where an animal was present. Most of the pictures that captured animals showed that they were not in contact with the bait, but instead they were passing by the bait (e.g. ameiva in Fig. 2), or perhaps searching or foraging nearby the bait. Seventy pictures from application one showed an animal in contact with a bait pellet. The first five days following application one was the only time period that rats were observed in contact with the bait (18–22 March), and of the 40 pictures involving animals during this period, 20 were of individual rats (Fig. 3). Although rats dominated bait contact (Fig. 4) during the first five days following bait application (especially so during the first two days), hermit crabs (Fig. 5) comprised 32% of bait contact events (Fig. 3). Most rats and hermit crabs contacting bait either removed it or consumed it in place. Ameivas, which contacted the bait in 13% of the pictures during the first five days, usually had a part of their body (e.g. leg, tail) contacting the bait, or they occasionally touched it with their snout, or on one occasion licked the bait and moved out of the frame. Thus, other than a single lick of the bait, ameivas were never seen consuming (biting, chewing, swallowing) or removing the bait. Finally, there were two insects (one appeared to be a grasshopper) seen in contact with a bait pellet during the first five days following bait application one (Fig. 3).

Fig. 2 An adult ameiva (*Ameiva desechensis*) triggers a trail camera positioned to monitor brodifacoum bait pellets on Desecheo Island, March 2016. Notice the two green bait pellets at the base of a pin-flag at the lower central position of the photo. Ameivas rarely were pictured in contact with the bait and were never documented consuming or removing the bait pellets.

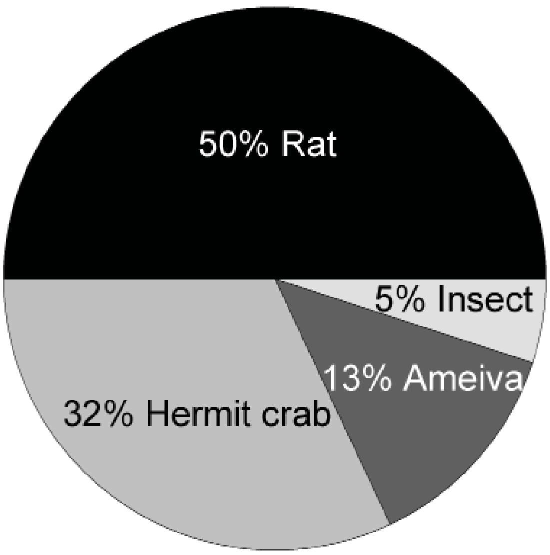

Fig. 3 Percentage of all trail camera results when an animal was in contact with a bait pellet (e.g. touching, eating, removing) during the first five days (18–22 March) after bait application one, on Desecheo Island, Puerto Rico. There was a total of 40 animals in contact with bait during this period (20 rats, 13 hermit crabs, five ameivas, and two insects), and these pictures were taken on the following five sites (Cliff, Overlap #2, Deflector #1, Ridge #4, and Long Valley #1; see Table 1 for site descriptions).

The last day when a rat was captured by motion-cameras on Desecheo was 23 March, which was the sixth day following application one. On this day, there was one rat pictured at Coastal Overlap #2 (grass/shrubland), and one at Ridge #2 (forest). Neither rat came into contact with the bait, but instead passed within 12 cm and 1 m of the bait pellets. These were the last two rats pictured by trail cameras on Desecheo despite the cameras being active and bait present in their field of view through to 15 April 2016.

There were 30 pictures from days 6–20 (23 March–7 April) following application one that showed an animal

Fig. 4 A black rat (*Rattus rattus*) triggers a trail camera positioned to monitor brodifacoum bait pellets on Desecheo Island, March 2016. Notice the bait pellet the rat is nearly touching with its nose. Black rats, being the target species, were pictured consuming and removing the bait pellets for the first five days following the first bait application (18 March 2016).

Fig. 5 A hermit crab (*Coenobita clypeatus*) triggers a trail camera while approaching a bait pellet on Desecheo Island. Hermit crabs were the primary visitors and consumers of bait pellets after the first week following application one.

in contact with a bait pellet. Because rats were no longer present or otherwise not pictured by the trail cameras, the proportion of animals documented contacting the bait shifted (compare Fig. 3 and Fig. 6), such that hermit crabs comprised nearly half (i.e. 14 of 30) of the pictures, and ameivas were pictured contacting the bait in 37% of the pictures during this period (Fig. 6). Insects, primarily grasshoppers, were contacting the bait in four pictures, and there was one picture of a black land crab (*Gecarcinus ruricola*) consuming a bait pellet during this period (Figs 6 & 7).

Sites tended to differ in the types of animals, and their relative abundances, captured on camera contacting bait pellets. In total, there were only five sites following application one that had pictures of animals contacting bait, even though all 11 sites had one to three cameras monitoring bait pellets and all 11 sites had pictures of some animals in the view. For example, the Cliff site only had pictures of hermit crabs contacting bait, whereas the Deflector #1 site only had pictures of insects (primarily grasshoppers) contacting bait (Fig. 8). Coastal overlap #2 and Deflector #1 were the only sites that had rats pictured contacting bait, and Long Valley #1 (valley bottom) and Coastal Overlap #2 were the only sites that had ameivas pictured contacting the bait pellets following application one (Fig. 8). It should be noted here that trail cameras were only monitoring, although continuously, a small subset of the total bait applied to Desecheo (i.e. only about 30 baits; 15 cameras monitoring two baits each).

Bait pellets were monitored during the first seven days following bait application two (Day 21–27), which occurred on 9 April 2016. There were approximately 31,000 pictures taken and reviewed during this period, and 176 pictures contained an animal. Similar to our findings after the first application, most of the pictures that captured animals showed that they were not in contact with the bait. There were 16 incidences where animals were in contact with bait pellets during the week following application two. There tended to be few proportional changes in animal-bait interactions that occurred from the 6–20 days of monitoring after bait application one and the first seven days of bait application two (Day 21–27). Hermit crabs continued to dominate bait interactions, and insect consumption of the bait had risen to the highest proportional levels of all previous measurements (Fig. 9). Ameiva interactions tended to decrease after application two relative to the 6–20 days following application one (Fig. 9). There were five incidences of animals contacting bait pellets during days six and seven: two hermit crabs, two insects, and one ameiva; thus, the first five days of bait interaction would have been similar to the first seven days of bait interaction. Furthermore, the pictures that captured animals interacting with bait occurred at five of the 11 sites (Cliff, Overlap #1, Ridge #1, Ridge #4, and Long Valley #1) during the week following bait application two. As

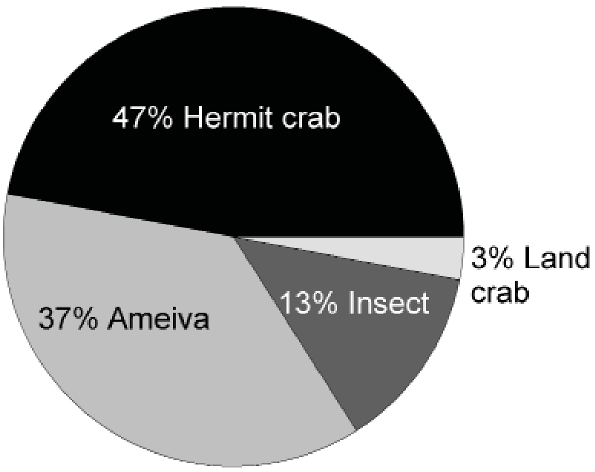

Fig. 6 Percentage of all trail camera results when an animal was in contact with a bait pellet (e.g. touching, eating, removing) during days 6–20 (23 March-7 April) following bait application one, on Desecheo Island, Puerto Rico. There was a total of 30 animals in contact with bait during this period (14 hermit crabs, 11 ameivas, four insects, and one black land crab), and these pictures were taken on the following five sites (Cliff, Overlap #2, Deflector #1, Ridge #4, and Long Valley #1; *see Table 1 for site descriptions*). Note that there were no rats pictured interacting with bait after five days, and rats were not pictured at all after six days following bait application one.

Fig. 7 A black land crab (*Gecarcinus ruricola*) triggers a trail camera while consuming a bait pellet on Desecheo Island. Black land crabs were rarely observed, and only active at night, on Desecheo Island.

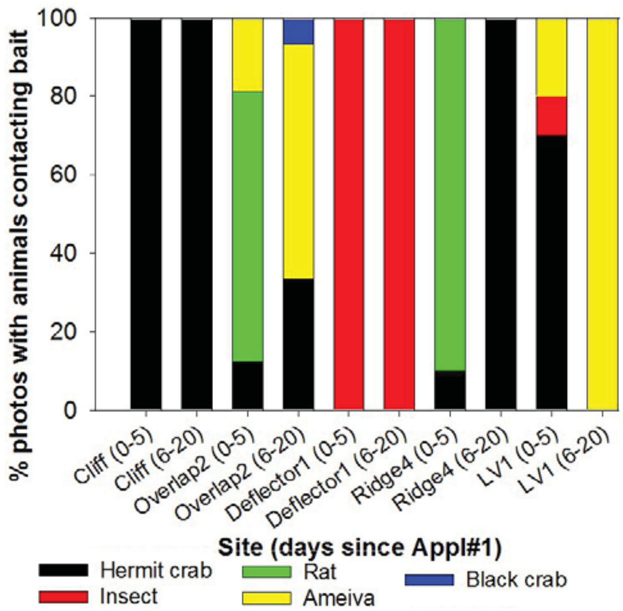

Fig. 8 Percentage of all trail camera results, separated by site, depicting when an animal was in contact with a bait pellet (e.g. touching, eating, removing) during the initial five days (18–22 March), and days 6-20 (23 March-7 April), following bait application one, on Desecheo Island, Puerto Rico. There was a total of 70 animals in contact with bait during this period (i.e. 40 during the initial five days, 30 from 6–20 days), and these pictures were taken at the following five sites (Cliff, Overlap #2, Deflector #1, Ridge #4, and Long Valley #1; *see Table 1 for site descriptions*). Note that there were no rats pictured interacting with bait after five days, and rats were not pictured at all after six days following bait application one.

with all previous pictures, hermit crabs and insects were observed consuming bait pellets, yet ameivas were not seen consuming bait.

DISCUSSION

Trail camera usage during the 2016 rat eradication on Desecheo Island allowed us to quantify, in near "real-time" fashion, the proportional visitation, removal, and consumption of bait pellets, and the timing of such visitation, by target rats and non-target species. Such quantification of bait interactions allows for upscaling to whole habitats and an island-wide understanding of the risks to non-target native species and the potential effectiveness of the eradication campaign at various timescales following initial bait application. Initially, most bait interactions involved rats, and the last rat documented by cameras was on the sixth day after initial bait application. Non-targets that consumed, removed, or otherwise contacted the bait pellets were numerous during the continuous 27 days of cameras monitoring bait pellets on Desecheo, and hermit crabs, ameiva lizards, and insects were the main non-target visitors to the bait pellets. Trail camera usage can therefore better inform rodent removal campaigns of potential animal exposure pathways and confirm target bait acceptance as they are occurring, and therefore should be considered for future rodent control and eradication operations.

Trail cameras revealed that bait was readily consumed by invasive rats on Desecheo during the 2016 rat eradication campaign. Results during the first five days following bait application, when averaged across all monitored habitats, revealed half of the bait that animals on Desecheo interacted (i.e. made contact) with was by rats, and these were most-likely bait consumption events. Without implementing trail cameras to monitor bait pellets, our sole indication that rats were consuming the bait would have not occurred until four days post-application when the first rats turned up dead (Shiels, et al., 2017a). Live rats were rarely observed during the day prior to and following bait application, and bait was never observed being visited by rats without the aid of trail cameras (Shiels, et al., 2017a). Furthermore, carcasses of rats may not always be found because of the expense to keep monitoring crews on the island for extended periods following bait application, rodents suffering from toxicosis often die belowground, and dead rats are quickly scavenged on many islands with a substantial land crab population (Pitt, et al., 2015). Although non-toxic bait uptake trials using biomarkers were performed prior to the 2012 rat eradication attempt on Desecheo (USFWS, 2011), trail cameras provided evidence during the 2016 rat eradication that rats were indeed consuming the bait.

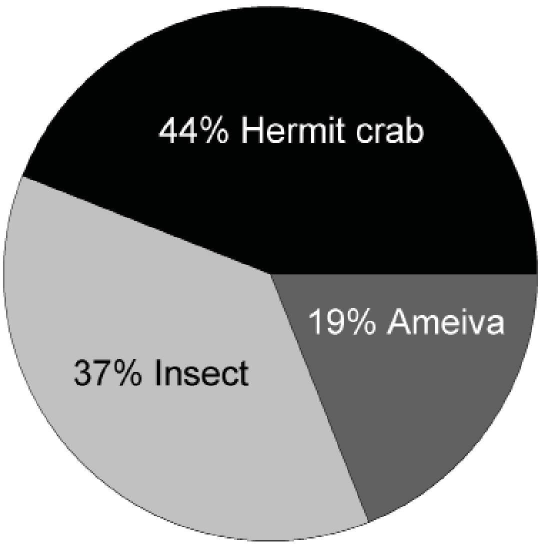

Fig. 9 Percentage of all trail camera results when an animal was in contact with a bait pellet (e.g. touching, eating, removing) during days 0–7 (9–16 April) following bait application two, on Desecheo Island, Puerto Rico. There was a total of 16 animals in contact with bait during this period (seven hermit crabs, three ameiva, six insects), and these pictures were taken at the following five sites (Cliff, Overlap #1, Ridge #1, Ridge #4, and Long Valley #1; *see Table 1 for site descriptions*).

If we use the trail camera findings to scale-up to the whole island, and assume that all pictures with rats contacting the bait resulted in the bait pellet being consumed by the rat, over half of the 5,325 kg of bait that was distributed across Desecheo in application one, and most (or all) of the 5,325 kg of bait in application two, was not consumed by rats. Furthermore, > 75% of the bait applied to Desecheo was consumed by non-target species or did not result in animal consumption (i.e. the bait disintegrated into the soil or was consumed by the microbial community). Clearly, accounting for non-target bait consumption is a critical part of the best practices associated with initial determination of bait application rates for island-wide rat eradications (Pott, et al., 2015). For example, six- to eight-times as much bait as the Brodifacoum 25W: Conservation parent label includes was applied to Palmyra Atoll, in the tropical Pacific, to account for the high density of land crab populations (Pitt, et al., 2015). Land crabs are a well-known non-target species that, like all other invertebrates, are not affected by the brodifacoum toxicant when they consume the bait, but they render the bait unavailable to target rodents (Cuthbert, et al., 2012). Our evidence from trail cameras during the Desecheo rat eradication demonstrates how common non-target bait interactions can be when rodenticides, such as brodifacoum bait pellets, are used for rodent removal. Furthermore, trail cameras revealed the importance of applying additional bait to Desecheo to account for non-targets, primarily hermit crabs, rendering the bait pellets unavailable to rats.

Substantial spatial variation of rat and non-target bait pellet interactions was present during the period following bait application on Desecheo, as bait interactions involving particular animal species tended to differ by habitat (Fig. 8). We must remind the reader that only a very small subset of the bait pellets applied to Desecheo were monitored with trail cameras, and there were far more appearances of animals in the camera view than there were animals that contacted the bait pellets. Additionally, several of the sites that had trail cameras continuously monitoring bait pellets did not have any rats that contacted the bait pellets. The spatial heterogeneity of rat and non-target events in various habitats also highlights the need for trail camera replication, and we feel that our sample size of 15 cameras is modest, and that substantially fewer cameras would be insufficient for an island of size and habitat heterogeneity like Desecheo. Additionally, we benefited from programmed interval-triggering for the cameras that supplemented motion-triggering because this helped capture insects and other small or slow-moving animals that would not trigger the cameras (Newey, et al., 2015). However, the trade-off of programmed interval-triggering, and 10 pictures per triggering, is the added human labour needed to view and analyse the large number of pictures.

Temporal variation of target rodent visitors to bait pellets can inform operational use of the rodenticide, and the trail cameras revealing an absence of rats after six days on Desecheo may suggest modifications to the operation plan to shorten the length of bait availability on the island. However, adjustments to operational plans are generally made to be more conservative (i.e. more bait for longer periods) rather than less conservative.

Our concerns of primary brodifacoum bait exposure to the Desecheo endemic lizard community were abated by the trail camera results, as there was an absence of pictures where lizards were observed consuming bait despite their interactions with the pellets. Additionally, there were no population level impacts to the lizard community as observed by the mark-and-recapture work completed in 2012 (Herrera Giraldo, et al., 2019). Ameivas were the only lizard species that were pictured in contact with the bait pellets during our monitoring, and there was no evidence of bait consumption aside from a single lick of the bait pellet by one individual. Most events where ameivas contacted the bait were by brushing the tail or legs on the pellet when passing by. Ameivas, and the other lizard species on Desecheo, are primarily insectivorous, and are commonly seen foraging in the leaf litter for insects (Shiels, et al., 2017a). Based on brodifacoum residue analysis following bait application, all three endemic lizard species had detectable levels of brodifacoum in their livers or bodies (Shiels, et al., 2017a), and the trail cameras and general diets of these lizards support consumption of contaminated insects as the most-likely pathway for such brodifacoum exposure. Although we could not definitively conclude that insects pictured on the bait pellets were consuming them, at minimum they would have gained exposure to the bait through direct contact, which probably facilitated exposure to higher trophic level predators. We were surprised that birds, particularly pearly-eyed thrashers (*Margarops fuscatus*), were not pictured consuming bait pellets as the few birds collected for residue analysis had evidence of brodifacoum exposure (Shiels, et al., 2017a); however, their omnivorous diet that includes invertebrates and vertebrates (Wetmore, 1916) favours brodifacoum exposure through this secondary pathway.

Trail cameras are a cheaper method than residue analysis to document primary exposure of target and non-target species during rodenticide campaigns. The USDA NWRC Chemistry Unit commonly charges between US$150–US$250 per sample for brodifacoum residue analysis, and this is a comparable fee to other laboratories. Additionally, brodifacoum residue analysis generally takes several weeks to complete. There is a wide price range in trail cameras, but some of the least expensive trail cameras can be purchased for <US$100 per camera (e.g. see https://www.amazon.com/). Inexpensive trail cameras are often adequate for most rodent removal campaigns because these cameras produce an image that is identifiable as a rat or a non-target (e.g. Bushnell brand from 2005 used in Shiels & Drake (2011)); the reliability, quality of the image, and flexibility of the cameras in customising image quality, triggering frequency, and sensitivity are all factors that are generally better in the Reconyx Hyperfire cameras (US$450–US$550 for those used in our study; http://www.reconyx.com/product/Outdoor_Series) than the less expensive alternatives (see Newey, et al. (2015) for a review). An important component that trail cameras cannot easily produce is evidence of secondary exposure of non-target species. One could, however, position rodent carcasses (or non-target carcasses of interest) on the ground such that trail cameras could document the scavengers of those carcasses. The potential brodifacoum exposure of local raptors is worrisome (e.g. Rueda, et al., 2006), and on Desecheo there are only a few resident kestrels, and several non-resident raptor visitors (several species of hawks), that would not be easily observable in their consumption of carcasses or any mortalities that may occur from rodenticide exposure on Desecheo.

Prior to rodenticide use, trail cameras can also help in surveying the potential target and non-target species at a site. Either singly or in combination with non-toxic bait uptake trials (Pott, et al., 2015), trail cameras can inexpensively help identify the potential animals without catching or harming them. Because rodenticide bait pellets are a mostly cereal-grain matrix, setting out 'home-made' mixtures or placing local fruits and seeds on the ground with monitoring cameras (*see* Shiels & Drake, 2011) may be a first step in determining some of the potential animal species that may visit rodenticide baits. This may be applicable for planning purposes, especially on isolated islands where visits to the island may be short or infrequent. Additionally, advanced trail camera technology now allows pictures to be checked remotely, via cellular transmission of the pictures to a cell phone or email account (Eason, et al., 2017).

Additional benefits of using trail cameras include assistance in the confirmation that the target rodent species is indeed the only rodent species on the island. Trail cameras producing high quality pictures, and multiple shots that can reveal multiple angles of the animal, allow for distinguishing features (e.g. tail length, ear size, body size) to be revealed and assessed. Furthermore, there are some occasions where rat-eradications have resulted in surprises such as house mouse populations 'suddenly present', or an explosion in their abundance, due to the mice being masked by the dominance of rats prior to rat eradication (Witmer, et al., 2007); trail cameras would be a viable method to document and act upon such surprises. Trail cameras may also be implemented to assess the particular prey (e.g. fruit and seed) that are most attractive or vulnerable to rodent predation (e.g. Shiels & Drake, 2011), and to document biological change after rodent removal by quantifying before and after native prey survival (e.g. Pender, et al., 2013). On Desecheo, there was a major caterpillar outbreak coinciding with rat removal (Shiels, et al., 2017b), and trail cameras could have been used to better document the development of the outbreak.

The use of trail cameras is an underutilised method of risk assessment for rodenticide use, particularly assessing primary rodenticide exposure that could be a substitute for, or an improvement upon, more expensive methods that require animal handling or sacrifice. Trail cameras can be placed across a variety of habitats, installed to monitor bait for extensive periods (days to months), and reliably record diurnal and nocturnal visitation of target and non-target animals while not substantially altering behaviours or harming resident animals. Trail cameras provide temporal and spatial information regarding the effectiveness of rodent removal, help inform the hazards of rodenticide use, and can be easily incorporated into rodent removal operations.

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REFERENCES

Croll, D.A., Newton, K.M., McKown, M., Holmes, N., Williams, J.C., Young, H.S., Buckelew, S., Wolf, C.A., Howald, G., Bock, M.F., Curl, J.A. and Tershy, B.A. (2016). 'Passive recovery of an island bird community after rodent eradication'. *Biological Invasions* 18: 703–715.

Cuthbert, R.J., de La Brooke, M. and Torr, N. (2012). 'Overcoming hermit-crab interference during rodent-baiting operations: A case study from Henderson Island, South Pacific'. *Wildlife Research* 39: 70–77.

Draft E.A. (2015). *Restoration of Habitat on the Desecheo National Wildlife Refuge Through the Eradication of Non-native Rats, Desecheo, Puerto Rico (Environmental Assessment).* Boquerón, Puerto Rico, USA: USFWS.

Duron, Q., Shiels, A.B. and Vidal, E. (2017). 'Control of invasive rats and priorities for future action'. *Conservation Biology* 31: 761–771.

Eason, C.T., Shapiro, L., Ogilvie, S., King, C. and Clout, M. (2017). 'Trends in the development of mammalian pest control technology in New Zealand'. *New Zealand Journal of Zoology* 44: 267–304.

Hanson, C., Hall, T.J., DeNicola, A.J., Silander, S., Keitt, B. and Campbell, K. (2019). 'Rhesus macaque eradication to restore the ecological integrity of Desecheo National Wildlife Refuge, Puerto Rico'. In: In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 249–255. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Herrera-Giraldo, J.L., Figuerola-Hernandez, C., Swinnerton, K.J., Holmes, N., Bermúdez-Carambot, E.N., Gonzalez-Maya, J.F. and Gomez-Hoyos, D.A. (2019). 'Survival analysis of two endemic lizard species during a rat eradication on Desecheo Island, Puerto Rico'. In: In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 191–195. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Hoare, J.M. and Hare, K.M. (2006) 'The impact of brodifacoum on non-target wildlife: Gaps in knowledge'. *New Zealand Journal of Ecology* 30: 157–167.

Howald, G., Donlan, J.C., Galván, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershey, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Meek, P.D., Ballard, G-A, Fleming, P.J.S., Schaefer, M., Williams, W. and Falzon, G. (2014). 'Camera traps can be heard and seen by animals'. *PLOS ONE* 9(10): e110832.

Newey, S., Davidson, P., Nazir, S., Fairhurst, G., Verdicchio, F., Irvine, R.J. and van der Wall, R. (2015) 'Limitations of recreational camera traps for wildlife management and conservation research: A practitioner's perspective'. *Ambio* 44: 624–635.

Pender, R.J., Shiels, A.B., Bialic-Murphy, L. and Mosher, S.M. (2013). 'Large-scale rodent control reduces pre- and post-dispersal seed predation of the endangered Hawaiian lobeliad, *Cyanea superba* subsp. *superba* (Campanulaceae)'. *Biological Invasions* 15: 213–223.

Pitt, W.C., Berentsen, A.R., Shiels, A.B., Volker, S.F., Eisemann, J.D., Wegmann, A. and Howald, G. (2015). 'Non-target species mortality and the measurement of brodifacoum rodenticide residues after a rat (*Rattus rattus*) eradication on Palmyra Atoll, tropical Pacific'. *Biological Conservation* 185: 36–46.

Pott, M., Wegmann, A.S., Griffiths, R., Samaniego-Herrera, A., Cuthbert, R.J., de L. Brooke, M., Pitt, W.C., Berentsen, A.R., Holmes, N.D., Howald, G.R., Ramos-Rendn, K. and Russell, J.C. (2015). 'Improving the odds: Assessing bait availability before rodent eradications to aid in selecting bait application rates'. *Biological Conservation* 185: 27–35.

Rueda, D., Campbell, K.J., Fisher, P., Cunninghame, F. and Ponder, J.B. (2006). 'Biologically significant residual persistence of brodifacoum in reptiles following invasive rodent eradication, Galapagos Islands, Ecuador'. *Conservation Evidence* 13: 38.

Seiders, V.M., Briggs, R.P. and Glover, L. (1972). 'Geology of Isla Desecheo, Puerto Rico, with notes on the great southern Puerto Rico fault zone and Quaternary stillstands of the sea'. *US Geological Survey Professional Paper* 739.

Shiels, A.B. and Drake D.R. (2011). 'Are introduced rats (*Rattus rattus*) both seed predators and dispersers in Hawaii?'. *Biological Invasions* 13: 883–894.

Shiels, A.B., Flores, C.A., Khamsing, A., Krushelnycky, P.D., Mosher, S.M. and Drake, D.R. (2013). 'Dietary niche differentiation among three species of invasive rodents (*Rattus rattus*, *R. exulans*, *Mus musculus*)'. *Biological Invasions* 15: 1037–1048.

Shiels, A.B., Pitt W.C., Sugihara, R.T. and Witmer, G.W. (2014). 'Biology and impacts of Pacific island invasive species. 11. *Rattus rattus*, the black rat (Rodentia: Muridae)'. *Pacific Science* 68: 145–184.

Shiels, A.B., Witmer, G.W., Samra, C., Moulton, R.S., Ruell E.W., O'Hare J.R., Eisemann J.D., Volker, S.F. and Goldade, D.A. (2017a). *Assessment of Bait Density, Bait Availability, and Non-target Impacts During an Aerial Application of Rodenticide to Eliminate Invasive Rats on Desecheo Island, Puerto Rico*. Final Report QA 2588. Ft Collins, Colorado, USA: USDA, APHIS, WS, NWRC.

Shiels, A.B., Haines, W.P., Swinnerton, K.J., Silander, S., Figuerola-Hernández, C., Will, D., García-Cancel, J.G, and Torres-Santana, C.W. (2017b). 'Sudden appearance and outbreak of *Eunica monima* (Lepidoptera: Nymphalidae) on Desecheo Island, Puerto Rico'. *Florida Entomologist* 100: 176–179.

St Clair, J.J.H. (2011). 'The impacts of invasive rodents on island invertebrates'. *Biological Conservation* 144: 68–81.

Swan, C.E., Hass, C.C., Dalton, D.C. and Wolf, S.A. (2004). 'Infrared-triggered cameras for detecting wildlife: an evaluation and review'. *Wildlife Society Bulletin* 32: 357–365.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?'. *Biological Invasions* 8: 863–891.

USFWS. (2011). *Final Environmental Assessment: Desecheo National Wildlife Rat Eradication to Promote Ecosystem Restoration. Puerto Rico: Caribbean Islands National Wildlife Refuge Complex*. Boquerón, Puerto Rico, USA: USFWS.

Wetmore, A. (1916). 'Birds of Puerto Rico'. *U.S. Department of Agriculture Bulletin* 326: 1–140.

Wetmore, A. (1918). 'The birds of Desecheo Island, Porto Rico'. *The Auk* 35: 333–340.

Will, D., Swinnerton, K., Silander, S., Keitt, B., Griffiths, R., Howald, G.R., Figuerola-Hernandez, C.E. and Herrera-Giraldo, J.L. (2019). 'Applying Lessons Learnt from Tropical Rodent Eradications: a Second Attempt to Remove Invasive Rats from Desecheo National Wildlife Refuge, Puerto Rico'. In: In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 154–161. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Witmer, G.W., Boyd, F. and Hillis-Starr, Z. (2007). 'The successful eradication of introduced roof rats (*Rattus rattus*) from Buck Island using diphacinone, followed by an irruption of house mice (*Mus musculus*)'. *Wildlife Research* 34: 108–115.

Rat and lagomorph eradication on two large islands of central Mediterranean: differences in island morphology and consequences on methods, problems and targets

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Abstract Montecristo and Pianosa islands, although approximately equal in surface area (c. 1,000 ha), differ greatly in substrate, human presence, vegetation and altitude (650 m vs. 30 m asl, respectively). The former island hosts one of the largest yelkouan shearwater (*Puffinus yelkouan*) populations in Italy, the latter a depleted remnant of once numerous Scopoli's shearwaters (*Calonectris diomedea*). Two consecutive EU-funded LIFE projects have been designed to protect these seabird populations. On Montecristo, rough and inaccessible, aerial delivery of toxic baits in January-February 2012 eradicated black rats (*Rattus rattus*) and feral rabbits (*Oryctolagus cuniculus*) (originally a non-target species), with no permanent consequences on a local, ancient population of wild goats (*Capra hircus*). Eradication on Pianosa, currently underway (started January 2017), is being performed by ground baiting, delivered by 4,750 dispensers placed on a 50 m × 50 m grid throughout the island. The latter operation is included in a multi-species eradication aimed at several other target species, among which was the brown hare (*Lepus europaeus*), apparently introduced around 1840. Genetic analyses on the first trapped hares showed that this was the last uncontaminated and viable population of *L. europaeus* subsp. *meridiei* in existence. Whether of natural origin or introduced, the commencement of eradication of this population has instead created the awareness of a taxon otherwise unavailable for conservation elsewhere. While both projects address the same conservation issues (protection of shearwater colonies and restoration of natural communities), they differ greatly regarding economic cost, public perception, effort needed to maintain results in the long term and effects on non-target species. In the present paper, specific attention has been paid to the comparison between bait delivering techniques, results obtained, the array of problems originating from the complex regulatory framework and reactions by the general public.

Keywords *Capra hircus, Lepus europaeus meridiei*, Montecristo, *Oryctolagus cuniculus,* Pianosa, *Rattus rattus,* Tuscan Archipelago

INTRODUCTION

We present here two eradications of invasive mammals carried out on Mediterranean islands, directly concerning black rat (*Rattus rattus*), but also affecting two species of lagomorphs (feral rabbit (*Oryctolagus cuniculus*) and brown hare (*Lepus europaeus*)) and the wild goat (*Capra hircus*) (Table 1). These actions are part of a comprehensive recovery programme of nesting areas of seabirds in the Italian islands (Capizzi, et al., 2016). The two islands concerned, Montecristo and Pianosa, belong to the Tuscan Archipelago National Park and are almost equal in size but very different in morphology, vegetation, fauna and human presence. These differences influenced the eradication approaches.

Montecristo is a 1,080 ha island located in the central northern Mediterranean Sea, in an intermediate position between Corsica and the Italian Peninsula (Fig. 1), with a rugged topography and a maximum altitude of 650 m. It is uninhabited, and access is strictly limited. There are very few trails and no roads. As a consequence, the only realistic method to eradicate rats was the aerial distribution of bait. The main conservation target here was the yelkouan shearwater (*Puffinus yelkouan*), with 400–750 breeding pairs whose reproductive success was heavily affected by predation on eggs and chicks by the black rat (Baccetti, et al., 2009). A population of feral rabbits was also present. Given the bait distribution technique chosen, the eradication of this species was considered possible, although unlikely, and was not declared as a project target. The species considered at risk of unwanted mortality (Table 1) by direct consumption of baits (<http://www.montecristo2010.it>) were mainly the yellow-legged gull (*Larus michahellis*) and Montecristo wild goat. The latter was introduced on the island in pre-Roman times from founders that were still at a very early stage of domestication (or just tamed). The species' historical origin, together with its current uniqueness in the western Mediterranean, motivated its role as a flagship species and led to the founding of Montecristo State Nature Reserve in the early 1970s. The cultural/historical value of this peculiar population makes it deserving of appropriate conservation efforts (Gotti, et al., 2014).

Pianosa is a 1,040 ha island, that is entirely flat (< 30 m altitude), 30 km NW of Montecristo (Fig. 1) with a

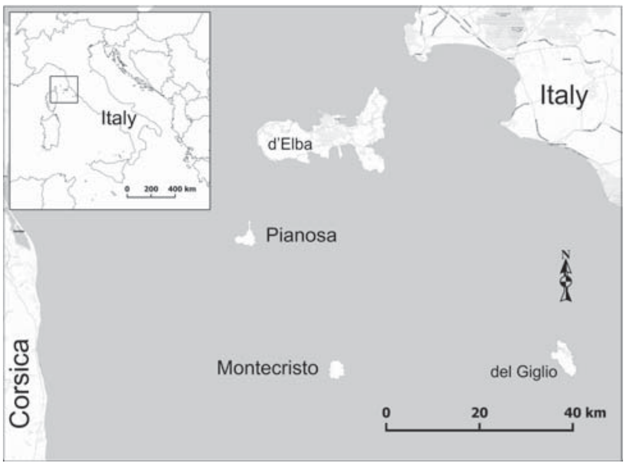

Fig. 1 Location of Montecristo and Pianosa islands.

In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) (2019). Island invasives: scaling up to meet the challenge, pp. 231–235. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

small village and some scattered (and usually abandoned) settlements. It was occupied as a prison until 1998, making it inaccessible even for researchers during that time. Currently it is permanently inhabited by 20–30 detainees with two–three guards and it is open to guided tours with a daily limitation of 330 visitors. The extensive road network is maintained in reasonably good condition; vegetation is relatively accessible, especially on formerly cultivated areas (roughly half of the island). The main conservation target is Scopoli's shearwater (*Calonectris diomedea*), threatened by black rat predation on eggs and chicks (Table 1), consisting of 30–50 nesting pairs on Pianosa and 150–250 on La Scola, a satellite islet located 240 m to the east of Pianosa (Capizzi, et al., 2016). Rats were removed from La Scola, only 1.6 ha in size, in 2001; however, the short distance from the main island allows periodical rat incursions (three in the period 2001–2011, by single individuals), which till now have been successfully eliminated (Capizzi, et al., 2016) by a set of bait stations permanently installed and refilled when necessary.

Black rat eradication on Pianosa is part of a multi-species eradication programme aimed at the restoration of the native animal community <(http://www.restoconlife.eu/en>), which originally included the removal of the brown hare (*Lepus europaeus*). The house mouse (*Mus musculus*) is widespread on the island and is not an explicit eradication target of the project (Table 1), due to the spacing of the bait stations chosen for an island of the size of Pianosa and for the primary target species. The existence of permanent settlements, the presence of tourists during the summer and the occurrence of several non-target species, together with legal constraints on distribution methods, forced the choice of a ground-based eradication. Diurnal raptors, owls, yellow-legged gulls and hooded crows (*Corvus cornix*) and finally some domestic cats (*Felis catus*) are among the non-target species potentially threatened by the operation (Table 1). For non-domesticated feral cats living in the wild another specific eradication action has been conducted.

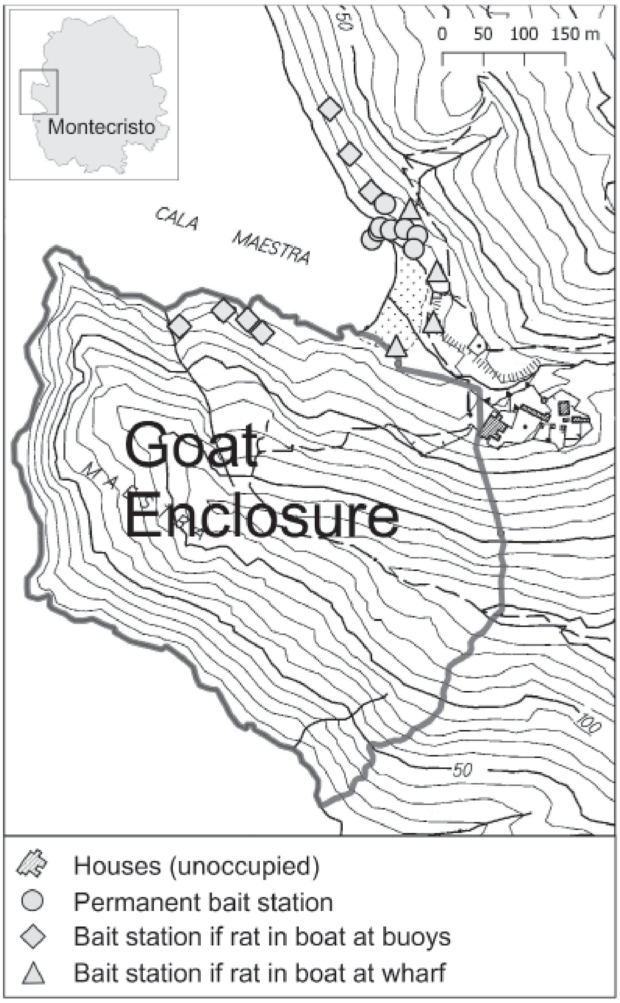

Fig. 2 Location of the goat enclosure, houses, and bait stations involved in the Montecristo biosecurity plan.

METHODS AND RESULTS

Rodent eradications

On Montecristo the first aerial baiting was conducted on 8 and 9 January 2012, with a pellet density of 10 kg/ha on the ground. The baits consisted of 2 g cereal (Brocum®, 0.005% brodifacoum as active ingredient, produced by Colkim Ltd).

A 30 ha area was excluded from the aerial drop (Fig. 2). This included unoccupied human dwellings, and an enclosure of about 25 ha, where 44 wild goats (at least 24% of the population size, assessed through direct counts, the rest having remained free) were kept to ensure survival of the population. This area was treated either by bait stations in the goat enclosure hand-broadcast of pellets outside the goat enclosure. Wax blocks (Solo Blocs®, 0.005% brodifacoum as the active ingredient, produced by Bell Ltd) were installed in the bait-stations.

The aerial distribution was originally planned along parallel transects, 50 m apart, to obtain a roughly complete overlap between parallel transects. However, the pilot had difficulties flying along such predefined routes without veering significantly from the flight line, as the geographic positioning system (GPS) based guidance system malfunctioned and caused frequent interruptions of the baiting. We changed our plans and opted in favour of baiting along two different sets of 100 m wide parallel flight lines, at right angles to each other. This would allow a greater tolerance for the helicopter's distance from the scheduled flight-lines and smaller areas without overlap. We then covered the coast and endeavoured to cover obvious gaps (Fig. 3). The second distribution of baits was initially expected to occur two weeks later, but the exceptionally dry weather that allowed pellets to persist on the ground in good shape and adequate amounts, made it unnecessary for a much longer time. The second baiting was done 45 days after the first delivery, covering 110 ha only, corresponding to the most critical areas: the coastline, a buffer zone around the excluded areas and an area where the first distribution appeared to have been less than optimal. The bait density was lower than in the first

Table 1 Species involved in the Montecristo and Pianosa rat eradication projects. Among non-target species, strong negative impacts on local populations have been recorded on Montecristo for the breeding pair of *Corvus corax* (with permanent reoccupation of the site in 2015–2016) and on Pianosa for *Tyto alba.*

Island	Conservation target species	Invasive species	Invasive non target species	Non target species
Montecristo	*Puffinus yelkouan*	*Rattus rattus*	*Oryctolagus cuniculus*	*Larus michahellis, Capra hircus, Corvus corax*
Pianosa	*Calonectris diomedea*	*Rattus rattus*	*Mus musculus*	*Larus michahellis, Corvus cornix,* owls, diurnal raptors

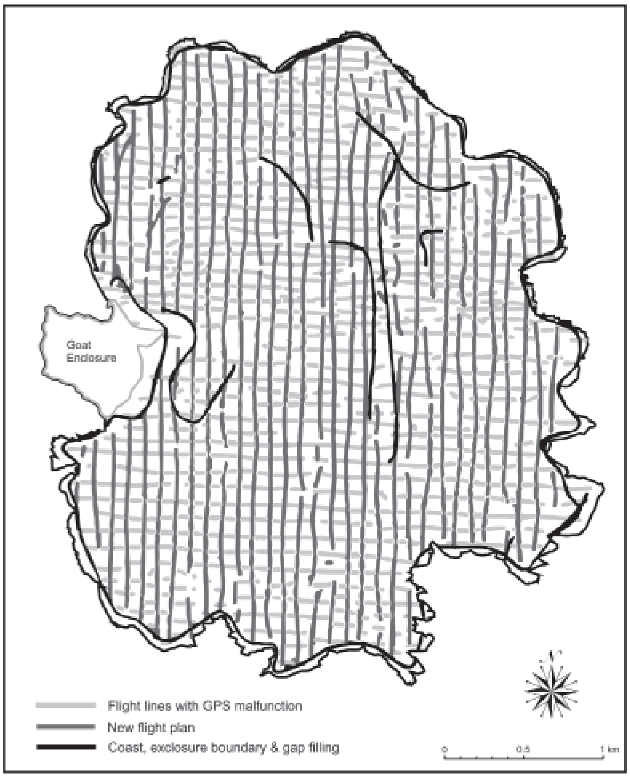

Fig. 3 Helicopter flight lines during the first bait drop.

distribution, i.e. 4 kg/ha on the ground. The eradication was successful, as the last sign of rats was detected 15 days after the first distribution (Sposimo, 2014). The cost of the whole operation, excluding preliminary analyses, planning and devices for protection of goats, was €226,800 (US $280,000).

On Pianosa Island, the eradication took place via ground-baiting with second generation anticoagulants inside bait-stations, placed at the nodes of a 50 m × 50 m grid covering the entire island. Bait density was doubled along the coastline and in urban areas, in consideration of locally higher rat densities. Approximately 4,750 bait-stations were deployed in January 2017, then checked and refilled monthly until May 2017. The percentage of consumed baits and/or any sign of rats were recorded. Bait stations were retrieved in October 2017. The bait consisted of wax-blocs with brodifacoum (Solo Blocs®, 200 or 20 g), except in the area occupied by human settlements where it was replaced by wax-blocks with 0.005% bromadiolone (Notrac Blox®, 225 or 28 g, produced by Bell Ltd) during the first and second baiting events, to reduce the risk of secondary poisoning of domestic animals (Buckle & Smith, 2015).

Rates of bait consumption are detailed in Fig. 4. After the initial and very high rate of bait consumption, the rate decreased by one order of magnitude during each of the two successive sessions; low and steady final values were assumed to be mainly due to house mice and invertebrates. In May, only one credible sign of black rat was found across the entire island, but the success of the eradication has to be confirmed by the implementation of monitoring activities that are still ongoing; the presence of house mouse was detected in nine bait-stations, suggesting that eradication of this species, as expected, is unlikely. This result would be consistent with other evidence of house mice being more difficult to eradicate than *Rattus* species (MacKay, et al., 2007), because of smaller home-range size that allows the survival of some individuals in bait-free areas between 50 m spacing of stations. The cost of the eradication, excluding preliminary analyses and planning, was €477,600 (US $590,351).

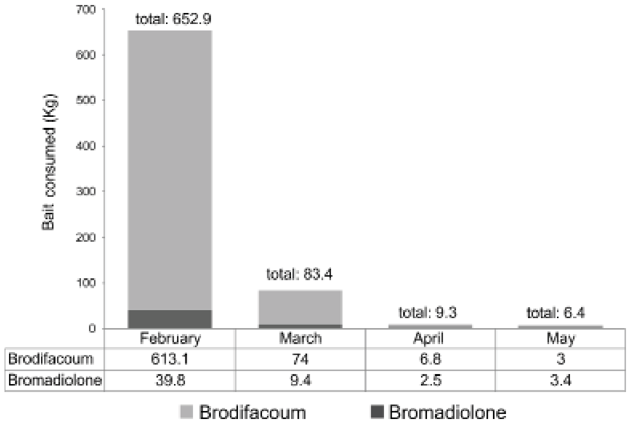

Fig. 4 Bait consumption on Pianosa during the first four sessions.

Lagomorphs

Feral rabbit: the aerial delivery of rat bait took place on Montecristo during mid-winter, when feral rabbits appear to be in the most critical annual phase and when local decreases in the population had been observed. Nevertheless, the presence of a 25 ha fenced area where the bait distribution had been implemented with bait-stations inaccessible to rabbits, and the lack of any specific effort to cull surviving rabbits (Murphy, et al., 2010), made the eradication of this species quite unlikely. Nonetheless, rabbits do seem to have been eradicated, probably due to the rabbit-permeable fencing of the goat enclosure and unusually long duration of pellets outside it, because of the dry weather. After the bait delivery of February 2012 there was just a single observation of rabbits (an adult with young) in April 2014. Traps were immediately set in the observation area without any result. The lack of any further finding of individuals or sign more than three years later makes it unlikely that any survivors might still be present on the island. Despite the lack of any specific monitoring, rabbits would not have escaped detection in the surroundings of the fenced gardens, where full time employed wardens are present, or along the transects regularly covered for goat counts.

Brown hare: the Pianosa multi-species eradication programme originally included netting and shooting brown hares, which had likely been introduced on the island for hunting purposes as early as 1840. Tissue and blood samples obtained from 35 individuals netted at the beginning of the operation, or shot with hunter-dog teams, were collected and genetically analysed, leading to the unpredicted result that the Pianosa brown hare population represented the last uncontaminated and viable population of *Lepus europaeus meridiei*, a subspecies once distributed in central and northern Italy, believed to be extinct due to genotype contamination with exotic hares massively imported in Italy for hunting (Mengoni, et al., 2018). Pianosa's insular status, habitats, and past restricted access due to the prison have resulted in maintaining a viable and representative stock of high conservation value. The number of individuals captured and culled in early 2016 had no consequence on the population size after one to two breeding seasons.

Biosecurity

Reinvasion risk varies greatly between the two islands, because of the different human presence and accessibility (Russell, et al., 2008b). On Montecristo an incoming rat would likely follow a single pathway covering the small area of the wharf and buoys. After the eradication, a biosecurity interception system was set up consisting of 15 bait stations placed all around the entry point at the landing bay. We experimentally tested the effectiveness of this system in December 2016, releasing 14 black rats, all adult males, equipped with VHF transmitters. Animals were released on the pier individually, over 19 hours, simulating a reinvasion event. Each bait station was armed with a lethal snap-trap. Individuals were tracked for 65 hours. Twelve individuals were intercepted by a bait station, the majority (10) within 20 m from the release point, while one disappeared shortly after release (and is suspected of dying from hypothermia after having been observed swimming) and the last one escaped the interception system to hide in a stone wall 100 m inland from the wharf. The average time between release and capture was 3.4 h, which is very short if compared with the results of similar experiments on brown rats (Russell, et al., 2008a). The biosecurity system was modified after these results and organised in three sub-systems (Fig. 2): one permanently active (eight bait-stations), concentrated in the vicinity of the wharf, and two more to be activated in case of potentially rat-infested boats docking at the wharf or buoys. A plan for contingency response has been set-up as well.

Biosecurity measures to be implemented in Pianosa, beginning in October 2017, are directed both towards ferries, to reduce the presence of rodents on board, and towards the implementation of an island-based interception system, roughly following Montecristo's scheme. In the likely case of an unsuccessful mouse eradication, the land system will require a more frequent bait replenishment, together with a permanent mouse-control strategy in the harbour area.

Effects on conservation targets

The effects of the Montecristo eradication on its conservation target species are shown in Table 2, where a dramatic increase in breeding success of yelkouan shearwater has taken place since the year of the bait delivery (2012), whereas breeding performance of Scopoli's shearwater on Pianosa has constantly remained poor. Evidence of new breeding sites, including nest boxes, being occupied by yelkouan shearwaters on Montecristo since 2012 is available, but an increase of nest density or population size remains to be quantified as of yet. A number of benefits were recorded on non-target avian species, such as minor increases of breeding scops owl (*Otus scops*) and European nightjar (*Caprimulgus europaeus*), and an obvious increase of sedentary and alien chukar partridges (*Alectoris chukar*).

Effects on non-target species

Thorough searches for gull and goat corpses were repeatedly carried out on Montecristo from one to four months after bait delivery, where gull casualties were recorded for at least a four month period. On Pianosa, the operators who checked bait-stations every month collected all corpses they found. Standardised counts were performed on both islands to assess any negative effects on non-target populations. Deaths recorded on Montecristo only occurred for two species, the wild goat (n=35) and yellow-legged gull (n=891), while the local pair of common raven (*Corvus corax*) was no longer observed, indicating presumed extirpation. Ravens permanently reoccupied the site only in 2015–16. Annual monitoring of the Montecristo goat population by distance sampling methods showed a temporal decrease of approximately 30–40% in the summer following the aerial treatment, while counts performed in all subsequent years attested to a fast recovery of the population to the pre-eradication level. On the contrary, yellow-legged gulls dropped from 1,036–1,833 breeding pairs in the two years before baits were delivered to 591 in 2012, 292 in 2013, with a steady, slight increase in all following years, up to 499 in 2017.

On Pianosa the impact of rodenticide indirectly consumed by native predators was more diverse, more concentrated in time, but less thoroughly recorded: findings of fresh corpses ceased around mid-March 2017 and included nocturnal raptors of two species (eight individuals), diurnal raptors of three species (seven individuals) and at least three hooded crows; no gulls were affected. Effects on native populations at Pianosa seem to be limited to the expected extirpation of breeding barn owls (*Tyto alba*).

Table 2 Breeding success of the two conservation target species on Montecristo (black rats eradicated 2012) and Pianosa (treated 2017, first rat free season still in progress).

Year	Montecristo target: *Puffinus yelkouan*		Pianosa target: *Calonectris diomedea*	
	No. nests	Reprod. success	No. nests	Reprod. success
2010	18	0.06	-	-
2011	-	-	-	-
2012	19	0.96	6	0.17
2013	28	0.93	-	-
2014	27	0.78	16	0.19
2015	26	0.80	19	0.16
2016	35	0.80	17	0.12

DISCUSSION

Black rats have been successfully eradicated on Montecristo and Pianosa seems to be on a similar trajectory, thanks to two projects performed with radically different techniques.

In order to maintain these achievements – and investments – in the long term, different efforts are needed. Biosecurity measures are relatively simple for Montecristo, as long as the current management of access is allowed. A field test has shown that currently adopted measures are adequate, suggesting minor adjustments enabling a slight reduction of effort needed for their maintenance.

Reinvasion risk is significantly higher for Pianosa, this island being affected by a permanent, yet currently moderate, flow of supplies and visitors, that could strongly increase in the near future due to already planned restoration of many buildings. This, together with the probable survival of the house mouse, results in the need for more complex and costly (due to bait consumption by mice) biosecurity measures. Risks for native species and the insular ecosystem deriving from a house mouse increase following black rat eradication was considered to be low, due to the presence of several species of specialised or generalist predators of rodents (three breeding species of owls and one of snake).

The unexpected disappearance of rabbits from Montecristo can likely be related to timing of the operations, that coincided with seasonal lows of the population, and to random factors such as a very unusual and prolonged drought for the season (January–April rainfall of 34.6 mm in 2012, vs. an average of 112.1 mm in the same period for the previous five years), which allowed longer bait persistence and possibly impacted rabbits more strongly. Similarly, unexpected results for different reasons were obtained in the case of the Pianosa brown hare, representing a taxon believed to be extinct and, thus, deserving appropriate management in future.

Effects on conservation target species were, and are expected to be, very positive. Although this is easily understandable – and already evident for Montecristo yelkouan shearwaters, based on the resulting local population size and productivity – in the case of Pianosa, for Scopoli's shearwaters a full evaluation should include: i) the huge potential for breeding sites, most of them currently unused by the depleted breeding stock; and ii) the effortless maintenance of permanently rat-free conditions on adjacent La Scola islet, where a large Scopoli's shearwater colony is already present. Moreover, since Pianosa is almost devoid of burrowing seabirds at present, but has suitable breeding sites, its value in the future attraction of species that are currently absent (e.g. yelkouan shearwaters from nearby Montecristo and Mediterranean storm petrel [*Hydrobates pelagicus melitensis*]) might even exceed its importance for Scopoli's shearwater.

Consequences on non-target species have varied greatly, depending on bait deployment methods, geo-morphological features and faunal composition of the two islands. Non-target mortality of Montecristo goats did not prevent the recovery of the population to its initial level in a few years and, after the aerial treatments, the widespread presence of goats did not limit the availability of baits for rats. The presence of the large enclosure to protect some goats and prevent population extirpation has to be considered as a prudent measure to ensure the long-term persistence of this valuable population. However, it posed a risk to the success of the rat eradication and demanded an alternative approach (bait delivery inside bait-stations).

The higher mortality of diurnal and nocturnal raptors observed on Pianosa (and probably underestimated) can be attributed to several factors: 1) their greater abundance, 2) the presence of house mice that are preyed upon by small-sized raptors that do not feed ordinarily on rats, and 3) possibly also the delivery of poison through bait stations, that may allow rodents to consume a much higher amount of poison than during an aerial distribution. The most striking difference between the casualties of the two projects was the massive impact of the aerial treatment on the yellow-legged gulls, compared to the absence of any effect on this species in the bait station-based operation. Losses could have been minimised with an aerial delivery planned earlier in the season, when fewer birds are on the breeding sites. Nevertheless, even these losses – of a human-dependent and super-abundant species – are negligible compared to the benefits achieved. A slightly earlier seasonal planning, however, has to be recommended in consideration of possible public reactions to the issue of gull mortality, which was a population decrease lasting more than one year following the operation's conclusion. Losses of barn owls on Pianosa, and their probable (albeit possibly only temporary) extirpation, represent possibly the highest biological cost of the programme.

Both projects triggered negative reactions from the public, particularly harsh towards the Montecristo eradication. Evidently, aerial baiting was perceived as a more threatening method by non-experts. Thus, projects planning to use this technique should employ greater communication efforts at different levels. The strategy for communicating with the public, structured as in many other LIFE projects (e.g. via a dedicated website) was clearly ineffective, despite the projects being on islands lacking human populations which should have restricted the potential audience. People from nearby towns on the coast or from nearby Elba Island were often unresponsive to any outreach efforts and usually exploited debated topics in favour of other agendas, such as anti-Park personal positions, or criticising the 'waste' of money (Baccetti, et al., 2016). Ambiguity of national regulations in force during the Montecristo eradication led to legal actions being taken against project managers, but were finally positively concluded for the defendants. Currently, the EU Biocide Regulation 528/2012 clarifies the situation, but the possibility to carry out aerial baiting remains undefined, depending on specific authorisations issued by nationally competent authorities.

Aerial baiting has allowed the black rat to be eradicated from an island primarily relevant for Mediterranean seabird conservation that could not be otherwise treated with traditional ground-based methods. Tangible drawbacks are not larger than those observed during a comparable operation implemented by a bait station distribution, while the economic cost was certainly lower. Nevertheless, at present, the opportunity to carry out similar operations is extremely uncertain in Italy, as well as across the rest of the EU.

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REFERENCES

Baccetti, N., Capizzi, D., Corbi, F., Massa, B., Nissardi, S., Spano, G. and Sposimo, P. (2009). 'Breeding shearwaters on Italian islands: population size, island selection and co-existence with their main alien predator, the black rat'. *Rivista Italiana di Ornitologia – Research in Ornithology* 78(2): 83–100.

Baccetti, N., Capizzi, D. and Sposimo, P. (2016). 'Rat Eradications on Italian Islands for the Conservation of Breeding Seabirds'. In: P. Yésou, J. Sultana, J. Walmsley and Azafzaf (eds.) *Conservation of Marine and Coastal Birds in the Mediterranean. Proceedings of the UNEP-MAP-RAC/SPA Symposium, Hammamet*, pp.106–113. Tunisia: The Regional Activity Centre for Specially Protected Areas.

Buckle, A.P. and Smith, R.H. (eds.) (2015). Rodent Pests and their Control. Wallingford: CAB International.

Capizzi, D., Baccetti, N. and Sposimo, P. (2016). 'Fifteen Years of Rat Eradication on Italian Islands'. In: F.M. Angelici (ed.) *Problematic Wildlife*, pp. 205–227. Switzerland: Springer International Publishing.

Gotti, C., Palladini, A. and Raganella Pelliccioni, C (2014). 'La Capra di Montecristo una Popolazione Unica in Mediterraneo/ The Montecristo Goat a Unique Mediterranean Population'. In: F. Zanichelli, F. Giannini, F. De Pietro and F. Puppo (eds.) *Quaderni del Parco, Documenti Tecnici, volume 2 PROGETTO LIFE + MONTECRISTO 2010. Eradication of Invasive Alien Plants and Animals and Protection of Native Species and Habitats in the Tuscan Archipelago*, pp. 34–43. Portoferraio: Parco Nazionale Arcipelago Toscano.

MacKay, J.W.B., Russell, J.C. and Murphy, E.C. (2007). 'Eradicating Mice from Islands: Successes, Failures and the Way Forward'. In: G.W. Witmer, C.W. Pitt and K.A. Fagerstone (eds.) *Managing Vertebrate Invasive Species: Proceedings of an International Symposium*, pp. 294–304. Fort Collins: USDA/APHIS/WS, National Wildlife Research Center.

Mengoni, C., Trocchi, V., Mucci, N., Gotti, C., Giannini, F., Mallia, E., Geminiani, C. and Baccetti, N. (2018). 'The secret of Pianosa island: An Italian native population of European brown hare (*Lepus europaeus meridiei* Hilzheimer, 1906)'. *Conservation Genetics*: 10.1007/s10592-018-1077-4.

Murphy, E., Crowell. M. and Henderson, W. (2010). *Workshop Proceedings: Improving the Efficiency of Rabbit Eradications on Islands. 4–5 February 2010, Christchurch*. Canberra, Australia: Invasive Animals Cooperative Research Centre.

Russell, J.C., Beaven, B.M., MacKay, J.W.B., Towns, D.R. and Clout, M.N. (2008a). 'Testing island biosecurity systems for invasive rats'. *Wildlife Research* 35: 215–221.

Russell, J.C., Towns, D.R. and Clout, M.N. (2008b). 'Review of Rat Invasion Biology. Implications for Island Biosecurity'. *Science for Conservation* 286. Wellington, NZ: Department of Conservation.

Sposimo, P. (2014). 'The Eradication of the Black Rat on Montecristo'. pp. 20–25. *I Quaderni del Parco, documenti tecnici volume 2*. Portoferraio: Parco Nazionale Arcipelago Toscano, <www.montecristo2010.it/stealthV3_pubblica/0840566ARG4637055061.pdf>.

Chapter 2: Other taxa

With Sections: A Mammals
B Birds
C Herpetofauna
D Invertebrates
E Plants
F Aquatic

Big island feral cat eradication campaigns: an overview and status update of two significant examples

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Abstract Feral cats have been known to drive numerous extinctions of endemic species on islands. Also, predation by feral cats currently threatens many species listed as critically endangered. Island faunas that have evolved in the absence of predators are particularly susceptible to cat predation. Australian islands, such as Dirk Hartog and Christmas, both formerly known to be high biodiversity islands, are no exception. In this paper we outline the techniques being used in the two eradication campaigns currently underway and provide an update on the status of the programmes on these islands that differ significantly in terms of climate, topography and habitation. Poison baiting and trapping are the methods used on both islands but have been managed differently to suit the local conditions.

Keywords: Christmas Island, Dirk Hartog Island, domestic cat, fence, land crab, planning, poison baiting

INTRODUCTION

There is extensive evidence that domestic cats (*Felis catus*) introduced to offshore and oceanic islands around the world have had deleterious impacts on endemic land vertebrates and breeding bird populations (e.g. Van Aarde, 1980; Moors & Atkinson, 1984; King, 1985; Veitch, 1985; Bloomer & Bester, 1992; Bester, et al., 2002; Keitt, et al., 2002; Pontier, et al., 2002; Blackburn, et al., 2004; Martinez-Gomez & Jacobsen, 2004; Nogales, et al., 2004; Ratcliffe, et al., 2009; Bonnaud, et al., 2010). Feral cats have been known to drive numerous extinctions of endemic species on islands and have contributed to at least 14% of all 238 vertebrate extinctions recorded globally by the IUCN (Nogales, et al., 2013). In addition, predation by feral cats currently threatens 8% of the 464 species listed as critically endangered (Medina, et al., 2011; Nogales, et al., 2013). Island faunas that have evolved for long periods in the absence of predators are particularly susceptible to cat predation (Dickman, 1992). Dirk Hartog and Christmas Islands, both documented as high biodiversity islands are no exception.

Dirk Hartog Island (DHI), an area of 620 km^2, is the largest island off the Western Australian coast (Abbott & Burbidge, 1995). Since the 1860s, DHI has been managed as a pastoral lease grazed by sheep (*Ovis aries*) and goats (*Capra hircus*). More recently, tourism has been the main commercial activity on the island undertaken by the former pastoralist family, the only permanent inhabitants on the island. Cats were probably introduced by early pastoralists and became feral during the late 19th century (Burbidge, 2001). Ten of the 13 species of native terrestrial mammals once present are now locally extinct (Baynes, 1990; McKenzie, et al., 2000) probably due to predation by cats (Burbidge, 2001; Burbidge & Manly, 2002; Algar, et al., 2011a). The house mouse (*Mus musculus*) has become established on the island, but other invasive species such as European rabbit (*Oryctolagus cuniculus*), red fox (*Vulpes vulpes*) and black rat (*Rattus rattus*) are not present.

Christmas Island (CI) occupies an area of 135 km^2 and is famous for the annual migration of tens of millions of red crabs (*Gecarcoidea natalis*) (Orchard, 2012; Misso & West, 2014). CI has a resident multi-cultural population of 2,239 residents (2015 records, <http://www.abs.gov.au/>), predominately Chinese, Malays and Europeans, who reside on the north-eastern tip of the island. Phosphate mining is a major economic driver on the island, with ecotourism becoming increasingly important. Cats were taken to CI at the time of first settlement in 1888 and a feral population established soon thereafter (Tidemann, et al., 1994). Four of the five mammal and two reptile species that were present on the island at settlement have since become extinct, with the introduction of cats playing a crucial role (Beeton, et al., 2010; Martin, et al., 2012). Two endemic rats, the bulldog rat (*Rattus nativitatis*) and Maclear's rat (*R. macleari*) disappeared shortly after black rats were introduced in 1900 (Green, 2014). In addition, several extant CI birds are listed as species likely to be adversely affected by cats (Beeton, et al., 2010).

Across Australia, cats have caused or contributed to population declines and extinctions on many offshore islands (Dickman, 1992; Dickman, 1996; Burbidge, et al., 1997; Burbidge, 1999). Today, the impact of cats is broadly acknowledged and control of feral cats is recognised as one of the most important fauna conservation issues in Australia. As a consequence of this, a national 'Threat Abatement Plan (TAP) for Predation by Feral Cats' has been developed (EA, 1999; DEWHA, 2008; DE, 2015). The TAP seeks to protect affected native species and ecological communities, and to prevent further species and ecological communities from becoming threatened. In particular, the first objective of the TAP is to "prevent feral cats from occupying new areas in Australia and eradicate feral cats from high-conservation-value islands".

DHI was established as a National Park in November 2009, and this now provides the opportunity to reconstruct the native mammal fauna (Algar, et al., 2011a). The island could potentially support one of the most diverse mammal assemblages in Australia and contribute significantly to their long-term conservation. Successful eradication of feral cats is considered to be a necessity prior to reintroductions. Similarly, the impact of cats on much of the biodiversity of CI has been of significant concern to island land management agencies and local residents. Eradication of cats on the island is necessary to mitigate the socio/health impacts and threat to those remaining extant species and to allow successful re-wilding of species such as the blue-tailed skink (*Cryptoblepharus egeriae*) that are currently restricted to captive breeding programmes.

The islands differ markedly in environmental and human factors but are linked in the agencies involved, that have iteratively resolved site-specific challenges associated with the removal of cat impacts on wildlife populations. In this paper we outline the cat eradication programmes currently underway on both islands, describe the strategies, techniques and application methodology and provide an update on the campaigns' progress.

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MATERIALS AND METHODS

Site descriptions

DHI (25° 50’ S, 113° 0.5’ E) lies within the Shark Bay World Heritage Property of Western Australia, 1.5 km from mainland Australia. The island is approximately 79 km long and a maximum of 11 km wide with its long axis in a south-east to north-west direction. Detailed description of geology and vegetation is provided elsewhere (Beard, 1976; Payne, et al., 1987; Algar, et al., 2011a). The climate of the region is ‘semi-desert Mediterranean’ (Beard, 1976; Payne, et al., 1987). The mean annual rainfall for Denham, located 37 km to the east of DHI is 224 mm (Bureau of Meteorology, 2017; long-term records 1893–2016).

CI (10° 25’ S, 105° 40’ E) is located in the Indian Ocean, 360 km south of the Indonesian capital of Jakarta. The oceanic island is composed primarily of Tertiary limestone overlying volcanic andesite and basalt (Tidemann, et al., 1994; EA, 2002). The island consists of a series of fringing limestone terraces, separated by rugged limestone cliffs and scree slopes, rising to an internal central plateau at about 200 m and extending to 360 m above sea level. A National Park was established in 1980 and extended in 1986 and 1989 to include most of the rainforest; it now covers 63% of the island (EA, 2002). There are four main vegetation types described in detail by Claussen (2005). CI has a typical tropical, equatorial climate with a wet and a dry season. The wet season is from December to April when the north-west monsoon blows. For the rest of the year south-east trade winds bring slightly lower temperatures and humidity, and much less rain. The island has a mean annual rainfall of 2,183 mm, high humidity (80–90%) which varies little between months and consistent temperatures (mean daily temperature: 22.9–27.4° C) (Bureau of Meteorology, 2017).

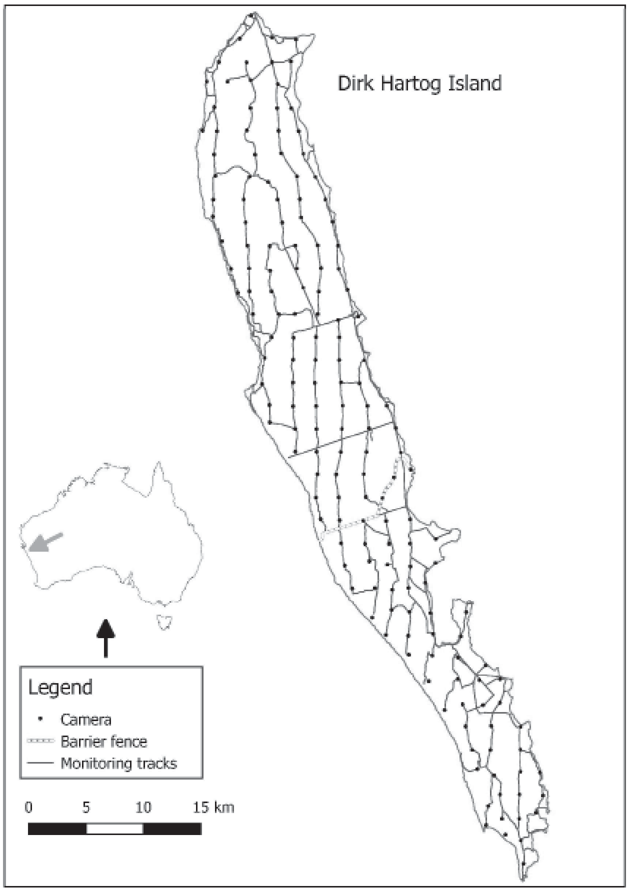

Fig. 1 Dirk Hartog Island.

Planning

To date, feral cats have been successfully eradicated from four Western Australian offshore islands: Serrurier Island (Moro, 1997); Hermite Island in the Montebellos (Algar, et al., 2002); Faure (Algar, et al., 2010) and Rottnest Islands (Algar, et al., 2011b) to enable reconstruction of the original fauna or protection of extant species. These successes and knowledge gained provide the confidence to tackle the more ambitious challenges of DHI and CI. There is a number of key elements used in the operational planning of a successful eradication strategy. The plan may include a pilot study that assesses the efficacy of proposed techniques as well as documenting the procedures to be used in the sequenced eradication phases, the monitoring programmes and the surveillance period prior to verifying eradication has been achieved. Plans for the DHI/CI eradication programmes build strongly on previous research conducted on both islands that examined eradication and monitoring techniques (Algar & Brazell, 2008; Algar, et al., 2010; Algar, et al., 2011a).

Central to the planning for DHI was the construction of a 13 km cat barrier fence. The island’s size, in particular its length, poses logistical constraints on conducting an eradication campaign across the entire island simultaneously. It is not practical to monitor for cat activity over such a large area and therefore, the eradication campaign is being conducted in stages either side of the barrier fence. The fence was constructed with a ‘floppy top’ and electrical hotwires facing to the north to prevent reinvasion of the southern area once it had been cleared (see Fig. 1). Use of a barrier fence has been demonstrated to reduce the cost and increase the overall likelihood of successful eradication on the island (Bode, et al., 2013).

Crucial in the planning for CI was the presence of a domestic cat population. Key land management agencies initiated the preparation of a cat management plan as a critical first step. The plan (Algar & Johnston, 2010) was developed with these agencies, interest groups and the broader community. It was supported and endorsed by the various organisations and has been embraced by the public. Initially, local cat management laws were revised to include a prohibition on the importation of cats, promoting responsible cat ownership, compliance and enforcement of cat management laws. A staged approach to eradicate cats entirely from the island has been adopted, which is complemented by the gradual decrease of owned cats as the de-sexed domestic population dies out. The amended local legislation required all domestic cats to be neutered, micro-chipped and registered with the Shire (Stage 1). Surveys of domestic cats and veterinary programmes are outlined by Algar, et al. (2011c) and Algar, et al. (2014). Stage 2 requires the removal of all stray cats within the township. Without implementation of Stage 2 a significant source of cats, particularly natal recruits, would be available to disperse into or reinvade territories vacated across the rest of the island. Stage 3 involves the implementation of the island-wide (i.e. the national park, mine leases and Unallocated Crown Land) feral cat eradication programme.

Eradication effort

Baits and baiting application

Baiting is recognised as the most effective method for controlling feral cats on mainland Australia (Short, et al., 1997; EA, 1999; Algar & Burrows, 2004; Algar, et al., 2007; Algar, et al., 2013), and has been used as the primary technique for eradicating cats on islands (Algar, et al., 2002; Algar, et al., 2010). World-wide, cat eradications have been attempted on a number of islands with 82 successful campaigns that range in size from 5–29,000 ha (Campbell, et al., 2011). There have also been eradication attempts on a further 15 islands that have failed (ibid.). All successful campaigns on islands >2,500 ha used primary poisoning with toxic baits, with the exception of Santa

Catalina (3,020 ha). Interestingly, seven failed campaigns on the five largest islands (all >400 ha) did not use toxicants (Campbell, et al., 2011). A locally developed bait known as *Eradicat*® (Algar & Burrows, 2004) containing 4.5 mg of directly injected toxin ‘1080’ (sodium monofluoroacetate) is used on both DHI and CI.

The primary eradication technique to be used in the DHI programme was aerial baiting. A pilot study conducted during March–May 2009 assessed the efficacy of this strategy (Johnston, et al., 2010; Algar, et al., 2011a). This achieved very positive results with 80+% of the feral cat population poisoned following bait consumption (ibid). These results demonstrated that a baiting programme, with the *Eradicat*® bait as the primary eradication technique, could be highly effective on DHI.

Deployment of baits from an aircraft was not considered feasible on CI at the commencement of this campaign due to the removal of baits by the abundant land crabs. However, targeted aerial baiting into discrete difficult to access areas is now being contemplated for late in the dry season when land crabs are less active (Johnston, et al., 2016). Preliminary baiting exercises on the island where baits were placed on the ground, highlighted the potential problem of non-target species removing ground-laid baits. Red crabs, robber crabs (*Birgus latro*), which dominate the forest floor, black rats and feral chickens (*Gallus domesticus*) readily removed baits laid on the ground. Bait removal by non-target species reduces bait availability to feral cats and therefore eradication efficacy. In a later trial, Algar & Brazell (2008) demonstrated a device to suspend baits above the ground that effectively reduced bait removal by non-target species yet provided ready access to feral cats. A key finding from this trial was that the bait suspension devices (BSD) would provide an effective primary cat eradication technique on the island. During the eradication campaign, BSD are located at 100 m intervals on both sides of the extensive 160 km road/track, staggered at 50 m intervals across the road/track. Each BSD suspends two *Eradicat*® baits tied at the link, considered a single bait for analysis purposes, at a height of about 550 mm using 6–8 lb fishing line. Baits are replaced when taken and as required to maintain palatability. Suspended baits were also deployed off-track throughout the forest at 50 m intervals in 2015 and, due to unprecedented rainfall, to a lesser extent in 2016.

The total number of toxic baits removed indicates the maximum number of individuals poisoned. The minimum number of individuals poisoned is calculated by ascribing bait removals from consecutive BSDs to the same animal. The actual number of feral cats poisoned would be between these two extremes. While one *Eradicat*® bait contains a lethal dose, it is likely that some cats would visit multiple BSDs given the delay between bait consumption and death.

Trapping

Trapping programmes are being used as the secondary eradication effort to remove those animals that survive the baiting programmes. On DHI, cats are being captured in padded leg-hold traps; (Victor ‘Soft Catch’□ traps No. 3 (Woodstream Corp., Lititz, PA.; U.S.A.) using a mixture of cat faeces and urine as the lure. Trapped cats are destroyed using a 0.22 calibre rifle. All animals captured are sexed and weighed; a broad estimation of age (as either kitten, juvenile or adult) is recorded using weight as a proxy for age. The pregnancy status of females is determined by examining the uterine tissue for embryos. Stomach contents are removed for diet analysis and a sample of hair and tissue taken for DNA microsatellite profiling. Also, prior to the commencement of the two aerial baiting programmes, a number of cats were trapped and fitted with a GPS data-logger/radio-telemetry collar (Sirtrack Ltd, New Zealand). Mortality of radio-collared animals following the baiting programmes was used to provide a measure of baiting efficacy.

On CI, the trapping programme commenced in the township to remove stray cats. Initially, cage traps were used rather than padded leg-hold traps to minimise the risk of injury to domestic cats. Cats were captured using wire cage traps (60 × 20 × 20 cm) with treadle plates (Sheffield Wire Products, Welshpool, Western Australia). All traps were covered with a hessian sack to provide shelter and protection to the captured animals until they could be collected. The traps were usually baited with cooked chicken wings. Outside the township, elevated trap platforms (ETPs) – where trap sets are raised above ground level – are used to exploit cats’ agility and ability to jump, while preventing trap interference from ground-dwelling non-target wildlife such as land crabs. Traps along roads and tracks are generally set on cleaned half 200 l fuel/oil drums in the same configuration and lured as ground sets on DHI.

Monitoring

Monitoring programmes use evidence of actual presence through camera trap images, spotlight records and sign, whether it be footprints, scats or hair, to detect the presence/absence of individuals in an area. In eradication campaigns, monitoring programmes provide information on where further effort is required and whether additional measures and/or resources are needed. A key component of these eradication campaigns is to employ monitoring methods that will provide quantitative estimates of the effectiveness of eradication operations; the techniques must also be capable of detecting animals at low density populations. The physical characteristics of DHI and CI differ significantly and required the adoption of a different suite of monitoring techniques across the two islands.

Of necessity, the monitoring of feral cat activity must be conducted across the entire island. CI has an extensive road/track network (see Fig. 2) whereas, on DHI, much of the former pastoral road network has regenerated, with many roads and fence lines being impassable. The monitoring programme on DHI is being conducted from All Terrain Vehicles (ATVs) which can traverse the entire island in a safe and efficient manner. Prior to implementing the monitoring programme, it was necessary to construct a network of survey tracks to allow monitoring of cat activity across the island. The spacing of these tracks

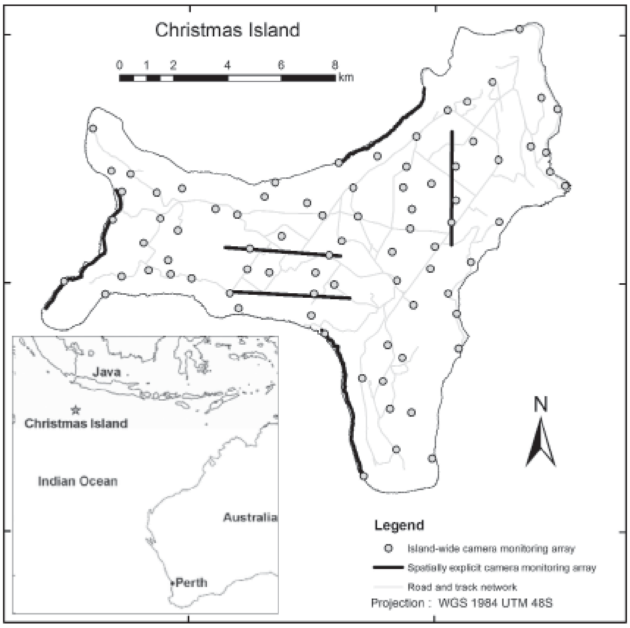

Fig. 2 Christmas Island.

needed to permit detection of any cat during the survey period (i.e. two weeks each month) and therefore provide confidence in the sensitivity of the survey technique. Information obtained from the GPS data-logger radio-collars during the pilot study (Algar, et al., 2011a) was used to determine the likelihood of detection and to optimise the proposed spacing of the survey tracks for the eradication programme. Track lines were parallel to the long axis of the island and the orientation of the dune system. This was the preferred course for survey tracks for logistic reasons and also to minimise disturbance and erosion to dunes. Analysis of daily movement patterns, pooled for all cats, suggested that placement of monitoring tracks at a width of approximately 2.0 km across the full length of the island (see Fig. 1) would be sufficient to enable detection of these animals within each survey period. Choice of this spacing for the monitoring tracks and separation of camera traps (see later) was further strengthened by data collected on home ranges (100% Minimum Convex Polygon) of the radio-collared cats in the pilot study which were 12.7 km^2 for males and 7.8 km^2 for females (Johnston, et al., 2010). Thus, every cat has a very high probability of its sign being observed over a 10-day monitoring period (Algar, et al., 2011a).

Camera trapping

Camera trap studies are useful in providing information on feral cat presence/absence and provide an ideal technique for monitoring the impact of eradication measures through the progression of the eradication campaign as they will allow remote monitoring of cats following each period. On DHI, camera traps were established at 2 km intervals along and overlooking the track network with 105 Reconyx HC600 (Reconyx, Wisconsin; USA) cameras north of the barrier fence and 64 cameras to the south (see Fig. 1). Additional cameras were installed at key locations such as fence ends and around the tourist resort on freehold land. A variety of visual, olfactory or audible attractants were used at camera sites, including no lure. On CI, 84 Scoutguard SG-560C (HCO Outdoors, Norcross, GA, USA) non-lured camera traps were located approximately 1.0 km apart in an island-wide array, with six spatially explicit transects nested within (see Fig. 2). Occupancy analysis and spatial mark/resight modelling was conducted to estimate density over time (The Analytical Edge Pty Ltd., Hobart, Australia).

Sign searches

The sandy surface on DHI enables the search and detection of cat footprints. The network of management tracks is searched daily by skilled observers riding ATVs over a 10-day period on a seasonal basis, that is, four times per year. Circuits ranging in length from 80–140 km, are ridden at a speed of <20 km/h which is adequate to identify footprints on the track surface. The observers alternate the direction of travel and the circuit they inspect on a daily basis. The track surface on CI is hard and does not lend itself to identification of footprints. Other sign monitoring techniques are currently being developed that will complement the use of camera traps to survey for cat activity.

Surveillance period and independent verification

The final phase of the campaign on DHI, an intensive and simultaneous island-wide surveillance period was initiated in October 2016 on the belief that that eradication had been achieved. Assuming no more cats are found, this third phase is expected to be of a two-year duration and will be used to confirm eradication success in October 2018.

On DHI, surveillance monitoring for cat activity is being conducted over a 10-day period in each of the southern and northern sectors every three months. Surveillance monitoring is employing both camera trap recording and cat sign searches. The cat sign searches are being conducted along the pre-existing tracks and the monitoring grid network. Opportunistic cat sign searches along beaches and other areas of interest (e.g. caves and seabird colonies) are also being conducted. The monitoring is undertaken across the entire zone the same day to avoid any issues associated with cat movement.

In addition, on DHI specialist detector dogs and their handlers (Latitude 42 Environmental Consultants Pty Ltd., Tasmania, Australia) have been contracted to further independently verify the absence of cats and corroborate that eradication has been successfully achieved. A team of six dogs and experienced handlers undertake the intensive search effort for cat sign during the winter when weather conditions are the most favourable.

On CI, surveillance monitoring, which is yet to commence, will primarily utilise the island-wide camera array with a range of lures as on DHI. Detector dogs are not being considered for use on CI because of quarantine regulations for re-importation back onto the mainland, the difficult terrain and cultural issues associated with the presence of dogs on the island. A community reporting system will be maintained as well as implementing an intensive and comprehensive spotlighting effort around the island.

Finally, independent verification of eradication success on both islands is to be undertaken by an impartial organisation using data summaries provided.

RESULTS

Dirk Hartog Island

Logistical issues associated with transport of fencing materials prevented construction of the fence on DHI until following the completion of baiting monitoring in 2014. As a result, most of the island (90%) was baited in 2014. However, once completed, the fence alignment has played a key role in restricting the ranging of cats on the northern side.

Data on cat home range size and degree of overlap from the 2009 pilot study were used to derive a best estimate of cat population size pre-eradication effort. This analysis, with multiple assumptions, suggested that a total of 439 cats (range 309–503) was likely present prior to the eradication campaign. Prior to the first baiting campaign in 2014, 17 cats were trapped and fitted with VHF/GPS collars in the southern zone during April 2014. Trapped cats were released at the location of capture. Of these, fifteen were known to be alive when *Eradicat*® baits were applied on the 27–28 May. Fourteen of these animals (>90%) died following bait consumption. The fate of the remaining cat is uncertain but as it was last detected alive in June 2014 and has not be relocated by VHF or photographed since this time. Five cats were trapped, fitted with VHF/GPS collars and released at the location of capture in the northern sector in April 2015 prior to the second baiting programme. All were alive when baits were applied on 25 May 2015. Only one of these cats died following consumption of an *Eradicat*® bait, the remaining four were recovered by trapping. The combined monitoring programmes have detected 36 individual cats that survived the baiting programmes and these animals have subsequently been trapped. January and April seasonal surveillance programmes have failed to detect the presence of any further cat activity. Detector dogs did not locate any fresh sign of cat activity south of the barrier fence in 2016 and examine the area north of the fence during July 2017.

Christmas Island

One hundred and eighty-four domestic owned cats have been registered on CI since 2010, with only 74 domestic cats remaining at the conclusion of the 2017 domestic cat survey. Deregistered cats had either died from natural causes or road fatalities, or were euthanised as the owners had moved off-island. Although the programme on CI commenced in late 2010, funding to commence the island-wide eradication effort (Stage 3) was not secured until 2015. Short-term control programmes were conducted around the township in 2013 and 2014 to protect the significant investment and gains achieved in controlling stray cats until a new source of funding could be obtained. Over the period 2011 to 2015, 336 stray/feral cats were trapped within the township and a further 216–311 were poisoned along roadsides/tracks that surrounded the area. From 2015 to 2017, cage trapping removed 46 stray cats within the township, outside the township limited ETP leg-hold trapping resulted in the removal of (12), shooting (11) and roadside BSDs a further (158–216) cats. An unknown number of cats was removed from forest baiting in 2015 and 2016 due to uncertainty in determining bait uptake by cats. Based on the upper and lower estimation method of baits taken on BSD, between 779–932 stray/feral cats have been removed since 2010. Preliminary results from the 2016 island-wide array camera monitoring estimated that a population of 225 (SE=23) feral cats remains across the island.

DISCUSSION

Globally, the Dirk Hartog project will become the largest island feral cat eradication campaign attempted to date and Christmas Island is a relatively large island with significant human inhabitation. The restoration of former species richness on DHI and recovery of the threatened wildlife populations on CI has required management of feral and domestic cats. The strategies used to achieve the reduction in cat populations have been tailored to suit the specific circumstances applicable to each island. Perhaps the largest challenge on DHI was to ensure that the monitoring tools were sufficiently sensitive given the scale of the island. Removal of cats from Christmas Island is characterised by improving the management of owned cats as well as mitigating the impact of land crabs on poison baiting operations. The guiding principles for successful eradication (Bomford & O'Brien, 1995) have been successfully met in both of these island programmes, although it is worth noting that maintaining the appropriate socio-political environment has been an ongoing and time-consuming component of both programmes. The eradication programmes on both islands have followed a logical progression of intensiveness that aimed to reduce the population rapidly from base levels and then use follow-up trapping to target remaining cats, that is, initial population knockdown with a low cost/broad-scale method followed by high cost/labour intensive mopping up. The monitoring programmes suggest that the cat populations have been reduced to low (CI) or non-detectable (DHI) levels bearing out the prescriptions provided in the operational plans.

Residents on CI have been involved in the development and maintenance of the owned cat population. This has also involved a compliance programme and importation ban that was necessary to maintain the closed population. Maintaining quarantine on DHI has been a more straightforward process given that one family is involved who are invested in the ecological restoration of the island given the anticipated benefits to their tourism enterprise.

Poison baiting has formed a critical part of the eradication tools on both islands but the variable results achieved in these programmes should be noted in preparation for similar programmes in the future. A low baiting efficacy consequently leads to a requirement for greater follow-up control with respect to investments in time and labour. A probable explanation for the observed differences in baiting success in 2014 and 2015 on DHI relates to the meteorological factors at the time of baiting. Just prior to 2014 baiting, a pulse of cooler weather was recorded which would have had the effect of reducing the availability of alternative prey such as small reptiles and mammals. In contrast, the 2015 season was characterised by a rodent irruption that may have been triggered by rainfall associated with Tropical Cyclone Olwyn. On CI, unprecedented rainfall in 2016 reduced baiting efficacy significantly and prompted the development of alternative trap sets that were effective under wet conditions. Alternative removal tools, such as different trap sets, must be ready to implement in situations where baiting is less successful (Robinson, et al., 2015). Project governance and budgeting would ideally include sufficient contingency to adapt or permit operational flexibility to account for environmental factors that influence on-ground outcomes (Springer, 2016).

Ultimately, the success of these programmes will be measured by the response of native wildlife species. On CI, there has been a dramatic increase in the nesting success rate in the red-tailed tropicbird (*Phaethon rubricauda*) populations following improvement in the management of urban cats (Algar, et al., 2012) as well as anecdotal reports of a positive response in forest birds such as the Christmas Island emerald dove (*Chalcophaps indica natalis*). It is premature to make claims about the recovery of extant species on DHI other than to note detections of species on cameras which were not detected in 2014. These include the little long-tailed dunnart (*Sminthopsis dolichura*), painted button quail (*Turnix varius)* and bush stone curlew (*Burhinus grallarius*). The wildlife response on DHI will be intensively monitored in subsequent years during the ecological restoration of the island.

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REFERENCES

Abbott, I. and Burbidge, A.A. (1995). 'The occurrence of mammal species on the islands of Australia: a summary of existing knowledge'. *CALMScience* 1(3): 259–324.

Algar, D., Burbidge, A.A. and Angus, G.J. (2002). 'Cat eradication on the Montebello Islands'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 14–18. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Algar, D. and Burrows, N.D. (2004). 'Feral cat control research: Western Shield review, February 2003'. *Conservation Science Western Australia* 5(2): 131–163.

Algar, D., Angus, G.J., Williams, M.R. and Mellican, A.E. (2007). 'Influence of bait type, weather and prey abundance on bait uptake by feral cats (*Felis catus*) on Peron Peninsula, Western Australia'. *Conservation Science Western Australia* 6(1): 109–149.

Algar, D. and Brazell, R.I. (2008). 'A bait-suspension device for the control of feral cats'. *Wildlife Research* 35: 471–476.

Algar, D., Angus, G.J., Brazell, R.I., Gilbert, C. and Withnell, G.B. (2010). 'Eradication of feral cats on Faure Island, Western Australia'. *Journal of the Royal Society of Western Australia* 93: 133–140.

Algar, D. and Johnston, M. (2010). *Proposed Management Plan for Cats and Black Rats of Christmas Island*. Western Australian Department of Environment and Conservation. 63 pp.

Algar, D., Johnston, M. and Hilmer, S.S. (2011a). 'A pilot study for the proposed eradication of feral cats on Dirk Hartog Island, Western Australia'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 10–16. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Algar D, Angus G.J. and Onus M.L. (2011b). 'Eradication of feral cats on Rottnest Island, Western Australia'. *Journal of the Royal Society of Western Australia* 94: 439–443.

Algar, D., Hilmer, S., Nickels, D. and Nickels, A. (2011c). 'Successful domestic cat neutering: first step towards eradicating cats on Christmas Island for wildlife protection'. *Ecological Management and Restoration* 12(2): 93–101.

Algar, D., Hamilton, N., Holdsworth, M. and Robinson, S. (2012). 'Cat removal implicated in improved nestling success in iconic red-tailed tropicbird rookery'. *Landscope* 27(4): 43–47.

Algar, D., Onus, M.L. and Hamilton, N.A. (2013). 'Feral cat control as part of 'Rangeland Restoration' at Lorna Glen (Matuwa), Western Australia: the first seven years'. *Conservation Science Western Australia* 8(3): 367–381.

Algar, D., Hamilton, N., and Pink, C. (2014). 'Progress in eradicating cats, *Felis catus*, on Christmas Island to conserve biodiversity'. *Raffles Bulletin of Zoology* Supplement No. 30: 44–52.

Baynes, A. (1990). 'The Mammals of Shark Bay, Western Australia'. In: P.F. Berry, S.D. Bradshaw and B.R. Wilson (eds.) *Research in Shark Bay—Report of the France-Australe Bicentenary Expedition Committee,* pp. 313–325. Perth, WA: Western Australian Museum.

Beard, J. (1976). *Vegetation Survey of Western Australia*, *Sheet 6. 1:1 000 000 Map Sheet and Explanatory Notes*. Perth, Australia: UWA Press.

Beeton, B., Burbidge, A., Grigg, G., Harrison, P., How, R., McKenzie, N. and Woinarski, J. (2010). *Final Report of the Christmas Island Expert Working Group to Minister for the Environment Protection, Heritage and the Arts*. Canberra: nt of the Environment, Water, Heritage and the Arts.

Bester, M.N., Bloomer, J.P., Van Aarde, R.J., Erasmus, B.H., van Rensburg, P.J.J, Skinner, J.D., Howell, P.G. and Naude, T.W. (2002). 'A review of the successful eradication of feral cats from sub-Antarctic Marion Island, Southern Indian Ocean'. *South African Journal of Wildlife Research* 32(1): 65–73.

Blackburn, T.M., Cassey, P., Duncan, R.P., Evans, K.L. and Gaston, K.J. (2004). 'Avian extinction and mammalian introductions on oceanic islands'. *Science* 305: 1955–1958.

Bloomer, J.P. and Bester, M.N. (1992). 'Control of feral cats on sub-Antarctic Marion Island, Indian Ocean'. *Biological Conservation* 60: 211–219.

Bode, M., Brennan, K.E.C., Helmstedt, K., Desmond, A., Smia, R. and Algar, D. (2013). 'Interior fences reduce cost and uncertainty when eradicating invasive species from islands'. *Methods in Ecology and Evolution* 4(9): 819–827.

Bomford, M. and O'Brien, P. (1995). 'Eradication or control for Vertebrate Pests'? *Wildlife Society Bulletin.* 23(2): 249–255.

Bonnaud, E., Zarzosa-Lacoste, D., Bourgeois, K., Ruffino, L., Legrand, J. and Vidal, E. (2010). 'Top-predator control on islands boosts endemic prey but not mesopredators'. *Animal Conservation* 13(6): 556–567.

Burbidge, A.A. Williams, M.R. and Abbott, I. (1997). 'Mammals of Australian islands: factors influencing species richness'. *Journal of Biogeography* 24: 703–715.

Burbidge, A.A. (1999). 'Conservation values and management of Australian islands for non-volant mammal conservation'. *Australian Mammalogy* 21: 67–74.

Burbidge, A. (2001). 'Our largest island'. *Landscope* 17(2): 16–22.

Burbidge, A.A. and Manly, B.F.J. (2002). 'Mammal extinctions on Australian islands: causes and conservation implications'. *Journal of Biogeography* 29: 465–473.

Bureau of Meteorology (2017). 'Climate data online'. <http://www.bom.gov.au/climate/cdo/about/cdo-selecting-data.shtml>. Accessed July 2017.

Campbell, K.J., Harper, G., Algar, D., Hanson, C.C., Keitt, B.S. and Robinson, S. (2011). 'Review of feral cat eradications on islands'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 37–46. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Claussen, J. (2005). *Native Plants of Christmas Island*. Perth: Success Print.

DE (Department of the Environment) (2015). *Threat Abatement Plan for Predation by Feral Cats*. Commonwealth of Australia. 47 pp.

DEWHA (Department of the Environment, Water, Heritage and the Arts) (2008). *Threat Abatement Plan for Predation by Feral Cats*. Canberra: Department of the Environment, Water, Heritage and the Arts.

Dickman, C.R. (1992). 'Conservation of Mammals in the Australasian Region: the Importance of Islands'. In: J.N. Coles and J.M. Drew (eds.) *Australia and the Global Environmental Crisis*, pp. 175–214. Canberra: Academy Press.

Dickman, C.R. (1996). *Overview of the Impact of Feral Cats on Australian Native Fauna*. Report to Australian Nature Conservation Agency.

EA (Environment Australia) (1999). *Threat Abatement Plan for Predation by Feral Cats.* Environment Australia, Biodiversity Group, Canberra.

EA (Environment Australia) (2002). *Third Christmas Island National Park Management Plan*. Environment Australia, Canberra.

Green, P. (2014). 'Mammal extinction by introduced infectious disease on Christmas Island (Indian Ocean): The historical context'. *Australian Zoologist* 37: 1–14.

Johnston, M., Algar, D., Onus, M., Hamilton, N., Hilmer, S., Withnell, B. and Koch, K. (2010). *A Bait Efficacy Trial for the Management of Feral Cats on Dirk Hartog Island*. Arthur Rylah Institute for Environmental Research Client Report. Heidelberg, Victoria: Department of Sustainability and Environment.

Johnston, M., McCaldin, G. and Rieker, A. (2016). 'Assessing the availability of aerially delivered baits to feral cats through rainforest canopy using unmanned aircraft'. *Journal of Unmanned Vehicle Systems* 4(4): 276–281.

Keitt, B.S.; Wilcox, C.; Tershy, B.R.; Croll, D.A. and Donlan, C.J. (2002). 'The effect of feral cats on the population viability of black-vented shearwaters (*Puffinus opisthomelas*) on Natividad Island, Mexico'. *Animal Conservation* 5: 217–223.

King, W.B. (1985). 'Island Birds: Will the Future Repeat the Past'? In: P.J. Moors (ed.) *Conservation of Island Birds*, pp. 3–15. International Council for Bird Preservation Technical Publication No. 3. Cambridge, England.

Martin, T.G., Nally, S., Burbidge, A.A., Arnall, S., Garnett, S.T., Hayward, M.W, Lumsden, L.F., Menkhorst, P., McDonald-Madden, E. and Possingham, H.P. (2012). 'Acting fast helps avoid extinction'. *Conservation Letters* 5: 274–280.

Martinez-Gomez, J.E. and Jacobsen, J.K. (2004). 'The conservation status of Townsend's shearwater *Puffinus auricularis auricularis*'. *Biological Conservation* 116: 35–47.

McKenzie, N.L., Hall, N. and Muir, W.P. (2000). 'Non-volant mammals of the southern Carnarvon Basin, Western Australia'. *Records of the Western Australian Museum Supplement No. 6*: 479–510.

Medina, F.M., Bonnaud, E., Vidal, E., Tershy, B.R., Zavaleta, E.S., Donlan, C.J., Keitt, B.S., Le Corre, M., Horwath, S.V. and Nogales, M. (2011). 'A global review of the impacts of invasive cats on island endangered vertebrates'. *Global Change Biology* 17: 3503–3510.

Misso, M. and West, J. (2014). 'Conservation management of the terrestrial biodiversity of Christmas Island: challenges and perspectives'. *Raffles Bulletin of Zoology Supplement* No. 30: 17–23.

Moors, P.J. and Atkinson, I.A.E. (1984). 'Predation on Seabirds by Introduced Animals, and Factors Affecting its Severity'. In: J.P. Croxall, P.J.H. Evans and R.W. Schreiber (eds.) *Status and Conservation of the World's Seabirds*, pp. 667–690. International Council for Bird Preservation Technical Publication No. 2. Cambridge, England.

Moro, D. (1997). 'Removal of a feral cat from Serrurier Island'. *Western Australian Naturalist* 21: 153–156.

Nogales, M., Martin, A., Tershy, B.R., Donlan, C.J., Veitch, D., Puerta, N., Wood, B. and Alonso, J. (2004). 'A review of feral domestic cat eradication on islands'. *Conservation Biology* 18(2): 310–319.

Nogales, M., Vidal, E., Medina, F.M., Bonnaud, E., Tershy, B.R., Campbell, K.J. and Zavaleta, E.S. (2013). 'Feral cats and biodiversity conservation: the urgent prioritization of island management'. *BioScience* 63(10): 804–810.

Orchard, M. (2012). *Crabs of Christmas Island*. Christmas Island Natural History Association. Christmas Island, Indian Ocean, Australia.

Payne, A.; Curry, P. and Spencer, G. (1987). 'An Inventory and Condition Survey of Rangelands in the Carnarvon Basin, Western Australia'. *Technical Bulletin No.73.* Perth, Australia: Western Australian Department of Agriculture.

Pontier, D., Say, L., Debias, F., Bried, J., Thioulouse, J., Micol, T. and Natoli, E. (2002). 'The diet of cats (*Felis catus* L.) at five sites on the Grande Terre, Kerguelen archipelago'. *Polar Biology* 25: 833–837.

Ratcliffe, N., Bell, M., Pelembe, T., Boyle, D., White, R.B.R., Godley, B., Stevenson, J. and Sanders, S. (2009). 'The eradication of feral cats from Ascension Island and its subsequent recolonization by seabirds'. *Oryx* 44(1): 20–29.

Robinson, S., Gadd, L., Johnston, M. and Pauza, M. (2015). 'Long-term protection of important seabird breeding colonies on Tasman Island through eradication of cats'. *New Zealand Journal of Ecology* 39(2): 316–322.

Springer, K. (2016). 'Methodology and challenges of a complex multi-species eradication in the sub-Antarctic and immediate effects of invasive species removal'. *New Zealand Journal of Ecology* 40(2): 273–278.

Short, J., Turner, B., Risbey, D.A. and Carnamah, R. (1997). 'Control of feral cats for nature conservation. II. Population reduction by poisoning'. *Wildlife Research* 24: 703–714.

Tidemann, C.R., Yorkston, H.D. and Russack, A.J. (1994). 'The diet of cats, *Felis catus*, on Christmas Island, Indian Ocean'. *Wildlife Research* 21: 279–286.

Van Aarde, R.J. (1980). 'The diet and feeding behaviour of feral cats, *Felis catus*, on Marion Island'. *South African Journal of Wildlife Research* 10: 123–128.

Veitch, C.R. (1985). 'Methods of eradicating feral cats from offshore islands in New Zealand''. In: P.J. Moors (ed.) *Conservation of Island Birds*, pp. 125–141. International Council for Bird Preservation Technical Publication No. 3. Cambridge, England.

Safeguarding Orkney's native wildlife from non-native invasive stoats

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Abstract The Orkney Islands, off the north-east coast of Scotland, support highly significant cultural and natural heritage. The combined land area of the 70 islands is 990 km^2 (380 sq mi), <1% of the UK, but they host over 20% of the UK's breeding hen harriers (*Circus cyaneus*) (declining over much of its mainland range), 8% of breeding curlews (*Numenius arquata*) (one of only two UK populations not in decline) and an internationally important assemblage of breeding seabirds. The Orkney Islands are naturally free of mammalian predators, and all bird species, including raptors, are ground-nesting in the largely treeless landscape. Rats (*Rattus* spp.), hedgehogs (*Erinaceus europaeus*) and feral cats (*Felis catus*) are present across the archipelago. Stoats (*Mustela erminea*) are native to mainland UK but not Orkney, yet were detected on Orkney Mainland in 2010. Orkney Mainland has an area of 523 km^2 (202 sq mi). Early attempts at removing them were not successful. By 2013 stoats were present across the Orkney Mainland and connected isles. In 2016, SNH and RSPB formed a partnership to eradicate stoats to protect the native wildlife and designated sites of the Orkney islands, and to secure the wider socio-economic and cultural benefits of thriving native wildlife. Difficulties faced in developing the project include predicting the effort required to remove stoats at a rate faster than they can reproduce, securing community support and access to private land and, in particular, funding large scale biodiversity restoration projects. A feasibility study determined that stoat eradication would be possible using DOC200 kill traps, and search dogs in later stages of the eradication. There are no legally available poisons that could be used on stoats in the UK. A Biosecurity Plan has been produced for the archipelago, with a current focus on preventing the spread of stoats to the un-invaded isles. The partnership is working to secure funds and community support for what will be the world's largest stoat eradication attempted to date. We present the findings of the feasibility study and our proposed methodology.

Keywords: biosecurity, feasibility, hen harrier, Orkney vole, predator eradication, Scotland, short-eared owl

INTRODUCTION

The Orkney Islands are situated 10 km from the north-east coast of Caithness, at the northernmost point of mainland Scotland. The archipelago is made up of around 70 islands, of which 20 are inhabited with a total population of around 21,000 residents (National Records of Scotland, 2016). The largest island, Orkney Mainland, is some 523 km^2 and is home to 75% of the human population. The islands of Burray and South Ronaldsay are connected to the Orkney Mainland by causeways carrying road infrastructure. The remaining islands are not physically linked, and are reachable via air or inter-island ferry.

Orkney supports a wide range of natural and cultural heritage for which it is world-famous. There is abundant native wildlife with seabirds, ground-nesting and wading birds, corncrake and sea mammals – all important parts of the ecosystem. Although the islands represent only 0.4% of the UK land area, they are home to a significant proportion of native UK seabirds and terrestrial species. About 14% of the UK breeding kittiwake (*Rissa tridactyla*) population, 34% of arctic skua (*Stercorarius parasiticus*), 10% of puffin (*Fratercula arctica*), 25% of arctic tern (*Sterna paradisaea*) and 14% of the global population of breeding great skuas (*Catharacta skua*) are found in Orkney. Orkney is a UK stronghold for hen harrier (*Circus cyaneus*) and short-eared owl (*Asio flammeus).* Over 20% of the UK population of hen harrier is known to breed in Orkney. The islands are one of the few remaining core areas for breeding corncrake (*Crex crex*) and a stronghold for breeding waders and especially the curlew (*Numenius arquata*) – the UK's highest conservation priority bird species. At a time of large-scale decline across the UK, wader populations in the lowlands of Orkney are thought to be stable or increasing, bucking the national and global trends. Densities in these areas are amongst the highest recorded in Europe.

The quality and importance of Orkney's natural heritage is recognised through the number of nationally and internationally designated sites across the islands. These cover approximately 30% of the islands' land area. There are 13 Special Protection Areas (SPAs), strictly protected sites for rare and vulnerable birds, and for regularly occurring migratory species, under the EU Birds Directive, as well as five Special Areas for Conservation (SACs) which offer strict protection for threatened habitats under the EU Habitats Directive. Orkney also has 36 Sites of Special Scientific Interest (SSSIs), a designation for sites which best represent Scottish natural heritage and are designated under the Nature Conservation (Scotland) Act 2004; one National Scenic Area (NSA), a designation representing Scotland's finest landscapes; two nature conservation Marine Protected Areas (NC MPAs), nature conservation sites in the marine environment designated under the Marine (Scotland) Act 2010; and three proposed marine SPAs. The unique natural and historic heritage of the islands underpins Orkney's distinctive culture and economy and supports a thriving tourism industry.

Although stoats (*Mustela erminea*) are native, widespread and common throughout mainland Britain and Ireland they are not native to the Orkney archipelago. The first confirmed sighting of a stoat in Orkney was reported in August 2010, following verbal reports of possible sightings in June and July of the same year. It is not known how stoats arrived in Orkney; possible vectors include accidental release from imported hay or straw, shipping, or deliberate release (e.g. to control rabbits). Sightings of stoats reported to Scottish Natural Heritage (SNH) have increased in frequency and stoats are now considered to be present across the entire Mainland and linked isles.

This paper examines the risks to Orkney's native wildlife from the impact of predation by stoats and describes efforts to date to deal with the problem. We present the findings of the feasibility study into eradication, our proposed methodology for eradication and outline some of the major challenges to what will be the largest removal of stoats anywhere in the world.

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MATERIALS AND METHODS

This paper summarises the results and discussions arising from stoat sightings across the islands, a desk study conducted to predict the likely impact of stoats on native wildlife, an independent technical feasibility study of stoat eradication (Harper, 2017a), and a Biosecurity Plan (Harper, 2017b) which identifies measures to prevent increase in range and re-colonisation post-eradication.

RESULTS

Likely impact of stoats

When stoats are introduced into ecosystems that have no native mammalian predators, such as those of the Orkney islands, they can have a devastating impact on the native species present. In New Zealand, the stoat is thought to be the main driver of declines and some local extirpations of many native bird populations (Dowding & Murphy, 2001). A desk study that was conducted predicts that the ecological consequences of stoat introduction to Orkney are likely to be devastating (Fraser, et al., 2015). It is highly likely that the presence of stoats on Orkney will have a catastrophic effect on ground nesting birds and mammals on Orkney due to the absence of other mammalian predators including the red fox (*Vulpes vulpes*) "*... it is highly likely that the introduction of stoats will profoundly change the ecology of Orkney and its value for birds of prey and the SPAs that have been designated for protecting these species.*" (Fraser, et al., 2015). Stoats have never been part of the ecosystem in Orkney and therefore many native species, cannot respond rapidly enough to the introduction of this predator. The potential scale and range of the impact of this non-native predator is such that little wildlife in Orkney is currently safe. Impact will be of national and international significance due to the proportion of populations living on the islands.

One critical linkage within the Orkney ecosystem is the predator-prey relationship of the Orkney vole with the hen harrier and short-eared owl. Fraser, et al. (2015) suggest that a decline in Orkney voles will have direct consequences on the hen harrier and short-eared owl populations because both of these species rely to varying extents on the Orkney vole as a component of their diet. The short-eared owl has developed a specialist hunting behaviour to match Orkney vole activity (Reynolds & Gorman, 1999). A range of evidence suggests that the abundance of Orkney voles (which do not display the cyclical population abundance observed in other *Microtus* species) is directly linked to the breeding success of the hen harrier and short-eared owl. It is therefore suggested that significant depredation of voles by introduced stoats will have an indirect detrimental impact on these species. The RSPB Orkney reserves data on numbers of hen harriers fledging suggests a sustained decline which started in 2011 (RSPB 2016, unpublished data). In relation to other species, Fraser, et al. (2015) raise concerns over a range of ground-nesting birds including curlew (another bird on the UK Red list of conservation concern), as stoats are well known to be significant predators, especially where other terrestrial mammalian predators such as foxes are absent, as is the case in Orkney – and so curlew and similar birds are now under severe risk.

Whilst there are other threats to Orkney's native wildlife, including climate change and changes in land management practices, the stoat is considered to be the most pressing and widespread current threat.

Decline in the native wildlife populations is predicted to have a significant effect on socio-economic benefits that Orkney's nature and landscapes provide in terms of tourism and farming. The 2012–2013 (the most recently completed) Islands Visitor Survey (Visit Scotland, 2013) shows that just over 142,800 people visited Orkney, spending over £31 million in the local economy over the period of a year. The main influence on visitors deciding to come to Orkney was an interest in the archaeology and history of Orkney, followed by the scenery and landscape. Given the importance of wildlife tourism to the overall tourism market in Orkney, the predicted declines of many native species caused by stoat predation is a cause for concern amongst tourism businesses.

Stoats are also predicted to affect free-range poultry operations. Free-range poultry farming is common practice in Orkney, where the absence of mammalian predators makes this an economically viable management option for poultry. If stoats continue to persist on the archipelago, future impacts on this industry are expected to include loss of stock to stoat predation as well as the financial impact of implementing predator control and mitigation measures.

Population expansion

No population estimates are available for the stoat population in Orkney, and the available information on their range comes from sightings reported by members of the public. Sightings (both those reported directly and through the 'Stoats in Orkney' Facebook page, maintained by interested local volunteers) have increased steadily since 2010, with a marked increase in sightings since 2016. However, caution must be used when correlating this to any indication of abundance of stoats. Press activity and awareness-raising campaigns by SNH and RSPB Scotland (including posters designed to increase recording of sightings – see Fig. 2) may have increased peoples' awareness of stoats. This could partially lie behind the recent increases in the rate of sightings, although no particular spikes in sightings have been recorded after media activity in the past (Fraser, et al., 2015). The overall distribution map of all sightings (Fig. 1) tends to reflect where people live, work and travel, rather than any accurate estimate of the distribution of stoats per se.

Stoats were first reported in two areas, one on Orkney Mainland and one on South Ronaldsay in 2010. Since this time their numbers and range have increased rapidly and they are now known to be distributed across Mainland

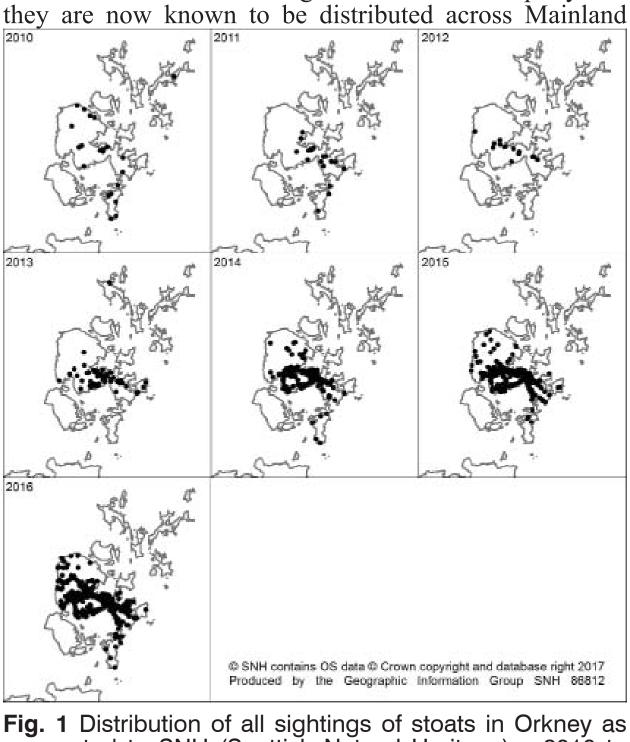

Fig. 1 Distribution of all sightings of stoats in Orkney as reported to SNH (Scottish Natural Heritage) – 2010 to 2016.

Fig. 2 Poster to raise awareness of stoats and their potential impact – and encourage sightings to be reported to SNH (Scottish Natural Heritage).

Orkney and the linked isles of Burray and South Ronaldsay (Fig. 1). Figure 3 shows the clear increase in sightings from 2010–2016. There are obvious seasonal peaks in stoat sightings which would correlate with seasonal activity of the animals (Fig. 4).

Although a number of live and lethal trapping efforts were implemented, they were unsuccessful in completely removing the species or controlling range and population growth. When stoats were first confirmed in 2010 an early 'rapid response' was put in place with volunteers using live traps to remove stoats from the two sites in South Ronaldsay and the west Mainland. Any stoats caught were relocated to mainland Scotland. This was most successful in South Ronaldsay where sightings decreased to two in 2011 and none at all in 2012 and 2013. However, it clearly did not remove the problem on West Mainland where sightings continued to be reported, and eventually increased in numbers and over a wider area.

In 2011–2013, SNH employed a contractor to remove stoats using kill traps mainly on the west Mainland, across an area of some 300 km^2 (115 sq mi). This was unfortunately unsuccessful, possibly due to inexperience of the contractor and/or an inadequate methodology. Fewer than five stoats were caught – nowhere near the numbers required to contain or remove the problem. In 2014, SNH recruited a relatively large number of volunteers from the local community following an awareness raising campaign and increasing concern from interested parties. Over 50 volunteers were trained in the use of kill traps and a trapping project was launched to remove stoats. The purpose of this project was three-fold; in addition to attempting to remove or control the spread of stoats, it was to test whether a large scale volunteer effort was feasible and sustainable in Orkney, and finally to trial approaches to data handling and management for an eradication project.

The volunteering project was ultimately scaled down in 2016 as it had been shown that sustaining a large volunteer force of this size was very resource intensive, few stoats (under 10) were trapped and keeping volunteers motivated when stoats were not being caught was very difficult.

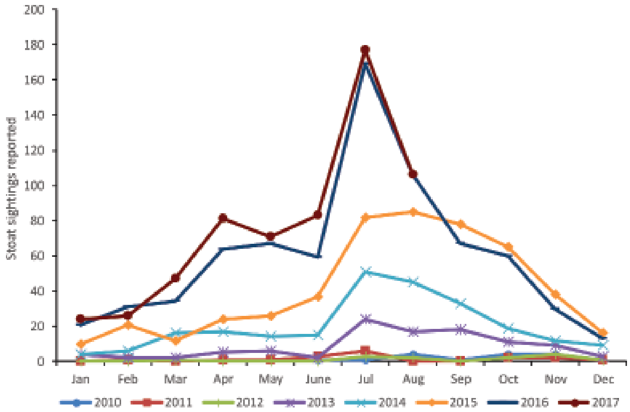

Fig. 3 Number of stoat sightings reported each month on Orkney between June 2010 and June 2017.

Feasibility of stoat eradication

A proposed methodology for eradicating stoats was initially developed by SNH following their experience gained during the Hebridean Mink Project (HMP) and the Uist Wader Project (UWP) which carries out live trapping and translocation of hedgehogs. It was revised following advice from the project's Technical Advisory Group (TAG).

This methodology and the technical, political and environmental feasibility of stoat eradication was assessed independently in an independent Technical Feasibility Study commissioned by RSPB Scotland and completed by Grant Harper of Biodiversity Restoration (Harper, 2017a) following best practice guidelines.

The feasibility study determined that an eradication project was feasible given the current range, but that a new methodology should be adopted. Draft costings were developed for this methodology to determine capacity to eradicate. This methodology has now been assessed by the TAG and adopted by the partnership.

The only legal method to remove stoats in the UK is humane trapping. There are no approved viruses or poisons available for use at present, nor are any likely to be approved within the time frame of eradication. The feasibility study identifies only NZ Department of Conservation (DOC) and Goodnature A24 self-setting traps as AIHTS (Agreement on the International Humane Trapping Standard) compliant in the UK. In Scotland only DOC traps are currently legally compliant. Dogs can be used to locate but not harm or kill an animal.

The legal circumstances surrounding land access are simplified as SNH has the power to issue a Species Control Order which allows them to compulsorily access land in order to control invasive species in the event that landowner permission is withheld. This is of course a last resort, and landowner permission, and the inclusion of landowning interests in the partnership, is of paramount importance.

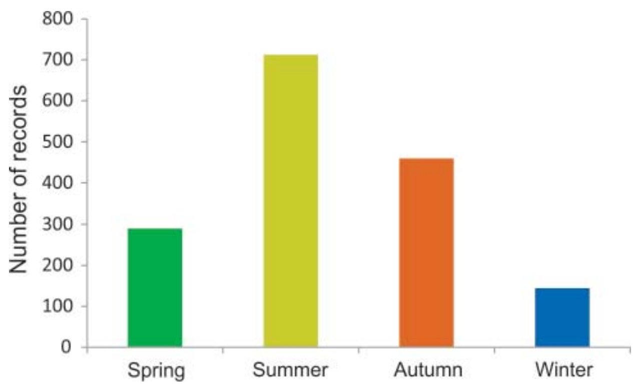

Fig. 4 Seasonality of reported stoat sightings on Orkney.

Environmental acceptability was assessed to determine if the impact of the project on native wildlife can be reduced to an acceptably low level and ensure eradication will not lead to permanent negative changes impact on non-target species. It is accepted that there is always a short-term risk to non-target species, and the project is designed so that this balances out to give a positive long-term change. Harper shows that the species most at risk are rats (which are also non-native to Orkney), but a smaller number of Orkney vole, mice and (potentially) small feral cats are also at risk (Harper, 2017a). However, based on New Zealand data, none of these bycatches are predicted to be of big enough volume to impact population size or stability of non-target species or will be anywhere near the estimated impact of stoats on their long-term populations (Harper, 2017a).

Setting and inspecting traps will inevitably create disturbance in areas of high breeding density for native wildlife. This risk will be minimised in the eradication phase by sensitive placing and minimising disturbance through remote monitoring of trap triggers. The effect of this is also asserted to be much less than the effect of stoats on the native wildlife.

To achieve eradication, a methodology must be implemented that removes animals at a rate faster than they can reproduce and target all of the animals within the population. Stoats in the UK can have home range sizes from 2–254 ha but average about 40 ha. Their home range in Orkney is unknown but a home-range analysis now, when the stoats still have room for expansion, would delay the project unacceptably. A precautionary approach is proposed that works on the assumption of the smallest home range size. This is considered to be required due to the year round food supply, density of food supply and novelty of the predator.

A methodology is proposed that uses a uniform trapping density of 16 traps per km^2 in the first instance. Baited DOC200 kill-traps in standard housing will be used and it is proposed that Goodnature A24 self-resetting traps are also deployed, to target trap-shy individuals that may avoid the DOC traps if these traps are made legal in Scotland. Dogs are also an option to locate trap shy individuals. Within the currently affected area, all land is considered easily accessible and well provisioned with access routes.

This density of trapping gives a total number of traps of 9270, each will be roughly 250 m apart based on a square grid. It is expected to take roughly two months to set these traps, based on 10 trappers setting 20–30 traps per day. Utilisation of habitats is likely to vary and it is expected that, in the largely open farmland habitat, stoats will use field margins that provide both cover and more food. In these habitats, traps will be set according to linear features including fences, walls, ditches, roads and tracks. The proposed grid will be used as a guide for the placement of the traps but trappers will have the discretion to move the traps up to 50 m from the proposed position to the best location on the ground for the interception of stoats.

Trap density and distribution in each habitat type will be reviewed as the project develops using an adaptive management approach.

Draft costings were developed to determine the financial feasibility and the capacity of the relevant organisations to deliver the project in a timely manner. These figures have been further developed by the partnership to ensure social feasibility through community support and involvement. The cost of the project is around £4.5 million. This resource is not available through government so external funding will need to be sourced.

The feasibility study did identify some aspects unique to this project and in particular interaction with the man-made environment. To date stoat eradications have been carried out in largely uninhabited areas so association and behaviour of stoats with areas such as gardens, and farmyards is not known. It is also worth noting that, whilst effectiveness of traps, baits and lures is well documented in New Zealand, this is less well known in the UK and there have been no trials on Orkney.

Biosecurity plan

Finally, the sustainability of the proposed project has been assessed through an RSPB commissioned independent biosecurity plan (Harper, 2017b) to determine if the likelihood of reinvasion is suitably low, or the risks of re-invasion can be reduced sufficiently, through affordable biosecurity measures. All potential invasion and re-invasion risks have been assessed. Two main pathways have been identified for the re-introduction of stoats to Orkney, firstly through intentional introduction and secondly accidentally through haulage of fodder and bedding. A high level of community engagement in an eradication and increased vigilance is considered to reduce the risk of intentional introduction and on-going use of sniffer dogs and an isolation and trapping method for cargoes to sufficiently reduce the risk of accidental re-introduction. It should be noted that this risk is considered small. For decades Orkney has had a high volume of bedding and fodder imported each year, but only the 2010 incident of stoat introduction. Measures can also be taken to reduce the source risk by wrapping and timely movement of bales.

The Biosecurity plan does identify a major risk of expansion to new islands. While no comprehensive survey of stoat population on Orkney has been carried out, the distribution and extent of stoat sightings across the whole of the Mainland and linked isles suggests that the stoat population has ended the invasion phase and it is suggested (Harper, 2017b) that they are at or near carrying capacity within the current range. There are over 60 islands that are still thought to be stoat-free. This means that the situation for stoats on Orkney is at a critical stage, as dispersal to other islands is highly likely. Stoats are thought to be good swimmers, and written accounts exist of stoats swimming 400 m in Ireland, and much further between islands in New Zealand (Veale, 2013), although this has not yet been observed on Orkney. There are many islands within theoretical swimming distance for stoats which could act as staging-posts for dispersing animals onto non-linked isles which are currently biological refuges. Whilst the plan clearly identifies eradication of stoats from the current range as the most effective biosecurity measure to prevent dispersal to new islands it is suggested that trapping on the Orkney Mainland can reduce the risk of spread until eradication can begin. There are five areas of Orkney Mainland coast that have been identified where stoat-free islands are within swimming distance. Immediate deployment of DOC traps in these areas will reduce risk of dispersal. Whilst eradication is considered currently feasible a successful eradication would already be three times greater in area than the largest successful eradication to date. Any extension of range, particularly to islands with less accessible land could make an eradication no longer technically or financially feasible.

DISCUSSION

Due to the native status of the stoat through most of the UK, there is little direct evidence of impact of stoat predation. However, due to the importance of the Orkney islands in a national and international context for wildlife, the SNH commissioned a report, assessing likely impacts of stoats on Orkney's native wildlife through comparison with impacts in other areas of the globe where they are not native. This report clearly demonstrates an ecological imperative to eradicate (Fraser, et al., 2015).

Our learning from early unsuccessful attempts to remove and contain stoats through volunteer response and small contracts has been put to good use in demonstrating that a full scale professional eradication project is required to deal with this issue.

An independent feasibility study (Harper, 2017a) has clearly shown that it is possible to eradicate stoats given the current range of presence across the Mainland and connected isles (as shown in Fig. 1). An assessment of sustainability has shown that we can sufficiently reduce the risk of re-invasion but that we must act now to prevent spread to other islands which could threaten feasibility of eradication (Harper, 2017b).

Since this work has been completed the SNH and RSPB Partnership have focused on developing a costed project, applying for funding for both the eradication and ongoing biosecurity measures, developing community support and implementing immediate biosecurity measures to prevent spread to more islands in Orkney.

We have developed a fully-costed plan, called the Orkney Native Wildlife Project, costing £4.5 million. We have applied to the Heritage Lottery Fund for support and are also in the process of developing and submitting two further grant applications for the eradication and supporting ongoing biosecurity measures. RSPB Scotland is about to commence biosecurity trapping measures on the Orkney Mainland, in accordance with the biosecurity plan. These measures will be kept in place until we start eradication. We also have developed a trial trapping phase within project development that will investigate success of traps, lures and baits in different habitats and will fill gaps in knowledge and be used to fine tune our trapping methodology.

The Orkney Native Wildlife Project is unique. It will be the first eradication of stoats in Europe, and also the first project to consider stoat eradication in areas which include urban and rural settlements.

REFERENCES

Dowding, J.E. and Murphy, E.C. (2001). 'The impact of introduced mammals on endemic shorebirds in New Zealand: a conservation perspective'. *Biological Conservation* 99: 47–64.

Fraser, E.J., Lambin, X., McDonald, R.A. and Redpath, S.M. (2015). 'Stoat (*Mustela erminea*) on the Orkney Islands – Assessing Risks to Native Species'. *Scottish Natural Heritage Commissioned Report No. 871.*

Harper, G. (2017a). 'Methodology Review and Assessment of Technical Feasibility of Eradicating Stoats from the Orkney Islands'. *RSPB Scotland commissioned report.*

Harper, G. (2017b). 'Stoat Biosecurity Plan for the Orkney Islands'. *RSPB Scotland Commissioned Report.*

National Records of Scotland (2016). 'Orkney Islands Council Area – Demographic Factsheet'. <https://www.nrscotland.gov.uk/files/statistics/council-area-data-sheets/orkney-islands-factsheet.pdf>. Accessed 15 February, 2018.

Reynolds, P. and Gorman, M.L. (1999). 'The timing of hunting in short-eared owls (*Asio flammeus*) in relation to the activity patterns of Orkney voles (*Microtus arvalis orcadensis*).' *Journal of Zoology* 247: 371–379.

Veale, A.J. (2013). 'Observations of stoats (*Mustela erminea*) swimming'. *New Zealand Journal of Zoology* 40(2): 166–169.

Visit Scotland (2013). 'Orkney Final Report (Islands Visitor Survey 2012–2013)'. *Island Visitor Survey.* <http://www.visitscotland.org/research_and_statistics/visitor_research/all_markets/islands_visitor_survey.aspx>. Accessed 2017.

Rhesus macaque eradication to restore the ecological integrity of Desecheo National Wildlife Refuge, Puerto Rico

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Abstract A non-native introduced population of rhesus macaques (*Macaca mulatta*) was targeted for removal from Desecheo Island (117 ha), Puerto Rico. Macaques were introduced in 1966 and contributed to several plant and animal extirpations. Since their release, three eradication campaigns were unsuccessful at removing the population; a fourth campaign that addressed potential causes for previous failures was declared successful in 2017. Key attributes that led to the success of this campaign included a robust partnership, adequate funding, and skilled field staff with a strong eradication ethic that followed a plan based on eradication theory. Furthermore, the incorporation of modern technology including strategic use of remote camera traps, monitoring of radio-collared Judas animals, night hunting with night vision and thermal rifle scopes, and the use of high-power semi-automatic firearms made eradication feasible due to an increase in the probability of detection and likelihood of removal. Precision shooting and trapping were the primary methods used throughout the campaign. Long-term monitoring using camera traps and observed sign guided a management strategy that adapted over time in response to population density and structure. Lessons learnt include, 1) macaques quickly adjusted their behaviour in response to human presence and removal methods, 2) camera traps and thermal scopes provided high detection likelihood compared to other methods, and 3) the use of Judas animals and night hunting with thermal and night vision rifle-scopes facilitated removals. The removal of macaques from Desecheo Island appears to be the first introduced non-hominid primate eradication from an island.

Keywords: conservation, invasive species, island restoration, Judas, *Macaca mulatta,* primate

INTRODUCTION

Islands occupy ~5.5% of Earth's terrestrial surface area but contain more than 15% of terrestrial species (Kier, et al., 2009), 61% of all recently extinct species, and 37% of all critically endangered species on the International Union for Conservation of Nature (IUCN) Red List (Tershy, et al., 2015). Non-hominid primates (NHPs) are intelligent and adaptable animals (Fooden, 2000). World-wide, 78 introduced insular populations are known on 63 islands (Jones, et al., 2018). Despite their potential for ecological impacts, including being implicated in 69 insular species extinctions and extirpations globally (Jones, et al., 2018), management is problematic as NHP's demonstrate behavioural traits making them challenging to remove and few practitioners are experienced in their control or eradication (Evans, 1989; Feild, et al., 1997; Breckon, 2000; Kemp & Burnett, 2003; Strier, 2016; Jones, et al., 2018). Six eradication attempts have been documented globally and all were unsuccessful (Jones, et al., 2018). Desecheo Island (Desecheo), has been the site of half of these attempts targeting a population of invasive rhesus macaques (*Macaca mulatta*).

Historically, Desecheo was a major seabird rookery. In the early 1900s tens of thousands of seabirds representing seven species were nesting on the island (Bowdish, 1900; Wetmore, 1918; Struthers, 1927; Meier, et al., 1989; Noble & Meier, 1989). The most numerous species, brown boobies (*Sula leucogaster*), numbered 8,000 - 15,000 individuals (Danforth, 1931 cited by Noble & Meier, 1989; Wetmore, 1918) with red-footed boobies (*S. sula*), brown noddies (*Anous stolidus*), and bridled terns (*Sterna anaethetus*) accounting for another 12–14,000 birds. Humans shooting birds and harvesting eggs, habitat destruction through farming, ranching and military munitions training, and introduced feral goats (*Capra hircus*) and black rats (*Rattus rattus*) reduced populations of most seabird species and restricted many species to less accessible areas of the island (Wetmore, 1918; Struthers, 1927; Evans, 1989; Meier, et al., 1989). Feral goats were recently eradicated (2009; Hanson, unpublished data) while black rats were eradicated in 2016 after an initial attempt failed in 2012 (Will, et al., 2019). However, predation by rhesus macaques (macaques), introduced in 1966 for research purposes, resulted in the complete loss of seabird breeding on the island and was considered the most significant threat to wildlife on Desecheo (Evans, 1989; Meier, et al., 1989; Noble & Meier, 1989). In 1969, massive raids by macaques on booby nests were reported, with macaques pushing boobies off their nests and consuming an estimated 200–300 eggs per week (Noble & Meier, 1989). In 1987, although nests were built and eggs laid, brown and red-footed booby nesting success was zero (Noble & Meier, 1989). Macaques contributed to the extirpation of at least five seabird species, one land bird species, and led to the depauperate state of resident land birds on Desecheo (Noble & Meier, 1989; Island Conservation, 2007). Macaques on Desecheo have also been implicated in modifying vegetation structure, contributing to the extirpation of several plant species, and preying on native reptiles including three island-endemic lizards (Evans, 1989; Breckon, 2000; Island Conservation, 2007).

In 1976, Desecheo was designated a National Wildlife Refuge and the island was transferred from the Department of Health, Education, and Welfare to the U.S. Fish and Wildlife Service (USFWS). At this time the removal of macaques was identified as an objective to restore the island's ecological integrity (Island Conservation, 2007). Between 1976 and 1988, three eradication attempts took place with a total of 155 animals removed (Herbert, 1987; Evans, 1989; Breckon, 2000; USFWS, 2007). An initial attempt was reported to have insufficient funding to proceed (USFWS, 2007). The second eradication attempt required multiple removal methods to target wary individuals. After 246 days of effort it was assumed all individuals had been removed, but less than a year later 15 individuals were confirmed on the island (Evans, 1989). The third eradication attempt ended prematurely in 1988 due to a lack of resources; it was believed at that time that two

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Fig. 1 Aerial images of Desecheo National Wildlife Refuge.

males and an unidentified juvenile were all that remained (USFWS, 2007). However, Breckon (2000) reported 11 animals in a single troop in 1996. Lack of funding, animals becoming educated to removal techniques, and unreliable detection methods contributed to the lack of eradication success. In April 2007, Island Conservation in partnership with USFWS developed a restoration plan (Island Conservation, 2007) that outlined a strategy and methods to eradicate macaques from the island. The planning effort coincided with the development of an environmental assessment covering the removal of non-hominid primates from the Commonwealth of Puerto Rico and its offshore islands (USDA, et al., 2008), including Desecheo. Here we report on the 2008–2017 eradication of macaques from Desecheo National Wildlife Refuge.

STUDY SITE

Desecheo is a small (117 ha) uninhabited hilly island (18° 23’ N, 67° 29’ W) situated roughly 21 km off the west coast of Puerto Rico. The vegetation is a mosaic of grassy patches, shrublands, woodlands, and semi-deciduous forest. The grassy patches and shrublands are on exposed ridges and screes, especially on the northern and north-eastern slopes, which face the prevailing winds. The woodlands are typically found covering coastal slopes, upper east- and south-facing slopes, along drainages, and within valley floors. The floral community of Desecheo is dry forest habitat. The island is composed primarily of Tertiary volcanic sandstones and rises to 218 m. Steep slopes fall away from five ridges interconnected by a perpendicular ridge which rises abruptly from the northeast coast (Fig. 1). There is no permanent surface water or spring on the island.

METHODS

Macaques carry B-virus (*Cercopithecine herpesvirus* 1), which can be lethal to humans (Huff & Barry, 2003), so animal handling was minimised where possible. The Desecheo macaque population originated from a population with a high occurrence of the disease (Shah & Morrison, 1969) and most likely had B-virus. When animals were handled, strict protocols were followed (Holmes, et al., 1995).

Several principles were employed to increase the likelihood of success: 1) target whole groups where possible, 2) limit opportunities to educate animals, 3) first utilise methods that would not impact the efficacy of other methods, 4) have sufficient methods to remove animals faster than the rate of reproduction, and 5) provide multiple detection methods that were independent of removal techniques, capable of detecting animals at very low densities. Variations of live-trapping and hunting were selected after a suite of possible techniques, including the use of toxicants, biological control, kill trapping, and immunocontraception, were evaluated for use on Desecheo (Island Conservation, 2007). The strategy to remove macaques was structured around three general phases and was adaptively managed from 2008 to 2017 (Fig. 2). The initial phase relied on live-trapping to provide a population reduction without educating animals to subsequent hunting methods. Select individuals captured were radio collared then released and tracked as sentinel (Judas) animals to facilitate hunting of a social species. The second phase aimed to remove remaining individuals through hunting and transitioned to a third phase where monitoring was anticipated to confirm eradication. A revised approach was required when macaques could only be detected by remote cameras. This involved specialised night hunting technology paired with the use of Judas animals and a distinct change in hunting strategy which primarily occurred outside of daylight hours.

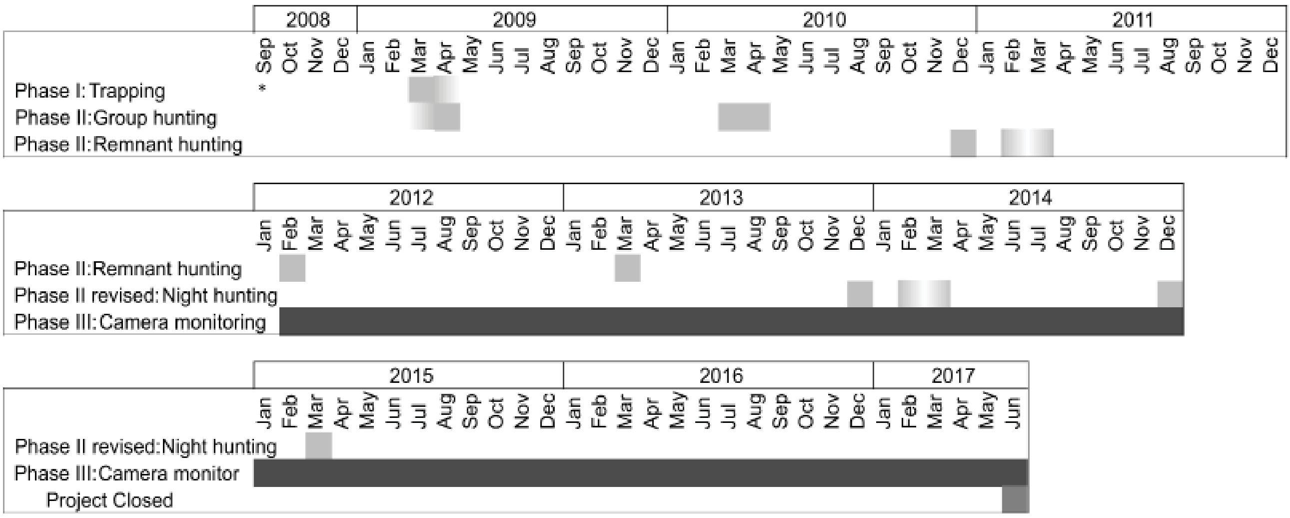

Fig. 2 Project timeline of events. (*) field work initiated by seasoning traps on site.

Phases relied on a team temporarily camped at one of two sites on the island with all equipment and supplies being delivered then removed each trip. The first campsite, serviced by helicopter, was located near the peak of the island. This site supported up to nine staff, was utilised from project initiation through the duration of group hunting (see Fig. 3) and allowed centralised access to the entire island. A second site was later established along the coastline to allow boat access and minimise logistic expenses for a reduced field team to complete the eradication.

Phase I. Trapping and Judas animal release (2008–2009)

Eighteen *#208 dual-door cage traps* (Tomahawk Live Trap, Hazelhurst, WI) were placed in groups of three across the island at sites of known macaque activity. Trap dimensions of 107 × 38 × 38 cm were considered large enough to capture multiple animals, based on mainland Puerto Rico trapping efforts (López Ortiz, 2015). Concurrently, a single, large 5 m wide group-style trap (Day, 2004) was built upon a flat, ridgetop location. This trap was constructed in an octagon shape with a wood frame, cyclone fence sides and skirt. A 60 cm overhanging eave and 60 cm vertical wall made from sheet metal faced internally to prevent animals from exiting. A remote-controlled drop-net was used in another site, comprising an 11 × 11 m reinforced net elevated above the ground by roughly 2 m around the edge with a tented peak of 5 m. Pre-baiting took place across all trap sites for two weeks with whole and sliced oranges. Oranges were chosen based on successful results experienced during previous eradication campaigns (Evans, 1989). Prior to departing the island, all cage traps and a single side of the octagon trap were wired open for animals to become familiar with their presence.

Seven months later all traps were activated and a network of 48 padded leg-hold traps was installed along areas suspected to have macaque activity. Traps were typically set in groups of two or more and each were accompanied by a magnetically triggered trap-monitor. Monitors were in place to support near-real-time monitoring of each trap's status which was communicated by radio-transmitter to a R-1000 telemetry receiver (Communications Specialist, Orange, CA); traps were monitored several times daily. A second pre-baiting effort took place during this time. To supplement oranges and provide greater variety, additional bait types including mangos, chicken eggs, and a water drip pan were utilised and replaced regularly. A remote-controlled audio lure (FoxPro Crossfire, Lewistown, PA) programmed with macaque calls also was deployed in association with baits at the drop-net location. Various leg-hold traps were set with lures including mirrors, wind chimes, streamers, feathers, or brightly coloured objects suspended above the trap site. Traps that did not receive a lure were set as a blind set with no distinguishing features separating it from the original site.

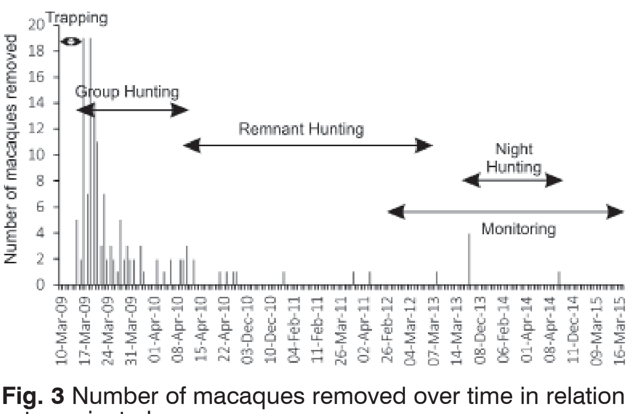

Fig. 3 Number of macaques removed over time in relation to project phase.

During this timeframe, a wild-caught adult male macaque from mainland Puerto Rico was quarantined, sterilised by vasectomy, radio-collared (Telenax, TXE-311C, Playa del Carmen, Mexico), and transported to Desecheo. This individual was released upon arrival and monitored daily as a Judas animal.

Phase II. Hunting (2009–2013)

Trapping activities from Phase I overlapped with this phase for one field trip. Hunting was intended to remove remnant individuals that were avoiding trap sets. Timing of this phase was based on the seasonally deciduous dominant tree species (*Bursera simaruba*) which leafed-out in response to rainfall. Field staff were selected from within Island Conservation and from White Buffalo Inc. (Connecticut, USA) based on their experience in precision shooting and demonstration of eradication ethic. In preparation, key vantage points were identified while conducting a census of the population before any removals took place. This assessment effort also was used to identify concealed shooting hides that offered a wide field of view for observation and clear shooting lanes. Hunting was considered capable of placing all individuals at risk of removal, particularly once population numbers were reduced with a successful trapping phase.

Troop removal (2009–2010)

Hunting predominately relied on an ambush-then-stalk strategy that collected troop characteristics (number of individuals, body size of individuals) and movements at dusk while macaques located a location to roost. In certain circumstances, where specific trees were identified as a roost site, field staff would proceed with hunting in the middle of the night while utilising spotlights and close-range shooting. In most circumstances, staff would wait until nightfall before returning to camp to develop a strategy of engagement for the following day. Before first light, field staff would be dispatched to pre-established hides or to new locations thought to offer a better vantage point of a troop's roost location. Field staff were equipped with high-capacity centrefire semi-automatic .223 Remington or 6.5 Grendel rifles with telescopic sights ranging from 4.5× to 20× magnification and reticles matched to each firearm's ballistics. Other field staff were stationed along known escape routes with high capacity 12-gauge semi-automatic shotguns.

Shooters would communicate via 2-way radio to assess the troop and attempt to identify the number of individuals, their hierarchy, and body size. Body size class was estimated based on body mass and ranked as one through five. Groups would only be engaged if it was considered a high likelihood that all individuals could be removed. Field staff that had a visual on the dominant individual would engage with the first shot, with other staff following by removing individuals that presented a lethal shot opportunity. Adult females (often dominant) were removed first, followed by adult males and juveniles. Field staff would continue to monitor the site while supporting shooters would be redistributed to areas where escapes were thought to have possibly occurred. Once macaque activity ceased in the canopy, field staff equipped with close-range firearms would enter the site to remove any remnant individuals. Removals were tallied and the animals' body size classes would be recorded. Follow-up visual confirmation of carcasses occurred whenever possible. Any known escapes were recorded, along with their size class. Confirmed removals and escapes would be cross-referenced with the troop size estimate. To improve the detection of roosting troops during this phase a commercial-grade handheld thermal camera (FLIR, P620, Wilsonville, Oregon, USA) was trialled.

Remnant removal (2011–2013)

After the initial knock-down of the macaque population, the project shifted focus to the detection and removal of lone individuals and reconstituted groups created after troops were fractured. Before dawn, field staff were stationed across the island to conduct focused observations over as much landscape as possible. Visual observation of canopy movement and audible cracks of tree limbs and masticated seeds were the primary cues of macaque presence prior to direct observation. If a detection was made, the number of animals was estimated and, if confidence existed that the group or individual could be removed, field staff would proceed by removing individuals through shooting. When assistance was required, additional field staff would be guided to the site offering the highest likelihood of removing the entire group. If escapes were thought to be probable, the team would reassess the opportunity and hold off until another situation presented greater confidence in removal.

Phase II revision. Night hunting (2013–2015)

Remote cameras (see monitoring) continued to detect macaques that were undetectable to field staff. Methods employed were re-analysed, leading to detection dogs and night hunting technology being considered. Dogs that could effectively track animals traveling on the ground and through forest canopy were considered necessary and a breed of mountain cur that is used to pursue squirrels was identified. Additionally, managers of NHPs on mainland Puerto Rico had sourced effective thermal hunting optics and began demonstrating success with night hunting.

In 2013, three macaques were selected from mainland Puerto Rico to be used as Judas animals to support night hunting. Young female macaques were chosen as they were considered more likely to readily associate with remnant animals on Desecheo. Replicating methods developed for Judas goats, each macaque was sterilised via tubal ligation, fitted with a radio-telemetry collar (ATS, M2950B, Isanti, Minnesota, USA), and received a Compudose® 200 (25.7 mg estradiol; Elanco, Indianapolis, USA) implant to induce prolonged oestrus (Zehr, et al., 1998; Campbell, et al., 2005; Campbell, et al., 2007). Radio telemetry collars had infrared (IR) reflective patches sewn in and a solar powered light-emitting diode (LED) epoxied to them to facilitate detection at night. Judas macaques were transported to Desecheo via boat and released.

Hunting methodology changed to working strictly at night, initially incorporating mainland Puerto Rico staff and their equipment to train the project team. Based on the success of these methods, thermal weapons scopes with a built in adjustable reticle (BAE Systems, Inc. ATS-6000M, Arlington, Virginia, USA), a 3rd generation night vision clip-on unit (Knight Optics Ltd., Krystal 950, Harrietsham, Kent, UK) used in combination with pre-existing telescopic firearms optics, and an IR laser illuminator (Jager-Pro LLC., JP-IR Laser, Fortson, Georgia, USA) were procured to improve detection and facilitate removals. Night operations continued with 2–3 field staff using the thermal scope to detect heat signatures of macaques in conjunction with telemetry scans for Judas animals (described in Phase III). When no Judas animals were present in a group all were targeted. When Judas animals were present night vision in conjunction with infrared illuminators were used to detect IR reflective patches sewn into collars to determine which macaque in the group was the Judas, facilitating removal of only uncollared macaques.

Phase III. Monitoring (2012–2017)

Monitoring occurred simultaneously with the removal of remnants and night hunting in Phase II. The presence of macaques was assessed through active and passive monitoring techniques; each independent of removal methods. Active monitoring occurred through visual observation of animals and the detection of fresh sign. Passive monitoring trialled acoustic recording units (Wildlife Acoustics, Maynard, MA), but relied primarily on a network of 16–26 Hyperfire PC900 no-glow cameras (Reconyx, Holmen WI). Cameras were placed in locations known to have had previous macaque activity or at sites which offered a clear field of view across a travel route. Specific attention was given to rocky bluffs, exposed patches of slope, or within tree canopies that were dominated by horizontal tree branches. Tuning the camera field-of-view used an integrated "walk-test" function which indicated where the camera would be triggered by movement. Lures made from cord passed through brightly coloured balls were installed at sites which could accommodate a swinging item without triggering the camera's motion sensor. Cameras were serviced every 3–9 months, where memory cards (32GB, 95mb/s write speed) were switched for empty ones and batteries were replaced if below 60% charge. Camera operational settings were programmed to operate from one hour prior to dawn to one hour after dusk, provide the highest sensor sensitivity, take five photos in succession, and reset immediately after a trigger event.

RESULTS

A total of 140 macaques were removed from Desecheo Island between 2009 and 2015, excluding Judas animals translocated from mainland Puerto Rico (Fig. 3). The cost of the 2007–2017 campaign was US$ 1.229m. The majority of costs (73%) were associated with implementation and monitoring from 2009 to 2015 at US$ 893k. Planning and preparation in 2007/8 utilised US$ 214k and US$ 121k was spent on confirmation over 2015–2017.

Phase I. Trapping

Baiting to encourage macaques into traps was ineffective. Additional lures such as a water drip and audio lures also proved unsuccessful as evidenced by camera traps. Non-target species, primarily black rats and hermit crabs, would consume any bait not suspended from the ground. Bait that did persist required regular replacement due to the arid climate on the island; fruits quickly desiccated and non-boiled eggs rapidly spoiled. After 26 days, unsuccessful traps that were located in remote sites and not easily accessible by field staff were closed to ration bait and improve the efficiency of trap monitoring. Traps left open were outfitted with florescent flagging as a visual lure and left open. A total of 546 trap nights accrued between cage traps, the group-style octagon trap, and the drop net with zero trap success.

Padded leg-hold traps were in place for 1,344 trap nights and resulted in the capture of 13 macaques; 10.7% of the population. Three received radio collars and were released as Judas animals; all but one was sterilised. Traps equipped with novel visual lures, particularly reflective materials, demonstrated a higher catch rate than non-reflective items. Two blind sets established as a part of a three-trap grouping, demonstrated success simultaneously. Traps placed at the base of trees where macaques would leap into the tree were particularly successful.

The adult male Judas animal transported to Desecheo from mainland Puerto Rico was found dead 16 days after its release for unknown reasons.

Phase II. Hunting

Hunting reduced the population of macaques to near undetectable levels by removing 118 individuals (84% of 140 macaques removed) over the span of two trips (46 days where hunting took place) across two years. Estimates of animals remaining at the end of each trip significantly underestimated the population. Three animals were known to be present at the end of the second trip, one of which was a sterilised Judas macaque. Follow-up hunting focused on the detection and removal of remnant macaques, which removed four individuals in five field trips (66 days where hunting took place) over three years. At this time, camera monitoring indicated six individuals remained and evidence of population recruitment, shown by one newborn juvenile.

Phase II. Revision

The introduction of night hunting strategies supported by thermal and night vision technology, field staff's intimate knowledge of the island's terrain, and leveraging Judas animal behaviour resulted in five macaque removals over five trips (50 days where hunting took place) over 2.5 years.

Phase III. Monitoring

The single most effective monitoring tool proved to be the remote camera network. Camera density ranged from one camera per 4.5–7.25 ha. Roughly 450,000 images were collected throughout the entire campaign. The volume of images varied greatly depending on the length of a monitoring period (2–9 months) with a mean of ~50K images. More than 2K macaque detections were compiled. Camera placement was impacted by vegetation growth over time leading to the majority of images being false captures. Once population numbers were reduced to five individuals the entire group could be tracked with at least one detection of each individual occurring per monitoring period. Judas animals, with unique physical and collar characteristics, could be identified within camera images and were used to indicate camera network efficacy; all Judas animals were detected per monitoring session and were easily distinguishable from wild individuals due to the presence of the collar.

Acoustic recording units were trialled but did not result in confident detections by monitoring macaque vocalisation. This tool was quickly discounted as an effective option to monitor animals at low density. The lack of vocalisation was corroborated by field staff who indicated macaques no longer vocalised with the same frequency once the population was reduced to less than ten individuals.

Additional monitoring took place though the tracking and assessment of Judas animals. Of the three animals captured on-island, one unsterilised male was found dead due to unknown causes, a second sterilised male was inadvertently shot during the hunting phase of the project, and a third sterilised female experienced a collar failure and integrated back into the population. This individual was one of the last macaques removed. Of the three additional Judas animals later captured on mainland Puerto Rico and released in Phase III, none experienced collar failure although the installed LED lights did not function in the field. After release, one was indistinguishable amongst a group of wild macaques and shot while night hunting with thermal optics. The remaining two Judas macaques formed an independent pair. One Judas animal was later removed to disrupt the social balance which resulted in a more consistent interaction between the remaining Judas animal and the two known remnant animals. Field staff removed one remnant animal before the final Judas animal was found dead in 2016. This Judas animal was associating with the last wild macaque known to remain on island.

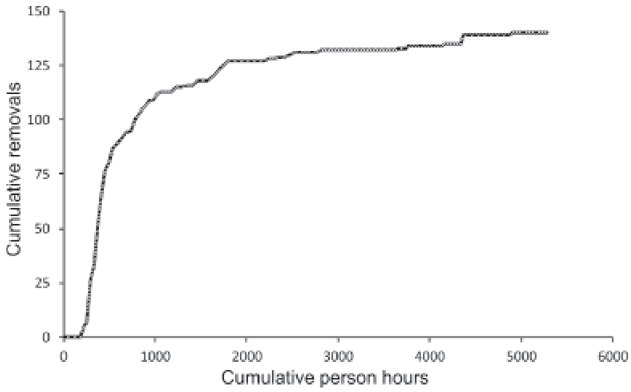

Fig. 4 Number of removals in relation to effort expended over time.

Twelve hunting trips were conducted totalling 5,280 detection hours (Fig. 4). A single wild adult matching the description of the last known wild macaque was detected on 15 occasions by the camera trap network over approximately 41 months indicating that this was the only wild individual that remained. Over the same timeframe, no juveniles were shot or detected, reflecting no reproduction. As a result, the project was closed in June 2017 with the understanding that the population was functionally extinct.

DISCUSSION AND LESSONS LEARNT

The campaign to remove macaques from Desecheo took 10 years, 17 field trips, variations of two primary methods – trapping and hunting – and a network of remote monitoring cameras to complete.

Pre-baiting attempts were unsuccessful, resulting in the ineffectiveness of baited traps. Trapping efforts may have benefited from trials and a longer pre-baiting period which also would take into account timing to allow learnt behaviours to transfer through the population. Locally available food items including nuts and berries were considered but discounted as they were found in abundance across Desecheo. Having a diet with limited exposure to novel food items on Desecheo is thought to have contributed to their disinterest in baits provided. Once baited trapping ceased, hunting and leg-hold trapping were then relied upon as the sole methods. Trapping efforts on mainland Puerto Rico that utilise a variety of fruits have resulted in up to 50% of project removals (López Ortiz, 2015) suggesting that trap success is variable across sites and should remain a management consideration.

When hunting was initiated, only troops where all individuals were thought to be at risk of removal were targeted. Although this method proved to be effective and efficient, it became apparent that escapes likely occurred unbeknownst to field staff as macaques quickly adjusted their behaviour in response to human presence and removal methods. Macaques increasingly avoided detection during the day, and if field staff were detected, would regularly select a route of escape that placed an object between them and the observer, limiting shot opportunities, as they fled to an adjacent watershed. This behaviour eventually nullified daylight hunting and required revised methods and tactics to improve the probability that an individual would be detected.

Advanced night hunting equipment facilitated both detections and removals. In many instances macaques were detectable only with a thermal scope, even when field staff knew the location of the animal. As a result, an integrated shooting reticle with the ability to remove and return the scope to the firearm without having any shift in the scope's point of aim was considered critical. Furthermore, having the ability to de-couple the night vision from traditional hunting scopes proved valuable as the firearm's point of aim did not need to be recalibrated for daylight hunting; only the thermal scope could be used in daylight without damaging the equipment or losing the capability to continue hunting into dawn with the firearm paired with night vision. A less sophisticated general-use FLIR thermal camera was trialled early in the project although low image resolution limited the unit's detection range. At ranges beyond ~150 m, individual pixels were estimated to be larger than a macaque's heat signature. The camera's limited range resulted in zero detections and thus general-use thermal tools were abandoned.

If the project had been initiated with advanced night hunting thermal equipment and Judas animals it is estimated that its duration and cost would have been significantly reduced. Hunting activities could have taken place regardless of seasonal variation in vegetation, detections would have been more frequent, and entire groups could have been removed with greater confidence, precision, and frequency. Furthermore, the incorporation of suppressed firearms with subsonic ammunition would have offered additional advantages. Suppressed firearms would likely have reduced the flight response of any associated macaques due to abated firearm report, projectile "crack," and identification of shot origin.

Camera traps provided high detection likelihood as compared to other passive detection methods, particularly once the density of animals was reduced to near-zero. Camera placement, and the decision to increase the size of the network, was guided by weeks of observation before and after removals took place and is believed to have significantly improved detection probability. Staff were familiar with the use of the same cameras with feral cats (*Felis catus*), however, a greater awareness of the camera's field-of-view and trigger window was necessary when setting cameras to monitor a three-dimensional environment. The presence of an accurate walk-test function offered confidence that cameras were set to detect animals at varying elevations and distances. In addition, a robust camera design offered confidence that cameras would have a low failure rate regardless of adverse field conditions including hurricanes, intense heat, and sustained humidity. Failures generally included screen malfunctions, walk-test function ceasing to work, and component corrosion due to termites burrowing into the camera case.

Ongoing commitment from the partners throughout a dynamic project enabled the eradication to succeed. There was a significant investment up front to start the project and the initial projected methodologies and associated tools did not result in eradication. As a result, the project lasted longer than expected and overall costs were higher than anticipated. These costs may have been reduced if the funding was available in larger amounts rather than in annual allocations, allowing higher intensity effort over a shorter period of time. However, it is also believed that the long periods between hunting trips was beneficial because macaques became less agitated, resumed routine behaviours, and were more likely to be detected.

Macaques becoming educated to removal techniques, unreliable detection methods, and a lack of funding were linked to previous failures on Desecheo. To reduce the probability that similar issues would impact the success of this attempt, the partnership routinely and transparently reassessed all aspects of the project including the funding required to proceed, equipment and field trips considered necessary to achieve eradication, and how to interpret results. These factors guided an adaptive management strategy that supported principles outlined within original project planning. This shared effort resulted in a robust relationship that was capable of addressing a dynamic project and uncertainty in a solution-oriented, step-by-step manner. As a result, a project with no precedent of success – incentivised by a high conservation reward – was completed in a conscious and calculated fashion.

CONCLUSION

Desecheo Island is the location of the first successful removal of introduced non-hominid primates from an island that we are aware of. The project was contingent on the strength of the partnership, specialised equipment, and commitment of an experienced field team with a strong eradication ethic that followed a plan based on eradication theory. These factors were all critical to the project's success after a protracted time-period. The challenges of this eradication required several revisions to the original methodologies and strategies, as well as continued funding beyond the original budget projections.

Desecheo Island, and the unique species that are found there are now safe from invasive mammals after nearly a century. This restoration action should enable the island's return as the most important seabird colony within the region.

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REFERENCES

Bowdish, H.S. (1900). 'A day on De Cicheo Island'. *The Oölogist* 17: 117–120.

Breckon, G.J. (2000). 'Revision of the flora of Desecheo Island, Puerto Rico'. *Caribbean Journal of Science* 36: 177–209.

Campbell, K.J., Baxter, G.S., Murray, P.J., Coblentz, B.E., Donlan, C.J. and Carrion, V.G. (2005). 'Increasing the efficacy of Judas goats by sterilisation and pregnancy termination'. *Wildlife Research* 32: 737–743.

Campbell, K.J., Baxter, G.S., Murray, P.J., Coblentz, B.E. and Donlan, C.J. (2007). 'Development of a prolonged estrus effect for use in Judas goats'. *Applied Animal Behaviour Science* 102: 12–23.

Day, T.D. (2004). *Feasibility of Pest Proof Fencing in Mauritius*. Cambridge, New Zealand: Xcluder Pest Proof Fencing Company.

Evans, M.A. (1989). 'Ecology and removal of introduced rhesus monkeys: Desecheo Island National Wildlife Refuge, Puerto Rico'. *Puerto Rico Health Science Journal* 8: 139–156.

Feild, J.G., Henke, S.E. and McCoy, J.G. (1997). 'Depredation on Artificial Ground Nests by Japanese Macaques: The Unspoken Exotic in Texas'. In: C.D. Lee and S.E. Hygnstrom (eds.) *Thirteenth Great Plains Wildlife Damage Control Workshop Proceedings,* pp. 151–155. Kansas State University Agricultural Experiment Station and Cooperative Extension Service.

Fooden, J. (2000). *Systematic Review of the Rhesus Macaque,* Macaca mulatta *(Zimmermann, 1780),* Chicago: Field Museum of Natural History.

Herbert, H.J. (1987). *Unpublished Final Report on Desecheo Island Monkey Removal Program, to Sean Furniss, Caribbean Islands National Wildlife Refuge*. Boqueron, Puerto Rico.

Holmes, G.P., Chapman, L.E., Stewart, J.A., Straus, S.E., Hilliard, J.K. and Davenport, D.S. (1995). 'Guidelines for the prevention and treatment of B-virus infections in exposed persons. The B virus working group'. *Clinical Infectious Diseases* 20: 421-439.

Huff, J.L. and Barry, P.A. (2003). 'B-virus (*Cercopithecine herpesvirus* 1) infection in humans and macaques: Potential for zoonotic disease'. *Emerging Infectious Diseases* 9: 246–250.

Island Conservation (2007). *Restoring Desecheo Island National Wildlife Refuge: Macaque Eradication Plan, December 2007*. Santa Cruz: Island Conservation.

Jones, H.P., Campbell, K.J., Burke, A., Baxter, G.S., Hanson, C.C. and Mittermeier, R.A. (2018). 'Introduced non-hominid primates impact biodiversity and livelihoods: management priorities'. *Biological Invasions*: <https://doi.org/10.1007/s10530-018-1704-5>.

Kemp, N.J. and Burnett, J.B. (2003). *Final Report: A Biodiversity Risk Assessment and Recommendations for Risk Management of Long-Tailed Macaques (Macaca fascicularis) in New Guinea*. Washington, DC: Indo-Pacific Conservation Alliance.

Kier, G., Kreft, H., Lee, T.M., Jetz, W., Ibisch, P.L., Nowicki, C., Mutke, J. and Barthlott, W. (2009). 'A global assessment of endemism and species richness across island and mainland regions'. *Proceedings of the National Academy of Sciences* 106: 9322–9327.

López Ortiz, R. (2015). *T-10 Project for the Control of Wild Primates in Southwestern Puerto Rico. Puerto Rico Department of Natural and Environmental Resources. Terrestrial Resources Division.* <http://Drna.Pr.Gov/Programas-Y-Proyectos/Proyecto-Para-El-Control-De-Primates-En-El-Suroeste-De-Puerto-Rico/>. Accessed 26 June, 2017.

Meier, A.J., Noble, R.E. and Raffaele, H.A. (1989). 'The birds of Desecheo Island, Puerto Rico, including a new record for Puerto Rican territory'. *Caribbean Journal of Science* 25: 24–29.

Noble, R.E. and Meier, A.J. (1989). *Status of Boobies,* Sula sula *and* Sula leucogaster*, on Desecheo Island, Puerto Rico*, Unpublished report to the U.S. Fish and Wildlife Service, Contract No. 14-16-009-1561.

Shah, K.V. and Morrison, J.A. (1969). 'Comparison of three rhesus groups for antibody patterns to some viruses: Absence of active simian virus 40 transmission in the free-ranging rhesus of Cayo Santiago'. *American Journal of Epidemiology* 89: 308–315.

Strier, K.B. (2016). *Primate behavioral ecology*. London, England: Routledge.

Struthers, P.H. (1927). 'Notes on the bird-life of Mona and Desecheo Islands'. *Auk* 44: 539–544.

Tershy, B.R., Shen, K.-W., Newton, K.M., Holmes, N.D. and Croll, D.A. (2015). 'The importance of islands for the protection of biological and linguistic diversity'. *BioScience* 65: 592–597.

USDA, APHIS and WS (2008). *Environmental Assessment (Pre-Decisional). Managing Damage and Threats Associated with Invasive Patas and Rhesus Monkeys in the Commonwealth of Puerto Rico*. Washington, DC: US Department of Agriculture, Animal & Plant Health Inspection Service, Wildlife Services.

USFWS (2007). *Summary of Desecheo Trip Reports for Macaque Eradication.* Compiled by Joseph Schwagerl. Unpublished.

Wetmore, A. (1918). 'The birds of Desecheo Island, Porto Rico'. *Auk* 35: 333–340.

Will, D.J., Swinnerton, K., Silander, S., Keitt, B., Griffiths, R., Howald, G.R., Figuerola-Hernandez, C.E. and Herrera-Giraldo, J.L. (2019). 'Applying lessons learnt from tropical rodent eradications: a second attempt to remove invasive rats from Desecheo National Wildlife Refuge, Puerto Rico'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 154–161. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Zehr, J.L., Maestripieri, D. and Wallen, K. (1998). 'Estradiol increases female sexual initiation independent of male responsiveness in rhesus monkeys'. *Hormones and Behavior* 33: 95–103.

Eradication of red deer from Secretary Island, New Zealand: changing tactics to achieve success

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Abstract Red deer (*Cervus elaphus*) established on 8,140 ha Secretary Island after swimming from the mainland in the early 1960s. Attempts to remove them began in the 1970s and after several starts and stops they were eradicated in late 2014. Since late 2006, 688 deer have been removed. Ground hunters killed 365 deer in 1,827 hunter-days, 320 deer were shot from helicopters in 211 flying-hours, two deer were trapped and one was known to have been killed by a fisherman. The campaign since 2006 was planned in three phases – an initial population reduction, a mop-up phase and a surveillance and rapid response to any new immigration phase. An initial reduction of 80% of the population, between 530 and 550 in 2006, was planned and achieved in the first two years. The removal of surviving deer was planned to take a further four years but despite 114 being shot and probably less than 14 deer remaining in 2013 eradication was not achieved using the methods that succeeded in the initial phase. The change in tactics in 2014 that allowed for eradication was to (a) ground survey the island and use camera traps to locate areas with deer, (b) identify individual deer from faecal DNA to estimate numbers, know when they were shot or still alive, and to estimate potential new immigration from the mainland – which was low, and (c) move from individual hunters seeking any deer within a widespread population, when about 10% of hunter-deer encounters led to a kill, to re-train hunters as teams using GPS/radio systems and integrate them with aerial hunting to seek individual deer at known locations, when 100% of encounters led to a kill. The change of tactics that led to eradication success required about half the costs, i.e. $25,000 to $10,500 per deer direct operational costs, expected if no change had been made.

Keywords: aerial hunting, catch-per-unit-effort, density, faecal DNA profiles, ground hunting, operational costs, World Heritage Area

INTRODUCTION

There are few published examples of successful deer eradication campaigns in the world. This is mostly because deer are generally valued as resources rather than as pests but, in New Zealand, red deer are an introduced species so there is interest in completely removing deer from some places in order to protect the native biota (Parkes & Murphy, 2003). Here we document a prolonged but ultimately successful campaign to remove deer from a large island in south-western New Zealand.

Secretary Island covers 8,140 ha and rises to 1,196 m a.s.l. at 45°14' S 166°55' E in Fiordland National Park, part of Te Wahi Pounamu South-west New Zealand World Heritage Area (Fig. 1). Red deer (*Cervus elaphus*) swam to Secretary Island from the mainland in the early 1960s (Mark & Baylis, 1975; Crouchley, et al., 2007) across a sea gap of at least 630 m. A population established and their impact on the pristine native forests was severe and rapid (Mark & Baylis, 1975; Mark, et al., 1991) so in the 1970s, New Zealand Forest Service attempted, unsuccessfully, to remove the deer (Tustin, 1977). However, in the early 2000s, the New Zealand Department of Conservation (DOC) initiated a new campaign (Brown, 2005; Crouchley, et al., 2007) that began in earnest in late 2006. This second eradication attempt was itself reassessed by DOC once the population had been reduced to very low numbers (estimated at 14 individuals) in 2012/13, resulting in changes in strategy and operational tactics that eventually led to successful eradication of the deer. In this paper, we briefly reiterate the results presented in early reports and in the second Island Invasives conference for the first eradication attempt, and update the results from the initial reduction phase (Crouchley, et al., 2011; Edge, et al., 2011). We then focus on the new data to report on the change in strategy and tactics to remove the last few deer from the island and compare the predictions of a catch-per-unit-effort (CPUE) model produced in 2012 (Nugent & Arienti-Latham, 2012) with the actual outcomes of the deer control during the final phase of the project.

MAIN FINDINGS

First eradication attempt: 1970–1989

Ground and aerial hunting began in the early 1970s and although 250 deer were reported as killed by the New Zealand Forest Service between 1970 and 1985 (Brown, 2005) the population, in the presence of abundant food (Mark & Baylis, 1975), continued to increase. Tustin (1977) guessed about 200 deer were present in 1975. A poisoning technique (1080 gel smeared on the leaves of deer-preferred plants; see Parkes, 1983) was trialled from 1975 to1987 (when 10% of the island was poisoned) but informal track and pellet counts suggested efficacy was moderate at best (Brown, 2005). The abundance of preferred food species and a perception that the difficult terrain on Secretary Island restricted ground access (later disproved when hunters covered the whole island to survey for surviving deer) were likely reasons this trial did not lead to eradication of the deer. In contrast, in an area on Stewart Island, where white-tailed deer (*Odocoileus virgineanus*) had removed most palatable food plants and accessibility to people was not difficult, the 1080-gel technique removed close to 100% of the population of deer in the treated area (Nugent, 1990). The best control methods depend on context, showing that successful precedent does not supply a recipe for new projects.

By the early 1980s it was concluded that neither hunting nor the 1080-gel method could remove all deer, so the policy shifted in 1985 to one of sustained control to low residual densities (Sanson & von Tunzelman, 1985). By 1989, official deer control on the island was halted because of budget constraints and the expectation that reinvasion would always compromise the project (W. Chisholm, 1989, unpubl. DOC Invercargill file ANI 4/6). Deer were still shot on Secretary Island by commercial venison recovery helicopter operators. However, the goals of restoring the island's ecosystems by controlling deer and stoats were

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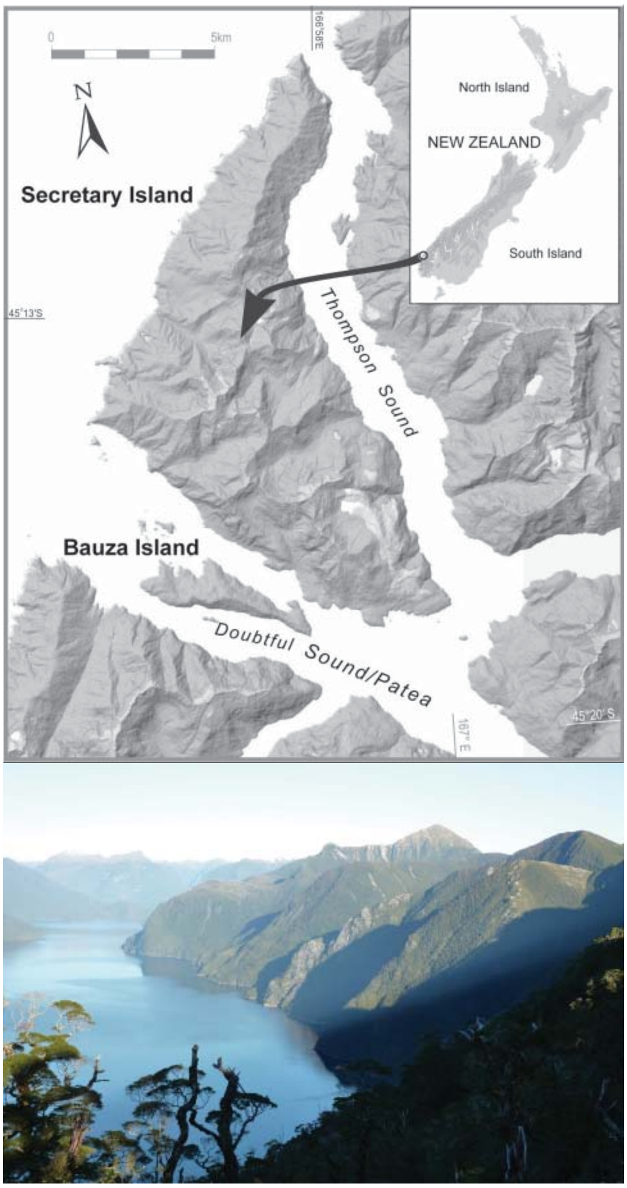

Fig. 1 Secretary Island, Fiordland National Park, New Zealand (photo L. Wilson).

not forgotten (Munn, 2001) and in the early 2000s a new eradication project was proposed (Brown, 2005) with a formal operational plan developed in 2007 (Crouchley, et al., 2007).

Second eradication attempt: November 2006–April 2013

A new decision to attempt eradication of red deer (and also stoats (*Mustela erminea*)) from Secretary Island and nearby Resolution Island (21,000 ha) was proposed in 2004 and a budget of NZ$7.1 million was allocated (Edge, et al., 2011). This second attempt adopted a more strategic approach, aiming to reduce the population by 80% within two years, then remove survivors within four years, and subsequently detect and remove any new immigrants in perpetuity (Crouchley, et al., 2011). It was expected that the initial knockdown would rely on two main methods (ground hunting with indicator dogs and helicopter shooting) but that a variety of 'niche' control methods (17 capture pens, fences, the use of telemetered deer) would probably be required during the 'mop-up' phase (Crouchley, et al., 2011). The ground hunting involved hunters (and their dogs) operating individually from nine huts across the island, so each hunter covered different areas in each hunting period (usually about nine days) with hunters often swapping areas between hunting periods (see Crouchley, et al. (2011) for a detailed description of the hunting and other methods). Aerial and ground hunting began in November 2006.

The 'rapid knockdown' aim was effectively met as 84% of the total deer killed were shot within three years (by the end of 2009). We estimated that hunters operating individually killed only about 10% of deer they 'encountered', i.e. seen, heard or known to be in the area being hunted from fresh sign. There was little motivation to persist with hunting a particular deer that escaped when there were plenty of other deer in the area being hunted. However, the aim to eradicate the population by the end of 2012 was not met as deer were still present. In retrospect, 98% of the final tally had been killed by then, but not the 100% required for eradication.

Final push: January 2014–August 2014

Failure to eradicate by 2012 (Fig. 2) led to a hiatus in activity while the strategy and tactics being used for the 'mop-up' phase were reconsidered. The surviving deer were extremely wary and could detect and escape hunters (with dogs) operating as individuals and were avoiding the open grasslands where they would be most vulnerable to aerial shooting. The Department of Conservation had no novel control tools to add to the mix it had already used so decided that they had to apply ground and aerial hunting in a different way to counter these learnt avoidance behaviours of the deer. A decision was made to shift from individual hunting to team hunting informed by all available information. To some extent this was informed by the experience of the new project manager (the senior author) who with a private company (Prohunt Ltd, now Native Range Ltd) had recently achieved eradication of feral pigs (*Sus scrofa*) from Santa Cruz Island (Parkes, et al., 2010). Technological advances available in the final phases of the Secretary Island project included the use of hand-held GPS and radios that allowed immediate contact and location details to be shared between hunters, high definition remote trail cameras, and the ability to identify individual deer from DNA in faecal pellets.

The first step, in February 2013, under the revised strategy, was to use hunters with indicator dogs to search the whole island from ridge tops to the sea along transects about 200 m apart for sign of deer. Analysis of the DNA in the mucus layer of fresh (i.e. moist with unbroken exterior estimated to be only a few day's old) faecal pellets (see Ramón-Laca, et al., 2014 for details of the methods; such

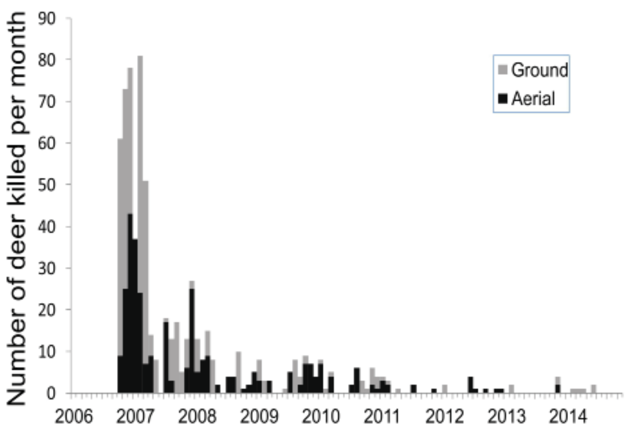

Fig. 2 Monthly kills of red deer on Secretary Island between the start of the second eradication campaign in late 2006 and the last deer killed in August 2014.

analyses currently cost about NZ$90 per sample depending on sample size) found during this survey allowed individual deer to be identified and the area in which they lived to be located. The whole-island sign survey suggested that possibly 14 deer remained at the end of April 2013 (Macdonald, 2013). The second step, in late 2013, was to select and train the hunters in the skills (and attitudes) required for a team hunting and targeting individual deer. The logic of this change in hunting method depended on (a) identifying from ground surveys for sign roughly where a targeted survivor was living, (b) using a helicopter to place a team of hunters at key exit points around that location, (c) then deploying the best hunter-and-dog teams in the suspected range of the deer to attempt to find and kill it, (d) and, where that failed and the deer also avoided the perimeter hunters, to then use the dog to track the deer and the helicopter to either relocate the perimeter ambushers or to shoot the deer if it became visible.

As hunting under a new strategy proceeded and the DNA taken from shot animals was compared with DNA found in an ongoing collection of faecal pellets it was estimated that only eight deer remained by the end of 2013. Deployment of 13 trail cameras at key sites around the island combined with ongoing DNA sampling did not identify any new 'unknown' deer at this stage of the mop-up. All deer shot after the island-wide survey in 2013/14 were (apart from two fawns shot with their mothers) identified with the DNA faecal pellet database, and all but one had an image captured by a trail camera.

Nine deer were shot in in 2014 under the new strategy. Seven were adults (3F, 4M) and two were fawns. Three deer were shot by ground hunters, two from helicopters, and four from helicopters after the deer had been flushed out of the forest by ground hunters and their dogs. The last known animals were shot during August 2014 – a pregnant female which was flushed out of the forest by hunters and their dogs and shot from a helicopter, and an adult male shot by the ground hunters.

Initial population size

The careful collection of hunting statistics – numbers of deer killed, their age and sex and hunting effort – allows us to construct models of the population size and structure at any point during the project since 2006. The ages of 78 females shot on the island in 2006/07 and classed as adults by the hunters were determined from tooth cementum layers (Fraser & Sweetapple, 1993). All other animals were aged into three classes (young of the year, yearling, and older) by the hunters in the field. The population size in 2006 can be estimated using a form of the 'minimum number known to be alive' (MNA) analysis of McCullough, et al. (1990). Simply, the age of each animal shot was used to determine if it was alive in 2006 and the pre-fawning MNA population size in December 2006 (fawns are assumed all born at this time of year) is all animals shot after December 2006 that had been born before December 2006, plus all deer killed in 2007 other than fawns born in December 2006, plus all deer shot in 2008 other than fawns born in 2008 and sub-adults born in 2007, and so on. After 2009 an unknown number of deer in the oldest age class may have been born after 2006. To subtract these from our estimate of the initial population we used the age-class distribution of the 78 deer accurately aged and assumed the proportions remained the same across the post-2009 deer that were killed. Given 84% of the estimate of initial population size accumulates in the first three years, the potential errors in using this age distribution for older deer born after 2006 are minor. We assumed all deer were accurately aged, particularly when allocated an age class in the field, there was no immigration from the mainland and hunting by the official hunters was the major cause of mortality.

Between November 2006 and August 2014, a total of 688 deer were killed, of which at least between 530 and 550 would have been alive at the start of the eradication project in late 2006: an MNA 2006 density of 6.7 deer/km^2. The actual number was probably slightly higher as our estimate is based on known deaths and does not include animals that may have been wounded and died, died naturally, or were shot by other hunters and not reported.

Costs

Assuming direct operational costs of NZ$950 per flying-hour and $330 per hunter-day (the hunters were contracted for set periods but paid whether they actually hunted on a particular day or not) and using a population reconstruction model with a starting population size of 530 animals and an annual recruitment rate of 24%, the cost per deer shot increased rapidly as deer density declined for both aerial and ground hunting methods (Fig. 3). The cumulative 2006–2014 direct operational costs totalled $732,830 plus unknown management overheads that are likely to be roughly similar across years as they are less related to hunting effort.

We fitted a negative power function to the cost per deer versus density data from 2006 to 2012 (Fig. 3). Extrapolation from that curve suggested that expenditure of > $200,000 in direct costs would be required to remove the estimated residual population of eight deer within one year if there were no change in tactics. However, with the change in tactics in 2014, the actual direct costs were only about $84,000, indicating that the change in tactics was not only successful but much more cost-efficient. This of course ignores the significant factor of good luck (or bad luck from the deer's point of view) at the end of such

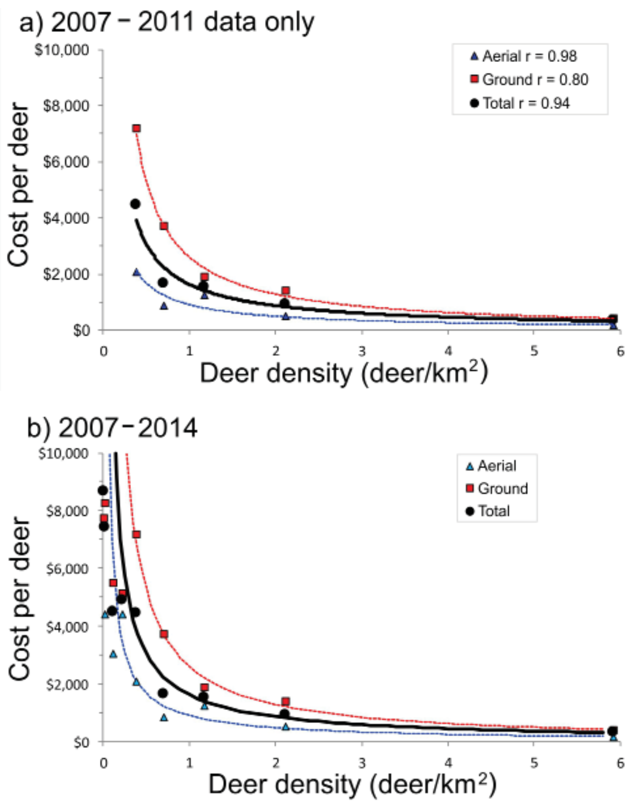

Fig. 3 Direct costs (hunter-days and flying hours) per deer killed with decreasing density, for aerial hunting, ground hunting and overall, for (a) the data from the first five years, and (b) for the whole campaign. The power curves in (b) are extrapolations from the 2007–2011 data and show that costs in the final stages (after adoption of new tactics) were lower than predicted from the initial data.

eradication projects, e.g. see the last pig from Santa Cruz Island which was shot incidentally to another task (Parkes, et al., 2010).

DISCUSSION

Eradication projects that rely on a succession of control events to eventually remove the population have one advantage over single-event projects, such as aerial baiting for rodents, in that information on progress and problems accrues as data are collected from each event. This allows managers to change tactics as the population is reduced and especially when surviving animals are less accessible or have learnt to avoid the control methods deployed at the start of the project. Previous successful and efficient eradication projects of this type have developed some practices (e.g. Ramsey, et al., 2009; Parkes, et al., 2010) that were, in part, used in the Secretary Island project.

The first success factor in such projects is that they reduce the population to very low densities as quickly as possible using control techniques that maximize the probability that every animal is killed at first encounter and thus minimize the possibility that surviving animals learn to avoid all control methods. It might be argued that live trapping in capture pens or 1080-gel on natural bait poisoning does not make surviving deer more wary, at least to subsequent hunting if not to the danger of traps, and should be used first. However, trapping is capital and labour intensive, unlikely to achieve rapid reduction in deer populations, while the earlier attempts at natural bait poisoning in Secretary Island were thought to be unsuccessful in achieving a large reduction. This left aerial and ground-based hunting as the only practical tools to achieve the initial population reduction, but which inevitably do not kill all deer at first encounter and so leave wary survivors. It is unknown whether the same successful initial reduction could have been achieved, and without creating wary survivors, by starting with the approach (team hunting with the additional improved GPS/radio and DNA technologies, and closer integration between ground and aerial hunting) deployed in the mop-up phase after 2013.

Many of the estimates of the number of deer left at various points across the campaign were essentially informed guesses. However, three tools were used to improve confidence in estimates of the number and identity of deer surviving on the island – a model based on catch per unit effort data, camera traps and the use of DNA from faecal pellets and aged and sexed shot individuals to determine presence of un-shot deer (pellets present for an individual not yet shot) and familial relationships (younger animal shot but not yet its parents) and potentially whether the DNA is from a resident survivor or an immigrant from the South Island.

The DNA from the deer shot during the campaign suggested they were all closely related (Crouchley, et al., 2011). This precluded trying to use the DNA in young animals (which were easier to shoot than adults) as a marker to see if their parents are eventually shot (e.g. see Nugent, et al., 2005). However, this is good news as the island deer had few of the rarer alleles present on the mainland. This suggests that the initial immigration in the 1960s had not been repeated, probably because deer populations throughout Fiordland were greatly reduced by commercial aerial hunting after that time (Nugent, et al., 1987). Therefore, the extirpation of the resident population on Secretary Island might indeed be eradication sensu stricto – still, a precautionary approach of surveillance and rapid response to any new incursions is intended.

Some general observations to ensure surviving deer did not escape are:

(a) to deploy hunters at optimal times/weather rather than on a set schedule,

(b) to know the general areas on the island where the surviving deer are living by extensive ground searches and use of camera traps,

(c) to know which individual deer have escaped the hunters by comparing DNA in faeces with DNA in animals shot and,

(d) to change the mindset of the hunters from ‘control’ to ‘eradication’, i.e. from acting as individuals, however skilled, each hunting any deer in their hunting block, to team hunters with appropriate technologies to act as a team and target individual deer.

The success on Secretary Island, and other smaller islands in Fiordland National Park, provides some templates for the proposed projects against red deer on similar islands. Eradication of red deer has been attempted on Resolution Island (21,000 ha), which is also in Fiordland National Park (Edge, et al., 2011). This project has not succeeded and is currently being reviewed (N. Macdonald, pers. comm.). The Government of Argentina is also considering whether to attempt to eradicate red deer and feral goats (*Capra hircus*) from Isla los Estados (Staten) Island (53,400 ha) in Tierra del Fuego – another remote, mountainous island dominated by southern beech forests (A. Schiavini, pers. comm.). New technologies to locate cryptic deer are also becoming available with improvements in infrared systems (FLIR) currently being deployed against black-tailed deer (*Odocoileus hemionus*) that have survived an eradication attempt on 1,637 ha Ramsay Island in British Columbia (N. Macdonald, unpubl. data).

The general strategy used on Secretary Island, of an initial rapid reduction in the deer population followed by removal of survivors, succeeded in its aim of eradication. However, in retrospect there is always going to be a difficult decision for managers when deciding when to deploy different control tactics across such a campaign. An ideal approach would be to begin with control methods that do not teach surviving animals to avoid later control, and then to apply control methods in a way that minimises the chance of animals escaping each encounter. On Secretary Island, and potentially for other deer eradication projects, we suggest that the team hunting system and coordination between ground and aerial hunting may have been better applied from the start of the 2006 hunting campaign rather than towards the end of the eradication.

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REFERENCES

Brown, D. (2005). *Secretary Island Deer Eradication Scoping Document*. Invercargill: Department of Conservation.

Crouchley, D., Brown, D., Edge, K-A. and McMurtrie, P. (2007). *Secretary Island Operational Plan: Deer Eradication*. Te Anau: Department of Conservation.

Crouchley, D., Nugent, G. and Edge, K-A. (2011). 'Removal of red deer (*Cervus elaphus*) from Anchor and Secretary Islands, Fiordland, New Zealand'. In: C.R. Veitch; M.N. Clout and D.R. Towns (eds.). *Island invasives: eradication and management*, pp. 422–425. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Edge, K-A., Crouchley, D., McMurtrie, P., Willans, M.J. and Byrom, A. (2011). 'Eradicating stoats (*Mustela erminea*) and red deer (*Cervus elaphus*) off islands in Fiordland'. In: C.R. Veitch; M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 166–171. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Fraser, K.W. and Sweetapple, P.J. (1993). *Assessing the Age and Condition of Deer from Jawbones*. Christchurch, New Zealand: Landcare Research.

McCullough, D.R., Pine, D.S., Whitmore, D.L., Mansfield, T.M. and Decker, R.H. (1990). 'Sex linked harvest strategy for big game management with a test case on black-tailed deer'. *Wildlife Monographs* 11: 1–41.

Macdonald, N. (2013). *Removal of Red Deer from Secretary Island, Fiordland National Park 2013–2014*. Invercargill: Department of Conservation. DOCDM-1294828.

Mark, A.F. and Baylis, G.T.S. (1975). 'Impact of deer on Secretary Island, Fiordland, New Zealand'. *Proceedings of the New Zealand Ecological Society* 22: 19–24.

Mark, A.F., Baylis, G.T.S. and Dickinson, K.J.M. (1991). 'Monitoring the impact of deer on vegetation condition of Secretary Island, Fiordland National Park, New Zealand: A clear case for deer control and ecological restoration'. *Journal of the Royal Society of New Zealand* 21: 43–54.

Munn, A. (2001). *Secretary Island. A Consideration of the Ecological Values of the Island and the Potential for Restoration*. Invercargill: Department of Conservation.

Nugent, G.; Parkes, J.P. and Tustin, K.G. (1987). 'Changes in the density and distribution of red deer and wapiti in northern Fiordland'. *New Zealand Journal of Ecology* 10: 11–21.

Nugent, G. (1990). *A White-tailed Deer Poisoning Trial on Stewart Island*. Christchurch, New Zealand: Forest Research Institute Contract Report FEW 90/4.

Nugent, G.; Whitford, J. and McEwan, M. (2005). 'DNA-based Mark-recapture of Wild Deer: Using Fawns to Capture their Mothers'. In: J. Parkes, M. Statham and G. Edwards (eds.) *Proceedings of the 13th Australasian Vertebrate Pest Conference*, pp. 91–95. Lincoln, New Zealand: Landcare Research.

Nugent, G. and Arienti-Latham, C. (2012). *Simple Models of the Secretary Island Deer Eradication Programme*. Lincoln, New Zealand: Landcare Research Contract Report LC1057.

Parkes, J.P. (1983). 'Control of feral goats by poisoning with compound 1080 on natural vegetation baits and by shooting'. *New Zealand Journal of Forestry Science* 13: 266–274.

Parkes, J.P.; Ramsey, D.S.L.; Macdonald, N.; Walker, K.; McKnight, S.; Cohen, B.S. and Morrison, S.A. (2010). 'Rapid eradication of feral pigs (*Sus scrofa*) from Santa Cruz Island, California'. *Biological Conservation* 143: 634–641.

Parkes, J. and Murphy, E. (2003). 'Management of introduced mammals in New Zealand'. *New Zealand Journal of Zoology* 30: 335–359.

Ramón-Laca, A.; Gleeson, D.; Yockney, I.; Perry, M.; Nugent, G. and Forsyth, D.M. (2014). 'Reliable discrimination of 10 ungulate species using high resolution melting analysis of faecal DNA'. *PLOS ONE* 9(3): e92043.

Ramsey, D.S.L.; Parkes, J.P. and Morrison, S.A. (2009). 'Quantifying eradication success: The removal of feral pigs from Santa Cruz Island, California'. *Conservation Biology* 23: 449–459.

Sanson, L.V. and von Tunzelman, J.R. (1985). *Wild Animal Control Plan for Secretary Island, Fiordland National Park*. Invercargill: New Zealand Forest Service.

Tustin, K.G. (1977). 'Secretary Island – a problem in deer control'. *What's New in Forest Research* No. 50. Rotorua, New Zealand: Forest Research Institute.

Large scale eradication of non-native invasive American mink (*Neovison vison*) from the Outer Hebrides of Scotland

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Abstract The Hebridean Mink Project was tasked with eradicating American mink (*Neovison vison*) from the Outer Hebrides, an extensive, complex island archipelago, amounting to 3,050 km^2. Hundreds of islands contribute to a coastline of approximately 2,500 km, 15% of Scotland's total. The geographical complexity continues inland with over 7,500 freshwater lochs, ~24% of Scotland's total, which enables invasive American mink, in suitable habitats, to reach densities seldom encountered elsewhere. With major funding from the EU LIFE programme, removal from the Uists began in 2001. By 2006 eradication was declared there, as no captures had occurred for 16 months. In 2007 the project extended into Harris and Lewis, adopting a systematic network of live capture traps (7,039 spaced at 450–500 m intervals utilising prominent features of the riparian network and coastline). The traps were checked in rotation until at least a 95% reduction in population had been achieved. An incremental, strategic change from systematic trapping to detection; by means of footprint monitoring, cameras and dog searching, followed by responsive trapping then occurred from 2011 onwards. By 2013 a lethal monitoring system utilising 'kill traps' was employed alongside remote alert systems which allowed the project to remove the remaining population of mink from Lewis and Harris, with a reduced staff resource, and increase the trap night total to in excess of 500,000. To date, 2,198 mink have been caught, but only two non-breeding females and associated males have been caught in Lewis and Harris in the last 18 months (no juveniles captured). The challenges of geographical scale, terrain, climatic conditions and a continuously reducing staff complement have required an adaptive management approach to achieve the project goal of a mink-free Outer Hebrides that benefits ground nesting birds and migratory fisheries. This is viewed as a highly effective eradication project, and lessons learnt can be put into place for other ambitious control programmes.

Keywords: adaptive management, anal gland lure, dog searching, monitoring techniques, remote alert systems, trapping

INTRODUCTION

The Outer Hebrides of Scotland support some of the most important breeding populations of waders in Europe. Species include redshank (*Tringa totanus*), snipe (*Gallinago gallinago*), lapwing (*Vanellus vanellus*) and oystercatcher (*Haematopus ostralegus*); with dunlin (*Calidris alpina*) and ringed plover (*Charadrius hiaticula*) nesting at the highest densities recorded anywhere in the world (Stroud, et al., 2001). In recognition of this, many of the nesting areas have been notified as Sites of Special Scientific Interest (SSSI) under the Nature Conservation (Scotland) Act 2004, and classified as Special Protection Areas (SPA) under the EC Birds Directive, covering an area of about 37,596 ha and 87,158 ha respectively.

At the international level, there are many more species of birds that are represented by important populations on these sites. Species include red-throated diver (*Gavia stellata*), black-throated diver (*Gavia arctica*), great northern diver (*Gavia immer*), hen harrier (*Circus cyaneus*), merlin (*Falco columbarius*), short-eared owl (*Asio flammeus*), greylag goose (*Anser anser*), mallard (*Anas platyrhynchos*), teal (*Anas crecca*), wigeon (*Anas penelope*), gadwall (*Anas strepera*), shoveler (*Anas clypeata*), tufted duck (*Aythya fuligula*), eider (*Somateria mollissima*), shelduck (*Tadorna tadorna*), red-breasted merganser (*Mergus serrator*), golden plover (*Pluvialis apricaria*), common sandpiper (*Actitis hypoleucos*), curlew (*Numenius arquata*), corncrake (*Crex crex*), common tern (*Sterna hirundo*). Ground nesting seabirds such as little tern (*Sternula albifrons*), arctic tern (*Sterna paradisaea*) (Clode & Macdonald, 2002) and arctic skua (*Stercorarius parasiticus*) also occur in significant numbers.

Historically, the introduction of mink in Scotland has been directly connected to the fur farming industry which was established in the 1950s (Dunstone, 1993; Bonesi & Palazon, 2007). In the Outer Hebrides this was mirrored when two fur farms on the Isle of Lewis went out of business in the 1960s resulting in a feral mink population becoming established (Angus, 1993). Small scale control operations carried out by sporting estates and an attempt by SNH to prevent the mink population spreading south had little effect. By 1999, breeding populations of mink were established on North Uist and Benbecula (Harrington, et al., 1999).

Invasive non-native species are one of the main causes of biodiversity loss worldwide (Genovesi, 2009) and predatory species, such as mink, can have a devastating impact on native species (Macdonald, et al., 2007). The need to manage non-native species is increasingly recognised as a necessity to minimise these impacts (Bryce, et al., 2011). In particular the impact of mink predation on ground nesting colonial seabirds can have a significant effect, on not only the breeding success of the species concerned but also the long term viability of the population (Craik, 1997; Craik, 1998). It is documented that mink at relatively low densities can also seriously affect salmonids (Areal & Roy 2006). Atlantic salmon (*Salmo salar*) is a species in decline, for which two Special Areas of Conservation have been established in the Outer Hebrides. The removal of mink can have significant beneficial consequences to a range of species, especially in island ecosystems (Nordström, et al., 2003).

In the Outer Hebrides the impacts of invasive mink over decades had become a significant concern and the most immediate effects were on the colonial nesting species such as tern which were being severely impacted both in terms of their productivity and also the loss of significant numbers of adult birds.

MATERIALS AND METHODS

The Outer Hebrides of Scotland are a highly complex archipelago of hundreds of islands which also includes the third biggest island in the UK, Lewis and Harris. It is characterised by vast expanses of moorland dissected

In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) (2019). Island invasives: scaling up to meet the challenge, pp. 261–266. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

by numerous convoluted freshwater lochs that amount to approximately 24% of the freshwater linear edge of Scotland's total. Due to the remoteness of some areas, and the general coastal nature of the American mink's behaviour, much of the work required the use of rigid hull inflatable sea-going boats that were used extensively, as well as Canadian open canoes in the complex freshwater habitats.

The project design was first established during the application process for LIFE funding but from its earliest conception it was regarded as an innovative trial of eradication techniques and an experimental project that required continuous critical appraisal of the progress being made.

Phase I of the Hebridean Mink Project was to remove all mink from the southern isles of the Outer Hebrides; South Uist, Benbecula and North Uist. The plan was also to reduce the mink density on South Harris to create a buffer zone between North Harris and Lewis (Helyar, 2005), minimising re-immigration, see Fig. 1.

Live capture traps were chosen as the core removal method due to the perception that kill traps were too much of a risk in terms of by-catch. Later in the project it was recognised that with experience, training and robust protocols these risks could be reduced to extremely low levels. Traps were made using 3 mm gauge wire mesh 18 × 15 × 60 cm and had galvanised steel doors. Caught mink were despatched using a .22 calibre air pistol.

From November 2001, for a period of three months, a total of 2,545 traps were dug into the ground and dressed in order that they became part of the landscape, although no more than 10% were open at any time. This provided a large number of pre-located traps, which could be used in

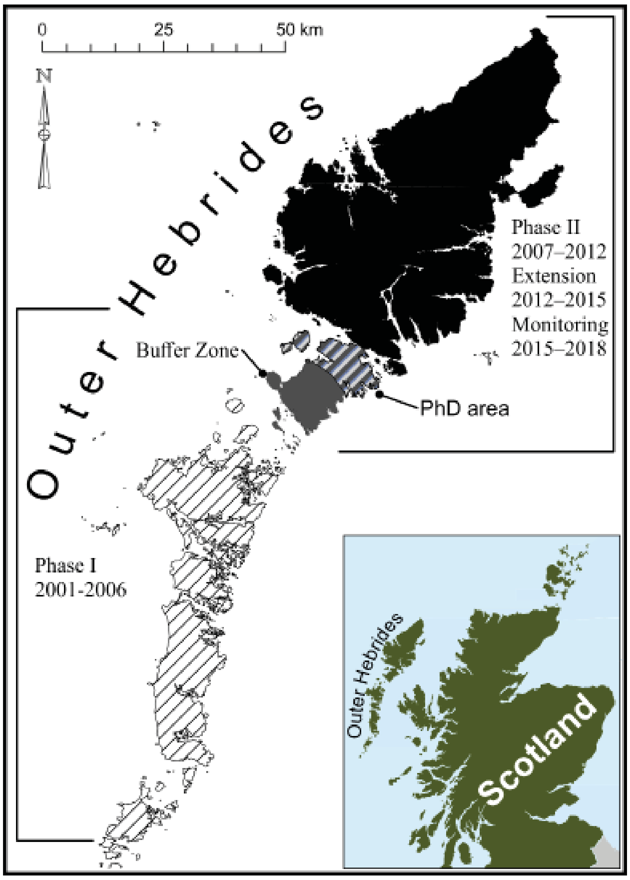

Fig. 1 The Outer Hebrides of Scotland showing the Hebridean Mink Project areas completed with timeframes.

rotation. The most efficient spacing of traps was established to be approximately 500 m apart, but with a higher trap density at individual den sites. Traps were initially baited with horse mackerel (*Trachurus trachurus*), that was later replaced or accompanied with anal gland lure which was more efficient (Roy, et al., 2006).

The team was comprised of a project co-ordinator, two trapper supervisors (one each on Harris and Uist), six permanent trappers, and seasonal/casual workers who assisted when required. The trappers worked a defined 37 hours per week, setting traps on a given route on a Monday and closing them on a Friday. This gave a weekly total of four trap nights per trap opened on any individual route. Traps were most efficient in the first few days of opening, see Fig. 2, and were left open for two weeks initially, reducing to one week in subsequent years.

During 2004 and 2005 the trapping was punctuated with high intensity trapping regimes. This involved co-ordinating a group of up to 25 individuals to trap simultaneously for a period of two to three weeks. The extra support was drawn from external organisational staff from DEFRA and the State Veterinary Service. The aim was to increase the likelihood of capturing any remaining, highly mobile mink.

Throughout Phase I, the most difficult areas to trap were the offshore islands. Two Rigid-Hulled Inflatable Boats (RHIB) were purchased and the associated training was given to trapping staff to enable them to reach all areas.

Dog searches were introduced as a technique during the summer denning period, when trapping is less efficient.

The final mink caught during Phase I was on 23rd March 2005 (see Fig. 2). This was followed by a further 5,567 trap nights and a 'summer' of dog searches with no further mink sighted or caught, bringing Phase I to an end in June 2006. In the interim between Phase I and II, two trappers were employed to keep the mink population low across the South Harris buffer zone.

Phase II of the project aimed to remove all mink from Harris and Lewis, to complete a full eradication from the entire Outer Hebrides. This project commenced in February 2007 and was initially due to end in March 2014, but at present is still ongoing.

Trap locations were pre-determined through the use of a GIS system. Trap positions were chosen by placing them at obvious intersections of linear riparian or coastal features, with 500 m buffer zones to ensure there were no geographical gaps. When in the field, staff were given a leeway of 50 m from the pre-positioned point to allow the

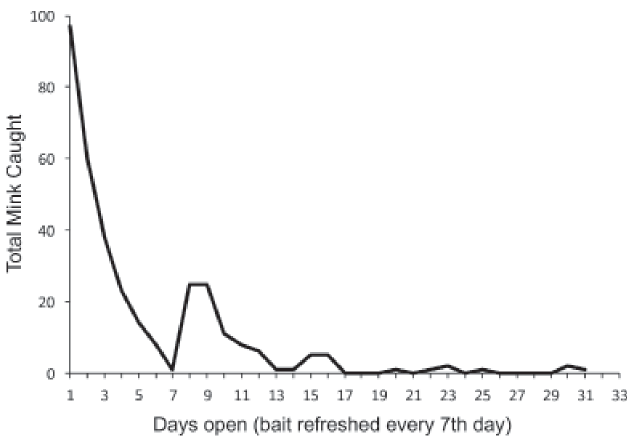

Fig. 2 The number of mink caught per length of time an individual trap remains open (SNH, 2006).

best position to be chosen in relation to the habitat. Once traps were installed, they were mapped on the GIS system to confirm absolute coverage of an area.

From 2007 there were 12 full-time trappers working 37 hours per week, reduced to six full-time trappers in 2012, and three in 2015. By 2008, 7,500 live capture traps had been permanently placed approximately 350–500 m apart, across Lewis and Harris. Traps within an area of approximately 100 km^2 were open at one time, for a period of four days. From 2008 to 2014, systematic trapping continued from the south-to-north, twice yearly. An exception was made in 2012 when the direction of trapping was altered to a north-to-south direction to ensure that specific areas were not always being trapped at the same time of year.

The mink population had been reduced to much lower densities by 2013 An assessment was carried out to ascertain whether the number of trap nights per 2.5 km^2 area was comparable, ensuring effort was distributed evenly across the entire project site. An extensive monitoring programme was set up in areas where there had previously been the highest mink densities, with 17 monitoring devices placed within 10 km^2 areas of interest. Monitoring devices included the use of footprint monitoring tunnels (clay and carbon plate), footprint monitoring rafts, camera traps and dog searches. These monitoring techniques were replaced with more efficient technology in the form of remote monitoring alarms (RMAs) which are activated when a trap is triggered. The monitoring devices are attached to a magnet which is pulled off when a trap is triggered, sending an SMS or email message to chosen team members. The devices were placed on traps situated in areas of good mobile phone coverage.

In 2014 the team reduced to three trappers. In order to maintain good monitoring coverage the live capture traps which had historically caught were replaced with 140 × 140 mm '116 Magnum bodygrip' spring traps contained in a bespoke designed wire mesh cage to exclude all non-target species. Over a period of two years almost 450 bodygrip traps were installed and 120 live capture traps were fitted with remote monitoring alarms.

Meanwhile on the Uist's, a few individual mink re-emerged in North Uist, which were immediately captured. In December 2014 another two mink were sighted in the northern end of North Uist, initiating another trapping project on the Uists. Staff from the Uist Wader project installed kill traps in a small area to detect any further mink. As more traps were installed, more mink were caught, and the trapping area was widened. From 2014 to the present there has been an increase in both trap nights and the number of mink caught on the Uists, with the trapping area now extending from North Uist down to Locheynort and due to be expanded to cover the entire Uists.

RESULTS

Phase I

A total of 532 mink from approximately 200,000 trap nights were caught during Phase I, see Table 1. Approximately half of those caught were on the Uists, compared with a similar number being caught in just the south of Harris (Fig. 3). This demonstrated that the mink population in the Uists had not yet reached carrying capacity, as south Harris has very similar terrain, and large areas of available habitat on South Uist had few captures. Between November 2004 and March 2005, only females were caught. This is likely a result of the trap density and the wider ranging behaviour of male mink. No mink were caught while trapping on the Uists between March 2005 and March 2006.

During the initial stages of Phase I it was quickly determined that the traps were most effective at catching during the first four days of being open. When opened for a further four days during the second week, the trap still caught mink but in far fewer numbers (Fig. 2).

In South Harris, due to a much higher trapper resource for the area available to trap, this number of trapping cycles per year was much higher, up to five times per year compared to just twice a year, and resulted in a very quick collapse in the territorial mink population. Thereafter, trapped animals were generally those immigrating, from the north, into the area, as indicated by a higher proportion of males caught during this period.

An important difference in the capture locations between the Uist's and South Harris became evident in the first few months of the project with the Uist's showing a significantly higher proportion of captures inland compared to coastal habitats. The difference was largely due to the

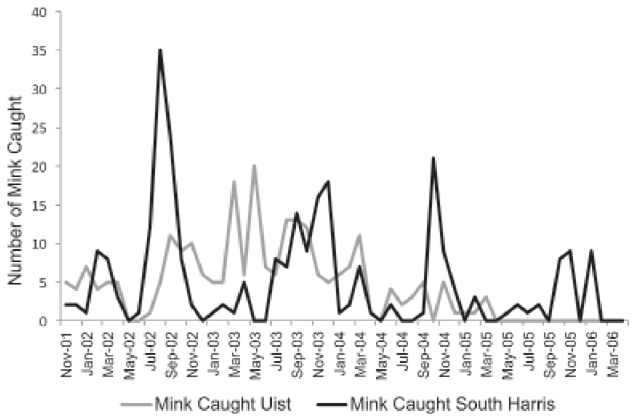

Fig. 3 Number of mink captured per month and year on both South Harris and the Uists during Phase I (SNH, 2006).

Year beginning	Trap nights		Mink captured by trapping		Mink captured per 1000 trap nights		Mink captured by dog searches (dependent young in brackets)	
	Uist	Harris	Uist	Harris	Uist	Harris	Uist	Harris
Sep-2001	22,155	15,350	42	73	1.85	4.76	0	6
Sep-2002	26,357	13,213	80	54	2.97	4.08	12 (18)	1 (2)
Sep-2003	30,064	10,325	56	64	1.86	6.20	4 (2)	(3)
Sep-2004	20,037	2755	13	38	0.65	13.79	1	3 (1)
Sep-2005	1,114	76	0	1	0	13.15	0	0
Total	100,824	41,674	191	230	1.89	5.15	37	18

Table 1 The numbers of trap nights, mink captures and trap successes in the Uists and South Harris during Phase I (Roy, et al., 2015).

greater availability of food resources inland in the Uist's, including a large number of duck and wader species closely associated with the freshwater edge but, importantly, the presence of field voles in the moorland habitat that were absent from Lewis and Harris.

Phase II

Trapping took place over large areas, only moving on when a low mink density had been achieved. This resulted in a significant drop to the overall mink population, with 51% of the final captures so far, being caught in the first two years. The final total of mink captures by March 2012 was 91% of the current figure. From April 2012 a further 116 mink were caught over the next three years, equating to a further 6% of the current final total.

Initially there was a two week live trapping cycle carried out but this was reduced to a one week cycle to increase the efficiency of the knock down phase. Whilst initially unpopular with the trapping staff, as they felt they were leaving animals behind, the speed with which the project reduced the mink population over a wide area soon became apparent and the staff bought into the techniques employed.

From 2007 to the present a total of 1,666 mink have been caught from 527,431 trap nights, across Lewis and Harris.

The major result of moving from live traps to a kill trapping regime was an increase in the total trapping effort, despite being reduced to a trapping team of just three. This can be seen in Table 2, where up to 14,000 trap nights per month were being achieved compared to approximately 2,000–2,500 per month when 12 trappers were employed for live capture trapping.

The captures per unit effort have declined over time but reflect the seasonality related to the trapability of more mobile mink during the rut and the naivety of young animals during the dispersal period (Fig. 4). The striking issue, however, is the extremely long tail to the graph which describes the extreme difficulty in catching the final animals over such a large geographical area with a declining staff resource (see Fig. 5). Two modelling exercises were completed, (Shirley, et al., 2012) and the modelling exercise carried out by Aberdeen University (Lambin, et al., 2014) did predict that this would be the case: 80% of

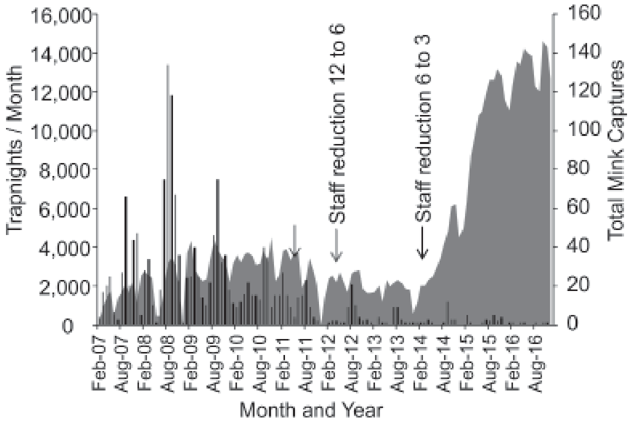

Fig. 4 Trap captures from Feb 2007–Nov 2016. Black bars are mink caught; grey area is the trapping effort.

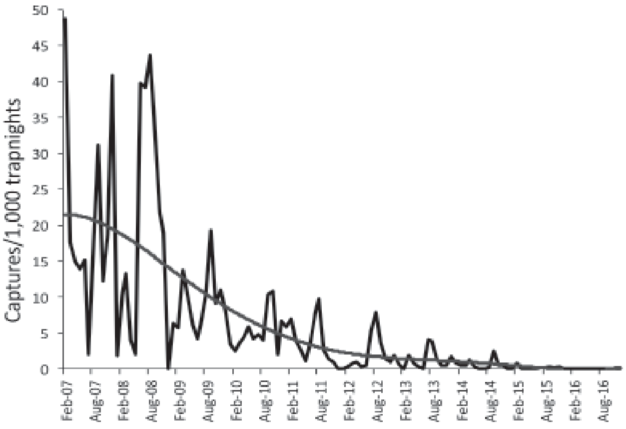

Fig. 5 Number of mink captures per 1,000 trap nights between February 2007 and December 2016.

iterations predicting eradication by 2017, using the data up to 2011 and a trapping regime based on live capture and a trapping regime of 12 trappers see Table 2.

During the final monitoring phase of the project in Lewis and Harris, the final 1.5% of the mink population was caught and functionally the population was eradicated with only isolated individuals, unable to find a mate and breed, left to track down. No juveniles have been caught in Lewis and Harris since August 2015

Through an increase in trapping effort and larger areas being monitored, there has been an increase in the number of mink being caught on the Uists since 2014 (Table 3). These animals are finally reducing in number as the same kill trapping regime used in Lewis and Harris takes effect.

Table 2 Actual trap nights and captures for all years of the project from 2007 onwards.

Year	Total trap nights	Total captures	Male	Female
2007	14,914	280	146	134
2008	24,755	527	266	261
2009	38,749	367	171	196
2010	40,894	212	98	114
2011	33,446	137	53	84
2012	26,665	56	31	25
2013	21,695	31	19	12
2014	41,954	26	16	10
2015	126,088	23	14	9
2016	158,271	7	5	2
2017*	87,000	4	3	1

*2017 figures to the end of June.

Table 3 Number of trap nights and mink caught from the re-emerged population in the Uists between 2014 and present.

Year beginning	Trap nights	Total captures	Male	Female	Unk
Jan 2014	36	5	5	0	0
Jan 2015	507	22	12	5	5
Jan 2016	3,776	63	38	21	4
Jan 2017*	4,799	41	23	14	3
Total	9,118	131	78	40	12

*2017 figures to the end of June.

DISCUSSION

In 2001 when this project was initiated, there were few, if any, successful eradications that used trapping as the main technique for the removal of an invasive non-native mammal; the only UK example being the coypu eradication in Norfolk, (Gosling & Baker, 1989). In addition, there was a limited range of literature available providing examples of wildlife management project design and best practice to follow (IUCN, 2000). The EU LIFE fund recognised that the project would need to adapt as it progressed and agreed to provide funding based on the understanding that it was innovative in its concept, scale and design.

During Phase I, one of the main lessons learnt was the necessity to ensure trap distribution was coordinated by the supervisor. Initially trappers were relied upon to distribute traps in the field according to their own judgement, with only a specific distance between traps to guide them. This meant that traps were situated in ideal locations for catching mink, but trappers on the ground were unable to ensure that there were no gaps in the overall trap coverage, leading to irregular densities. Over time, the emergence of better GPS technology enabled trappers to be more efficient in the field and able to provide more accurate trap locations. Establishing the most effective trapping schedule was important as it was not possible to trap the entire area at once with the staff available. A twice yearly minimum trapping cycle of the entire trap network was vital to ensure that all areas maintained sufficient trap nights, while removing animals in a timely manner to avoid successful breeding.

Despite the ongoing learning process during the first phase, the project managed to achieve the removal of the majority of the mink from the project area in just under three and a half years, followed by a summer of monitoring. It was thought at this point that it was very unlikely that any mink remained in the Uists and Benbecula and that eradication from these islands had been achieved.

The second phase of the project was an absolute requirement if the gains of the first phase were to be secured over an even larger geographical area and the investment in the previous five years was to be protected. Scottish Natural Heritage demonstrated significant commitment in proceeding with Phase II, helped with funding from the Esmeè Fairbairn Foundation, but from the outset the budgetary constraints on the project were clear. The modelling work undertaken by the Central Science Laboratory (now Animal and Plant Health Agency) indicated that 16 trappers would be ideal (Moore, et al., 2003) but due to budgetary constraints, the project proceeded with just 12. Restricted resources continued into the project extension and the monitoring phases and required significant adaptive changes to strategy and efficiency in order to give the project the greatest chance of success. It is undoubtedly true that the project has taken longer due to these budgetary constraints and that, if fully funded for the entire requirement of 10 years plus two extra years to ensure eradication, significant savings could have accrued over this period. This type of consecutive long-term funding is simply not available in the UK, (Lambin, et al., 2014), as it does not fit with the funder's requirement to demonstrate success, generally within five years, and exceeds the acceptable commitment levels between political administrations.

Throughout the project, different methods were employed at various stages to overcome the challenges of limited resources. The addition of the bodygrip traps instead of solely live traps enabled a high level of trapping effort to be maintained with limited staff. Bodygrip traps meant that trappers did not have to respond to triggered traps immediately as the mink would be dead upon capture. The initial concern of accidental by-catch was reduced to an acceptable level through very strict protocol in the practical setting of the trap, including the bespoke tunnels which excluded all non-target species, and camouflage technique.

Monitoring such a huge geographical area with only six trappers was challenging and several monitoring devices and techniques were trialled. Footprint rafts were not able to withstand the extreme weather of winter months either through wind or high water spate events, the cameras had slow triggers and reset times which led to missed targets, while the clay/carbon footprint monitoring required careful set-up and protection from the elements to provide useful data. In addition, the time between detecting the mink and being able to initiate the trapping was too long to catch a highly mobile individual. The acquisition of trap RMAs were particularly useful for the monitoring period, giving a precise time stamp for when a trap caught and enabling further traps to be installed in the area immediately. This was immediately effective as the mink population had begun to cluster in their distribution, not only during the rutting period which would be expected, but animals would also set up territories next to existing ones rather than be isolated and alone. This helped greatly once an individual was trapped, as a localised trapping campaign could be mobilised to catch a few additional animals.

The Hebridean Mink Project is now into its 16th year, and has cost a total of £5.26M. The learning process has been difficult and expensive and these lessons should be passed on to others. There is a requirement for simple tools to be developed that will allow projects to recognise the key stages of eradication from the data they collect. These comprise: population crash completion (knock down), identification of groups of target species (cluster effect) and difficult to trap areas to allow targeted action (trap everywhere at the same intensity), detection of individuals and their rapid removal (find the right monitoring technique), effective and efficient long-term monitoring and biosecurity (ensure the last individuals are not left behind or re-introduced).

Clearly there are vast amounts of data associated with this project that could provide a lifetime of analysis opportunities of which only a tiny fraction has been used here. Some of the intuitive assumptions made within this paper need to be statistically analysed to provide definitive proof of behaviours such as clustering, which appear so obvious from mapping the capture data geographically over time.

CONCLUSIONS

Phase II of the Hebridean Mink Project commenced with a wealth of knowledge, practical scientific information, techniques and trapping scheme models, not to mention a core of well-trained staff. This no doubt contributed to the success in greatly reducing the population of American mink to near eradication. With the re-emergence of mink in the Uists, the main lesson that can be learnt from Phase I, is the importance of ensuring a sufficiently long monitoring period with a sustained level of effort is implemented once the last mink is thought to have been captured. Maintaining sufficient resources to continue monitoring during the final years following eradication is crucial to ensuring the project's success (Rout, et al., 2009). Any lapse in funding before eradication is declared could result in the mink being able to breed successfully and repopulate, leading to financial losses that are both immediate and exponential.

If eradication can be achieved in the Outer Hebrides this would represent the largest mammalian eradication initiative worldwide using just trapping techniques.

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Most of all, a huge thanks to all who have been involved with the project over the past 17 years. Too many to name individually, this has been a collective long-term achievement due to the hard work of hundreds of people: in particular the trapping staff, who have walked the equivalent of several times round the world in order to check the trapping network in the worst of weather that the west coast of Scotland could throw at them. Lastly, but not least, the people of the Outer Hebrides for their help and co-operation throughout the project. Thank you all.

REFERENCES

Angus, S. (1993). ‘A mink control programme for Lewis and Harris’. *Hebridean Naturalist* 11: 78–84.

Areal, F. and Roy, S.S. (2006). ‘A management decision tool for mink (*Mustela vison*) control in the Western Isles of Scotland (UK)’. *International Journal of Design and Nature and Ecodynamics* 4: 16–31.

Bonesi, L. and Palazon, S. (2007). ‘The American mink in Europe: Status, impacts, and control’. *Biological Conservation* 134: 470–483.

Bryce, R., Oliver, M.K., Davies, L., Gray, H., Urquhart, J. and Lambin, X. (2011). ‘Turning back the tide of American mink invasion at an unprecedented scale through community participation and adaptive management’. *Biological Conservation* 144: 575–583.

Clode, D. and Macdonald, D.W. (2002). ‘Invasive predators and the conservation of island birds: The case of American mink *Mustela vison* and terns *Sterna spp*. in the Western Isles, Scotland’. *Bird Study* 49: 118–123.

Craik, C. (1997). ‘Long-term effects of North American mink *Mustela vison* on seabirds in Western Scotland’. *Bird Study* 44: 303–309.

Craik, J.C.A. (1998). ‘Recent mink-related declines in gulls and terns in west Scotland and the beneficial effects of mink control’. *Argyll Bird Report* 14: 99–110.

Dunstone, N. (1993). *The Mink.* London: T & AD Poyser.

Genovesi, P. (2009). ‘Invasive alien species in a changing world’. *Biodiversity* 10: 3–4.

Gosling, L.M. and Baker, S.J. (1989). ‘The eradication of muskrats and coypus from Britain’. *Biological Journal of the Linnean Society* 38: 39–51.

Harrington, L.A., Grogan, A.L., Bonesi, L. and Macdonald, D.W. (1999). *Survey of American Mink, Mustela vison, on North Uist*. Unpublished report to Scottish Natural Heritage. Inverness: Scottish Natural Heritage.

Helyar, A. (2005). ‘The Ecology of American Mink *(Mustela vison)*: Response to Control’. PhD thesis. York, UK: University of York.

IUCN. (2000). *Guidelines for the Prevention of Biodiversity Loss Caused by Alien Invasive Species*. Information Paper for CBD COP 5. Gland, Switzerland: IUCN.

Lambin, X., Cornulier, T., Oliver, M.K. and Fraser, E.J. (2014). *Analysis and Future Application of Hebridean Mink Project Data. Scottish Natural Heritage Commissioned Report No. 522*. Inverness: Scottish Natural Heritage.

Macdonald, D.W., King, C.M. and Strachan, R. (2007). ‘Introduced Species and the Line between Biodiversity Conservation and Naturalistic Eugenics’. In: D.W. Macdonald and K. Service (eds.) *Key topics in Conservation Biology*, pp. 186–205. Oxford, UK: Blackwell Publishing.

Moore, N.P., Roy, S.S. and Helyar, A. (2003). ‘Mink (*Mustela vison*) eradication to protect ground-nesting birds in the Western Isles, Scotland, United Kingdom’. *New Zealand Journal of Zoology* 30: 443–452.

Nordström, M., Högmander, J., Laine, J., Nummelin, J., Laanetu, N. and Korpimäki, E. (2003). ‘Effects of feral mink removal on seabirds, waders and passerines on small islands in the Baltic Sea’. *Biological Conservation* 109: 359–368.

Rout, T.M., Thompson, C.J. and McCarthy, M.A. (2009). ‘Robust decisions for declaring eradication of invasive species’. *Journal of Applied Ecology* 46: 782–786.

Roy, S.S., Chauvenet, A.L.M. and Robertson, P.A. (2015). ‘Removal of American mink (*Neovison vison*) from the Uists, Outer Hebrides, Scotland’. *Biological Invasions* 17: 2811–2820.

Roy, S.S., Macleod, I. and Moore, N.P. (2006). ‘The use of scent glands to improve the efficiency of mink (*Mustela vison*) captures in the Outer Hebrides’. *New Zealand Journal of Zoology* 33: 267–271.

SNH. (2006). *Mink Control to Protect Important Birds in SPAs in the Western Isles*. Final Report to EU LIFE III–Nature, September 2006. Scottish Natural Heritage. Inverness: Scottish Natural Heritage.

Shirley, M., Rushton, S. and Lurz, P. (2012). *Simulation Model for the Management of American Mink, Neovison vison, on Harris and Lewis*. Scottish Natural Heritage, Commissioned Report No.452. Inverness: Scottish Natural Heritage.

Stroud, D.A., Chambers, D., Cook, S., Buxton, N., Fraser, B., Clement, P., Lewis, P., Maclean, I., Baker, H. and Whitehead, S. (2001). *The UK SPA Network: Its Scope and Content*. Three vols. Peterborough, UK: Joint Nature Conservation Committee.

Ecological restoration of Socorro Island, Revillagigedo Archipelago, Mexico: the eradication of feral sheep and cats

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Abstract Socorro Island is part of the Revillagigedo National Park, Mexico. At 132 km^2, it is the Mexican island with the highest level of endemism. It provides habitat for 117 vascular plant species, 26% of which are endemic. There is also an endemic blue lizard (*Urosaurus auriculatus*) and eight endemic terrestrial birds. Socorro's ecosystem had been heavily degraded by invasive mammals for the past 140 years. Feral sheep (*Ovis aries*) destroyed one third of the island's habitat and feral cats (*Felis catus*) severely impacted the island's avifauna and the Socorro blue lizard. Together, feral sheep and cats are responsible for the extinction in the wild of the Socorro dove (*Zenaida graysoni*) and the Socorro elf owl (*Micrathene whitneyi graysoni*) and have been a serious threat to other vulnerable species, particularly Townsend's shearwater (*Puffinus auricularis*). As such, the island's restoration is a high priority. We conducted a feral sheep eradication from 2009 to 2012, using aerial and terrestrial methods, aided by Judas sheep and trained dogs, to kill 1,762 animals. The vegetation recovery has been remarkable, as well as the improvement of soil properties such as compaction, nitrogen, organic carbon, phosphorus, and calcium. In 2011, we initiated a feral cat control programme, which soon became an eradication project. The ongoing feral cat eradication has been a challenge, due to Socorro's large size, vegetation and topographical complexity. By December 2016, 502 cats had been dispatched, using soft leg-hold traps equipped with telemetry transmitters and lethal traps: a total effort of 50,000 trap-nights. Cat abundance has decreased very significantly and catch per unit of effort indicates that the eradication is nearing completion. The abundance of the Socorro blue lizard and terrestrial birds has already increased. We estimate completing the feral cat eradication by the end of 2017, when we will shift to a verification of eradication phase.

Keywords: exotic mammals, habitat recovery, outcomes of eradications

INTRODUCTION

Mexican islands are known for their high biodiversity richness. They are home to many endemic species and are important breeding grounds for a variety of birds and marine mammals (Aguirre-Muñoz, et al., 2011). Unfortunately, these ecosystems are suffering serious impacts resulting from human activity. Exotic species are among the main causes of biodiversity loss and ecological disequilibrium in many environments (Courchamp, et al., 2003). Herbivores, like feral sheep (*Ovis aries*), have caused serious ecological impacts on insular ecosystems. In 1869, 100 sheep were introduced to Socorro Island, in Revillagigedo National Park, Mexico (Fig. 1) for ranching. Over time, they became feral, successfully adapting to island conditions (Levin & Moran, 1989; Álvarez-Cárdenas, et al., 1994; Brattstrom, 2015). In the absence of natural predators, the sheep population grew to be about 5,000 individuals by 1960 (Villa, 1960). This reduced to around 2,000 in 1988 as a result of increased hunting by the Mexican Navy (Walter & Levin, 2008), but they became the main cause of the island's poor ecological condition (Richards & Brattstrom, 1959; Veitch, 1989). Since their introduction, feral sheep have caused huge modifications to the natural habitat. Erosion rates and loss of vegetation caused by the presence of sheep were documented, along the southern-central region of the island (León de la Luz, et al., 1994; Maya-Delgado, et al., 1994; Rhea, 2000). Nearly 30% of the original soil and vegetation on Socorro Island was lost due to erosion caused by feral sheep (Ortega-Rubio, et al., 1992). Among the most significant changes to the original floral composition has been an increase in the presence of grasses and shrub species, as well as a reduction in the area covered by native flora. Sheep aid the propagation of introduced plant species, dispersing seeds in their coat and excrement. The change in native vegetation has been observed in every habitat that sheep occupied (SEMARNAT, 2004). Another serious threat is the feral cat (*Felis catus*), which have severely impacted the island's bird communities and the endemic Socorro tree lizard (*Urosaurus auriculatus*) (Arnaud, et al., 1993; Arnaud, et al., 1994). Together, feral sheep and cats are responsible for the extinction in the wild of the Socorro dove (*Zenaida graysoni*) and the Socorro elf owl (*Micrathene whitneyi graysoni*), and pose a serious threat to other vulnerable species, such as Townsend's shearwater (*Puffinus auricularis*) (Martinez-Gomez & Jacobsen 2004). The eradication of feral cats represented another serious challenge, as Socorro is a large and complex island, and little baseline information was available on the distribution and abundance of cats (Arnaud, et al., 1994). Fortunately, technologies have been developed on other islands of Mexico and the world to achieve the eradication of these predators (Bester, et al., 2002; Wood, et al., 2002; Algar, et al., 2010; Aguirre-Muñoz, et al., 2011; Luna-Mendoza, et al., 2011; Parkes, et al., 2014). The successful implementation of an eradication campaign of this type is essential to determine the basic aspects of the species, the impact of the methods applied on the native fauna, and the

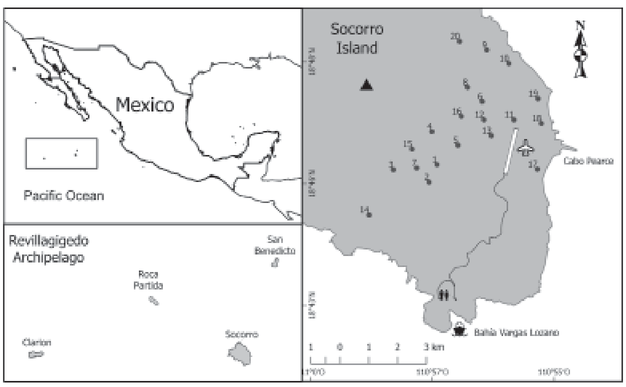

Fig. 1 Location of sampling sites of vegetation and soil.

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development of an official eradication plan (Veitch 1989; Arnaud, et al., 1994; Donlan, et al., 2003; Dowding, et al., 2009).

MATERIALS AND METHODS

Study site

The volcanic Socorro Island is the largest and most diverse portion of the Revillagigedo Archipelago, a World Heritage site in Mexico that was listed by UNESCO in July 2016. It is located in the tropical eastern Pacific, 460 km from the Baja California Peninsula and 700 km from Manzanillo, Colima. It has an area of 132 km^2 and a maximum altitude of 1,040 m (18º 47´ N, 110º 58´ W). Due to its remoteness, the island is a strategic point for Mexico's military personnel (40–50 people) situated at a naval base located in the southernmost part of the island. Additionally, Socorro Island has critical biodiversity significance through a high level of endemism, due to its isolated position A remarkable number of its native biota are exclusively found in this part of the world. Approximately one third of the 118 species of native vascular plants inhabiting the island are endemic. The native fauna is comprised of one endemic reptile species and almost 101 species of birds, of which eight of the terrestrial birds are endemic (SEMARNAT, 2004).

Feral sheep eradication

Monitoring: To identify the main areas where sheep were distributed, several flights (on a Beechcraft Bonanza aircraft) were made over the island during October 2005. At the same time, Mexican Navy officers conducted land surveys on foot and in motorised vehicles (ATV´s).

Aerial hunting: The aerial hunting stage was carried out using a single turbine helicopter (model MD369D), between April 20 and 29, 2009; supported by a GPS to record the flight trajectories. Two hunters were shooting simultaneously during the flight, using semi-automatic rifles and shotguns. All flights took place between 07:00 and 11:00 h, and between 16:00 and 19:00 h; at an average speed of 42 km/h and average height of 35 m.

Judas sheep: During the aerial hunting, 12 live animals were captured, to be used as 'Judas' sheep (individuals that serve to help locate remaining herds; Taylor & Katahira, 1988). These animals were neutered and fitted with radio-telemetry collars (Telenax, Mexico). These Judas sheep were deployed back to the sites where they were captured.

Terrestrial hunting and trapping: From February 2010 to April 2012, 4–7 experienced hunters carried out terrestrial hunting. Each one was equipped with a handheld GPS to record their hunting tracks, rifles (calibre .222, .243 and .308) with telescopic sights, as well as a 12-gauge shotgun with cartridges. Supported by the Judas sheep, it was possible to identify sheep herds. Simultaneously, leg-hold traps (Oneida Victor Soft Catch # 3) and snare traps were used on previously identified sheep trails; both types of trap were checked daily.

Hunting dogs: for the last stage of the sheep eradication, we used two hunting dogs (beagle and foxhound) to track down the remaining sheep herds; the dogs were fitted with GPS collars to record their locations and movements (Ortiz, et al., 2016a).

Feral cat eradication

Trapping: The eradication method consisted mainly of catching cats using leg-hold traps (#1 ½) and lethal traps (Conibear Bodygrip traps 10": Rauzon, 1985; Twyford, et al., 2000; Phillips, et al., 2005; Rodríguez, et al., 2006; Rauzon, et al., 2008; Luna-Mendoza, et al., 2011). Leg-hold traps with pads were used in 220 sites over the duration of the expedition (21–51 days), baited with a commercial cat bait made of seafood, tuna or fried sardine (Brothers, 1982). Traps were checked daily from 7:00 to 10:00 h. Lethal and leg-hold traps were located in sites of difficult access, equipped with telemetry systems (ATS, mammal trap monitor Series M4000) to determine whether they had been activated from a distance (Will, et al., 2010). Once cats were captured, these were euthanised by intramuscular injection of an anaesthetic and lethal intracardiac injection (pentobarbital). As a secondary method, night hunting was conducted using .222 calibre rifles with telescopic sights, and lamps (Kohree 80,000 lux: Ortiz, et al., 2016a).

Soil quality assessment

Soil compaction: In 2013, soil penetration resistance measurements were taken using a penetrometer (Soil Compaction Tester Dickey-john®) within 20 vegetation transects (Fig. 1). Sites were categorised as: bare soil sites, those with 50% recovered vegetation and those with 100% recovered vegetation. Additionally, soil compaction measures were taken in sites with 100% vegetation coverage, not previously disturbed by the sheep (ND = not disturbed). Fifty replicates were obtained in each category, resulting in a total of 200 measurements. An analysis of variance and Tukey's honest significance test were performed to analyse the differences among the different categories of vegetation cover.

Physicochemical soil parameters: soil samples of approximately 1 kg each, were collected from each transect at a depth of 0–10 cm: 16 samples were obtained in each one of the soil categories (N= 64). Subsequently, the following physicochemical parameters were determined: pH and electrical conductivity, total nitrogen by the Dumas method in a LECO nitrogen analyser; organic matter by the method of Walkley-Black; phosphorus by colorimetric reading of a spectrophotometer, and calcium and magnesium by the EDTA method.

Vegetation recovery assessment

Field assessment: Prior to eradication, in 2009, vegetation data collection was started to obtain a baseline scenario of the degraded environment, with the aim of making subsequent comparisons possible, and to detect signs of recovery after sheep removal. The estimation of sheep overgrazing consequences on the island was determined by selecting 20 plot sites. Transects of 10 m × 100 m were established in the more disturbed areas, to identify pioneer species on eroded soils; all plants were identified and counted. Plot sites were categorised in: forest (six replicates), mixed scrub (six replicates) and eroded surface (seven replicates) (León de la Luz, et al., 1994). The vegetation monitoring continued from 2011 to 2016. Analysis of variance was performed (rANOVA) to analyse differences in vegetation cover and in the number of species over the years of the study (Ortiz, et al., 2016b).

Normalised Difference Vegetation Index (NDVI): To identify changes in vegetation cover the photosynthetic vegetation vigour of the island was obtained, quantified with the Normalized Difference Vegetation Index (NDVI). Supported with QGIS software, two maps were generated. A "pre-eradication" map, created using a QuickBird satellite image, dated on May 11, 2008; and a "post-eradication" map, generated with a WorldView 2 satellite image, dated on May 9, 2013. Finally, the change in vegetation cover between the two dates was determined by subtracting the 2008 image NDVI raster pixel image values from the image of 2013, considering only differences exceeding 0.2 (bare soil).

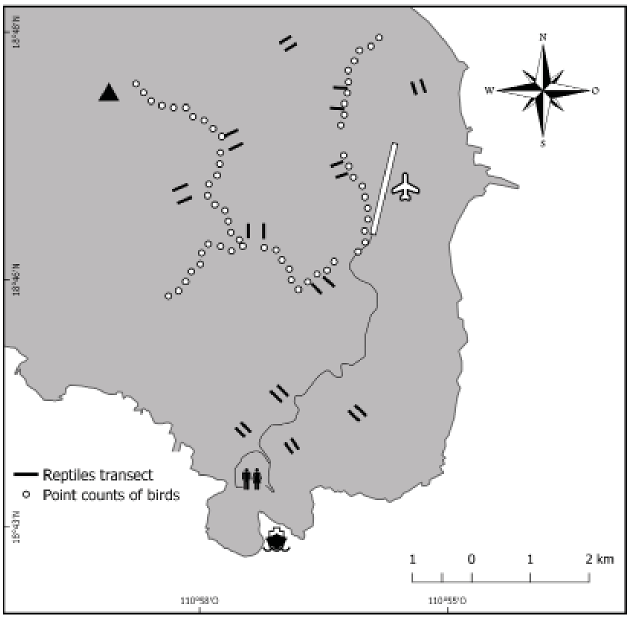

Fig. 2 Location of transects of Socorro tree lizard (lines) and count-points for birds (quadrats).

Monitoring of native fauna

Socorro Island tree lizard: To evaluate if the vegetation recovery was promoting any improvement of native animal populations, we monitored the Socorro Island tree lizard, during April and October, from 2012 to 2017. Twenty-four transects were set up in three different types of habitats (eight replicates per habitat type): forest, deciduous scrubland and eroded surfaces or areas impacted by sheep (León de la Luz, et al., 1994), each measuring 6 m × 100 m (Fig. 2). Transects were each visited on three consecutive days, between 10:00 and 12:00 h, during two different seasons (dry and rainy). Density (D) was estimated using the formula: $D = (n/2wL)$, where n is the number of individuals recorded, L is the total transect length, and w is the width of the transect on each side of the line (Gallina & López-González, 2011). A one-way repeated measure analysis of variance (rANOVA) was conducted to determine differences in seasonality and in habitat type on tree lizard density, the statistical software R (Version 3.2.2) was used for the analysis.

Terrestrial birds: Terrestrial birds were also monitored, using the point-count technique. Six transects were established during April and October (two seasons per year) from 2012 to 2017 (Fig. 2). The monitoring was carried out from 6:30 to 9:30 h and was repeated on three consecutive days, during the dry and rainy season, respectively. At each site, all birds observed within a radius of 25 m in a time span of five minutes were counted. Subsequently, the observer moved to the next counting point, located 250 m away, with a five-minute break before starting the next count. The statistical test rANOVA was run to determine the effect of season and habitat type on the total number of birds, plus Student t-tests for paired samples with a Bonferroni adjustment, to compare sightings during the different seasons.

RESULTS

Sheep eradication

Aerial hunting: a total of 35 flight hours was achieved in one week during the aerial hunting stage, in which most of the island was covered, with an average flight time of 1h 20 min per event; this resulted in removal of 1,257 individuals. The aerial hunting ceased when sheep became difficult to locate, and relatively few animals were being shot within a flight event. This method was selected due to its proven effectiveness in achieving rapid eradication (Campbell & Donlan, 2005) and was ideal on this island owing to its tropical conditions, which allowed the carcasses to decompose rapidly.

Ground hunting and trapping: 505 sheep were dispatched during the ground hunting stage, which comprised a nine-month period of hunting, over two years (March 2010–April 2012). Judas sheep were mostly effective when there were more remaining sheep, due to an increased probability of aggregation. Hunting dogs were used only at the final stage of eradication to locate the last ten remaining animals, which were difficult to locate for hunters. A total of 1,762 sheep were dispatched from Socorro Island in a three-year eradication campaign (Table 1).

Feral cat eradication

By December 2016, 502 cats had been removed, using soft leg-hold traps equipped with telemetry transmitters and lethal traps (body grip). Traps were placed in more than 250 sites on the island. Up to that date, there was an effort of more than 50,000 trap-nights. The success of cat capture during the trapping period fluctuated throughout the year (greater catch in January–May, dry season; and lower catch in June–October, rainy season). However, a clear trend to a smaller population was noted at a multi-year timescale (Fig. 3). In general terms, the success of capture is greater in the dry season and decreases during the rainy season. It is expected that cat eradication will cease in 2018; if that is the case, absence confirmation monitoring will be carried out in 2019.

Soil quality assessment

The results of the soil compaction assessment showed that eroded soils were the most compacted and a trend towards compaction reduction on areas with recovered

Table 1 Feral sheep dispatched on Socorro Island.

Year	Months	Personnel	Hunter hours	Distance (km)	Judas sheep	Trap nights	Captured sheep	Dog hours	Sheep removed
2009	May		35 (helicopter)						1,257
2010	Mar–Apr	7	1,323	815	53	900	41	-	355
	Jul	6	588	460	18	-	-	-	48
2011	Apr	5	512	433	11	650	8	-	67
	Aug–Sep	4	728	644	4	-	-	-	25
	Nov–Dec	4	420	385	-	-	-	49	8
2012	Apr	4	240	216	-	-	-	-	2
Total			3,811	2,953	86	1,550	49	49	1,762

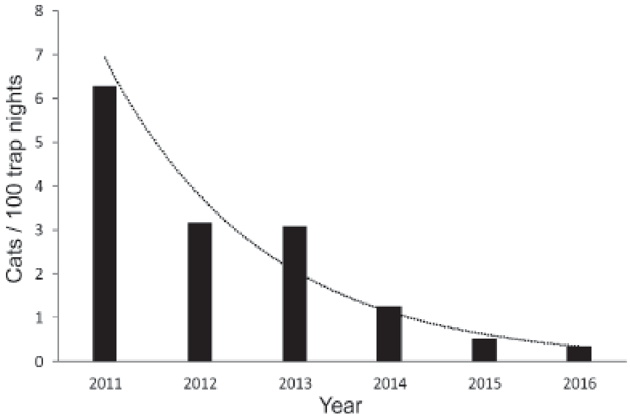

Fig. 3 Success of capture of feral cats.

vegetation was observed (50% and 100% recovered vegetation cover). Transects that retained eroded soils (0% vegetation) because of the sheep trampling, showed greater penetration resistance (>300 pounds-force per square inch, or psi, to 12 inches deep); at sites with 50% and 100% recovered vegetation cover, soils were also compacted and shallow (100–120 psi to three inches deep) and became more compacted at greater depths (300 psi to 24 inches deep); at sites with 100% coverage without disturbance (ND), the soil showed little variation (230–300 psi until 21 inches depth), which was in the optimal range for the growth of most plants (from 200–400 psi to 24 inches), which could be due to the constant, stable conditions. Significant differences ($p < 0.001$) were observed among sites with 0% and those with 50% and 100% recovered vegetation cover. The results of physicochemical analyses of soil samples showed increased nutrients: pH values remained close to neutral, showing a significant difference ($p < 0.021$) between sites without vegetation and 50% vegetation cover (results in Ortiz, et al., 2016b). Electrical conductivity, which is an indicator of salt presence in soil, was also significantly different ($p < 0.013$) between the eroded and 100% vegetation covered sites, although no difference was observed between eroded soils and those that were not disturbed. In the case of total nitrogen, organic carbon, phosphorus and calcium, sites with recovered vegetation were significantly different ($p < 0.001$) to those with erosion. Both nitrogen and organic carbon doubled, while phosphorus and calcium values almost tripled in places with increased vegetation cover compared to the eroded sites. Meanwhile, magnesium showed significant differences among the eroded sites (0%, 50%, and 100% recovered vegetation cover) and undisturbed sites (100% ND; Ortiz, et al. 2016b). The sites that were never altered by the presence of sheep exhibited a magnesium concentration twice that of disturbed sites.

Vegetation recovery assessment

Calculations (comparison of the images from 2008 and 2013) showed a difference of 1452 ha, which is equivalent to vegetation recovery of 11% of the island surface. The eastern part of the island was the area with the greatest habitat disturbance (Álvarez-Cárdenas, et al., 1994), and where the greatest vegetation recovery seemed to have occurred within the analysed period. Due to the presence of sheep, most of the evaluated sites lacked vegetation, and few species were present in 2009 (Fig. 4). Additionally, trails made by the sheep were observed to have compacted soils. Statistical tests showed significant differences from 2009 to 2013 in the number of species present in the eroded sites as well as in percentage cover. It was possible to record obvious recovery in all the habitats in 2013, i.e. the forest habitat with the highest values, followed by the mixed scrub, and then the eroded surface.

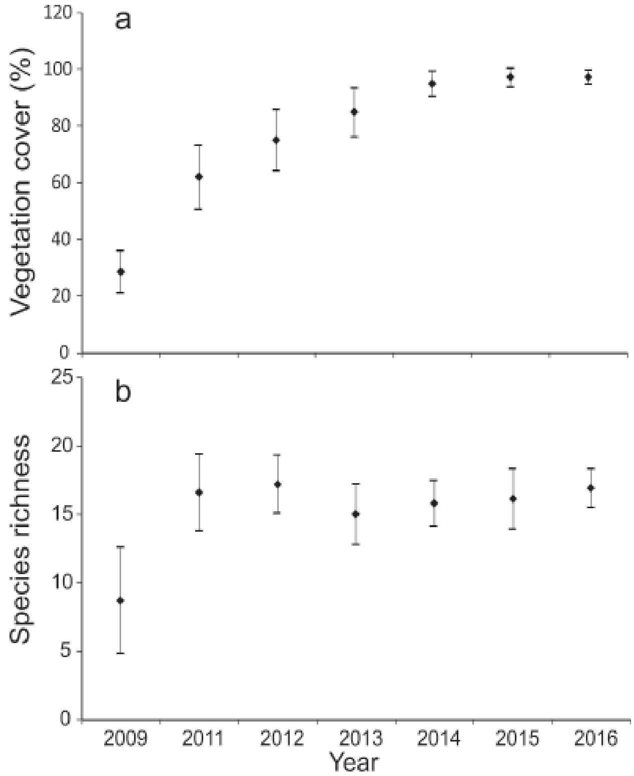

Fig. 4 Increase in vegetation cover (a) and species richness (b) on Socorro Island.

Monitoring of native fauna

The results of tree lizard monitoring reveal that the population is increasing, taking into consideration both the dry and rainy seasons (Fig. 5). Lizard density fluctuated significantly between seasons since the trapping of cats started on Socorro Island ($p = 0.014$). The number of birds sighted from 2012 to 2015 also showed significant differences between seasons ($p = 2.2 \times 10^{-4}$). Although population fluctuation is evident over the years of the study, there is an increase of birds in the dry seasons of 2014 and 2015 (Fig. 6). Significant differences were found between November 2012 and the rest of the monitoring time points (except November 2014). No significant differences were found between dry seasons during the years 2013 to 2015. The most abundant species was the Socorro tropical warbler (*Setophaga pitiayumi graysoni*), followed by the Socorro wren (*Troglodytes sissonii*), and the towhee (*Pipilo maculatus socorrensis*), all of them endemic to the island.

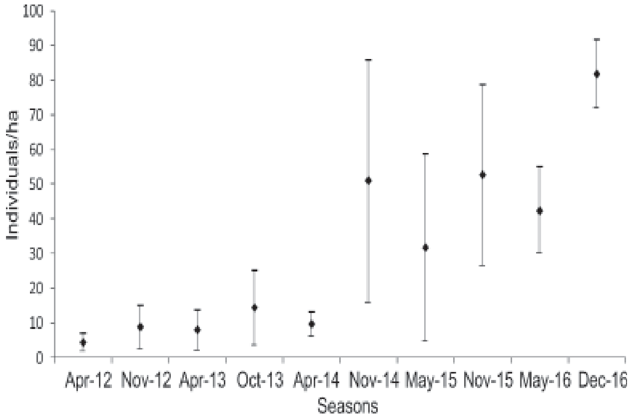

Fig. 5 Density of the Socorro tree lizard.

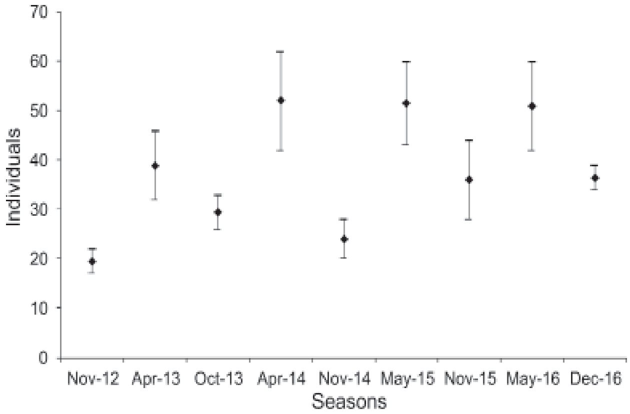

Fig. 6 Numbers of birds sighted.

DISCUSSION

Compared with other islands where goats or sheep have been eradicated (Van Vuren, 1992), the Socorro Island feral sheep eradication can be considered highly effective as it was completed in three years, when similar projects have taken 3–5 years, and even decades, to conclude (Campbell & Donlan, et al., 2005). Moreover, the methods used reduced project cost, which was US$ 38/ha, while other sheep eradication projects, such as that developed on Santa Cruz Island, California, cost US$ 80/ha, due to capture and transport of sheep to the continent (Faulkner & Kessler, 2001).

The capture of cats increases during the dry season (January–May) and decreases in the rainy season (June–October). The results differ between wet and dry seasons, since moist land interferes with the installation of leg-hold traps, and dry substrate is unavailable to cover them. Rain also compacts and hardens the substrate covering the traps, hence restraining their activation. At the same time, another key factor that affects trapping in the rainy season is the higher abundance of land crabs (*Gecarcinus planatus*), which consume the bait placed in traps, or activate traps when attempting to reach the bait. The combination of lethal traps and telemetry devices is essential during trapping in the most remote areas of the island. In this way, traps do not have to be checked daily but every five to seven days any bait lost to insects (mainly ants) and crabs is replenished (Parkes, et al., 2012).

The changes in soil physicochemical properties on Socorro Island seem to be related to the gradual recovery of vegetation after the eradication of feral sheep. Prostrate *Chamaesyce* sp. and *Erigeron socorrensis* have been observed to have a great capacity to retain soil. *Hyptis pectinata* and *Pteridium caudatum* established in high densities; in addition to retaining soil, they have generated much organic matter. Possibly the most successful species to colonise disturbed areas has been *Dodonaea viscosa*, which has a great ability to germinate in eroded soil (Campa-Molina, 1989), generating organic matter and preventing the germination of other species (Castellano & Valone, 2007). In the absence of trampling, soil aggregate stability increases, which enhances permeation, reduces erosion, and may promote nutrient accumulation and soil retention (Allington & Valone, 2010). As pioneer plants began to establish, the ground became less compacted because the roots of plants, particularly annual grasses in this instance, act as biological perforators, also incorporating organic matter into the soil. Once the roots die and shrink, these pores are large enough to allow the roots of perennial shrubs to penetrate (Sellés, et al., 2012). Greater ease of water movement in the soil matrix, coupled with heavy rainfall, could be causing leaching and replacing cations with H+ ions, acidifying the soil.

Both the results obtained with the NDVI calculation and field observations suggested that some pioneer plants had the ability to germinate on eroded soils and were instrumental in the succession process by providing the right conditions for seeds of tree species to germinate. The progressive increase in vegetation cover reduces soil compaction and restores the biogeochemical cycles of essential nutrients, such as nitrogen, phosphorus, and calcium, which are essential for the recovery of communities and the ecosystem in general, as well as the incorporation of carbon on the ground, which is essential for the proper functioning of important microbiological components. Any change in the habitat that produces changes in litter production, soil aeration, or any other factor affecting microorganisms will be reflected in changes in biogeochemical cycles, such as those of carbon and nitrogen (Hartmann, et al., 1997).

We found differences in the number of species and vegetation cover in the sampling area between 2009 and 2013. The forests and mixed scrub areas showed the greatest recovery, probably favoured by their vegetal components and the permanence of seed banks, due to a more stable landscape, water availability, and precipitation patterns. The endemic tree species recovering were *Guettarda insularis* and *Psidium socorrense*. The smaller number of plant species found in the isolated patches of mixed scrub included in large expanses of erosion could be due to the steepness of slopes and wind exposure. Gravity also makes the permanence of naturally occurring soil seed banks difficult. Some species of exotic grasses have increased with sheep eradication because they are no longer grazed.

The Socorro Island tree lizard was found at higher densities in the deciduous scrubland, being less abundant in forests at higher altitudes. The results of this particular study show that the density of lizards on eroded surfaces was as high as 43 individuals/ha after cat abundance was reduced, however Galina et al. (1994) reported not having observed lizards in these areas. This may be due to a gradual recovery of the vegetation resulting from the recent eradication of sheep (Ortiz-Alcaraz, et al., 2016a; Ortiz-Alcaraz, et al., 2016b) and to the sustained trapping of cats in these areas. Lizard abundance was slightly higher during the rainy season, likely due to greater food availability. As the cat eradication programme in the eastern area of the island has progressed, the predation pressure of cats on the lizard population has decreased. Lizards are a major component of the cats' diet (50% of faecal samples of cats analysed contained lizard remains; Arnaud, et al., 1993). The vegetation type where the highest number of birds was observed was the forest (*Ficus-Guettarda-Ilex*), especially in the highest parts of the island, where the recovery of vegetation resulting from the absence of grazing sheep has led to greater availability of food and shelter against predators (Rodríguez-Estrella, et al., 1994). On the other hand, special efforts have also been made to eradicate cats in the forest, aiming to protect the native bird species, such as the Townsend´s shearwater (Ratcliffe, et al., 2009).

The plans developed by Veitch (1989), Arnaud, et al. (1993) and Parkes, et al. (2012) are all in agreement that feral cats can be eradicated using traditional techniques: trapping and night hunting. However, the experience on the island has shown the importance of using detection dogs to locate the remaining cats, either during the day, in their dens (placing traps to catch them), or at night, killed by hunting (Tortora, 1982; Veitch, 2001), as well as the statistical confirmation of absence (Ramsey, et al., 2011).

CONCLUSION

The aerial hunting method proved to be an ideal technique for the eradication of sheep from Socorro Island. It enabled the eradication team to dispatch a high number of animals in a few days of work, while allowing the hunters to access difficult terrain. The use of Judas sheep and hunting dogs was crucial for completing the eradication.

Removing the exotic herbivorous species from the island is a conservation tool, which is evident in recovery of the natural environment. Habitat fragmentation and degradation caused by the presence of sheep was evident on the island, where the main impact was on vegetation. The resistance of native species has been important, not only in the relatively rapid recovery of the vegetation cover, but also in offering the possibility of recovering the former island vegetation. The results reflect the important role of vegetation in erosion control, both for establishing mechanical support due to plant roots in the soil structure and in capturing water flow and nutrients, providing fresh organic matter to the soil and restoring biogeochemical cycles and ecosystem processes.

With habitat recovery and progress in the feral cat eradication, wildlife recovery is expected as food availability and resources for the native species of the island gradually increase and predation decreases. Socorro Island seems resilient enough to recover over a relatively small-time scale, after the removal of the pressures caused by exotic mammals.

REFERENCES

Aguirre-Muñoz, A., Samaniego-Herrera, A., Luna-Mendoza, L., Ortiz-Alcaraz, A., Rodríguez-Malagón, M., Méndez-Sánchez, F., Félix-Lizárraga, M., Hernández-Montoya, J.C., González-Gómez, R., Torres-García, F., Barredo-Barberena, J.M. and Latofski-Robles, M. (2011). 'Island Restoration in Mexico: Ecological outcomes after systematic eradications of invasive mammals'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 250–258. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Algar, D., Angus, G.J., Brazell, R.I., Gilbert, C. and Withnell, G.B. (2010). 'Eradication of feral cats on Faure Island, Western Australia'. *Journal of the Royal Society of Western Australia* 93: 133–140.

Allington, G.R.H. and Valone, T.J. (2010). 'Reversal of desertification: The role of physical and chemical soil properties'. *Journal of Arid Environments* 74: 973–977.

Álvarez-Cárdenas, S., Castellanos-Vera, A., Galina-Tessaro, P. and Arnaud, G. (1994). 'Aspectos de la Población y el Hábitat del Borrego Doméstico (*Ovis aries*)'. In: A. Ortega-Rubio and V. Castellanos (eds.) *La Isla Socorro, Reserva de la Biosfera Archipiélago de Revillagigedo, México*, pp. 301–317. México: Centrode Investi-gaciones Biológicas del Noroeste.

Arnaud, G., Rodríguez, A., Ortega-Rubio, A. and Álvarez, C.S. (1993). 'Predation by cats on the unique endemic lizard of Socorro Island (*Urosaurus auriculatus*), Revillagigedo, México'. *The Ohio Journal of Science* 93(4): 101–104.

Arnaud, G., Rodríguez, A. and Álvarez, S. (1994). 'El Gato Doméstico (*Felis catus*), Implicaciones de su Presencia y Alternativas para su Eradicación'. In: A. Ortega-Rubio and A. Castellanos-Vera (eds.) *La Isla Socorro, Reserva de la BiosferaArchipiélago de Revillagigedo, México,* pp. 341–359. S. C. La Paz, Mexico: Centro de Investigaciones Biológicas del Noroeste.

Bester, M.N., Bloomer, J.P., Van Aarde, R.J., Erasmus, B.H., Van Rensburg, P.J., Skinner, J.D. Howell, P.G. and Naude, T.W. (2002). 'A review of the successful eradication of feral cats from sub-Antarctic Marion Island, Southern Indian Ocean'. *South African Journal of Wildlife Research* 32: 65–73.

Brattstrom, B. (2015). 'Food webs and feeding habits on the Revillagigedo Islands, Mexico'. *Pacific Science 69*: 181–195.

Brothers, N.P. (1982). 'Feral cat control on Tasman Island'. *Australian Ranger Bulletin* 2: 9.

Campa-Molina, M.A. (1989). 'Flora y Potencial Apícola de Isla Socorro, Archipiélago de las Revillagigedo'. MSc thesis. Jalisco, México: Universidad de Guadalajara.

Campbell, K. and Donlan, J. (2005). 'A review of feral goat eradication on islands'. *Conservation Biology* 198: 1362–1374.

Castellano, M.J. and Valone, T.J. (2007). 'Livestock, soil compaction and water infiltration rate: Evaluating a potential desertification recovery mechanism'. *Journal of Arid Environments* 71: 97–108.

Courchamp, F., Chapuis, J.L. and Pascal, M. (2003). 'Mammal invaders on islands: Impact, control, and control impact'. *Biological Reviews* 78: 347–383.

Donlan, C.J., Tershy, B.R., Campbell, K.J. and Cruz, F. (2003). 'Research for requiems: The need for more collaborative action in eradication of invasive species'. *Conservation Biology* 17: 1–2.

Dowding, J.E., Murphy, E.C., Springer, K., Peacock, A.J. and Krebs, C.J. (2009). 'Cats, rabbits, myxoma virus, and vegetation on Macquarie Island: A comment on Bergstrom et al. (2009)'. *Journal of Ecology* 46: 1129–1132.

Faulkner, K.R. and Kessler, C.C. (2011). 'Live capture and removal of feral sheep from Eastern Santa Cruz'. In: Veitch, C. R.; Clout, M. N. and Towns, D. R. (eds.). 2011. *Island invasives: eradication and management*, pp. 295–299. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Gallina, S. and C. López-González (eds.) (2011). *Manual de Técnicas para el Estudio de la Fauna. Volumen I*. Querétaro, México: Universidad Autónoma de Querétaro–Instituto de Ecología, Civil Association.

Hartmann, A., Assmus, B., Kirchhof, G. and Schloter, M. (1997). 'Direct Approaches for Studying Soil Microbes'. In: J.D. van Elsas, J.T. Trevors and E.M.H. Wellington (eds.) *Modern Soil Microbiology*, p. 172. New York: Marcel Dekker Inc.

León de la Luz, J.L., Breceda, A., Coria-Benet, R. and Cancino-Hernández, J. (1994). 'Asociaciones Vegetales'. In: A. Ortega-Rubio and V. Castellanos (eds.) *La Isla Socorro, Reserva de la Biosfera Archipiélago de Revillagigedo, México*, pp. 115–141. Mexico: Centro de Investigaciones Biológicas del Noroeste.

Levin, J.A. and Moran, R. (1989). *The Vascular Flora of Isla Socorro, México.* University of Michigan, USA: San Diego Society of Natural History.

Luna-Mendoza, L., Barredo-Barberena, J.M., Hernández-Montoya, J.C., Aguirre-Muñoz, A., Méndez-Sánchez, F., Ortiz-Alcaraz, A. and Félix-Lizárraga, M. (2011). 'Planning for the eradication of feral cats on Guadalupe Island, México: home range, diet, and bait acceptance'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 192–197. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Maya-Delgado, Y., Salinas-Zavala, F. and Troyo-Diéguez, E. (1994). 'Estado Actual del Suelo y Propuestas para su Conservación'. In: A. Ortega-Rubio and V. Castellanos (eds.) *La Isla Socorro, Reserva de la Biosfera Archipiélago de Revillagigedo, México*, pp. 63–75. Mexico: Centro de Investigaciones Biológicas del Noroeste.

Ortega-Rubio, A., Castellanos-Vera, A., Arnaud, G., Maya-Delgado, Y., Rodríguez, R., León de la Luz, J.L., Cancino, J., Jiménez, C., Llianas, J., Álvarez, S., Galina, P., Breceda, A., Troyo, E., Salina,s F., Díaz, S., Servín, R., Romero, H., Rodríguez, A. and Coria, R. (1992). 'Estudio de los recursos naturales de la Isla Socorro, Revillagigedo'. *Ciencia* 43: 175–184.

Ortiz-Alcaraz, A., Aguirre-Muñoz, A., Méndez-Sánchez, F. and Ortega-Rubio, A. (2016a). 'Feral sheep eradication at Socorro Island, Mexico: A mandatory step to ensure ecological restoration'. *Interciencia* 41: 184–189.

Ortiz-Alcaraz, A., Maya-Delgado, Y., Cortés-Calva, P., Aguirre-Muñoz, A., Rojas-Mayoral, E., Cordoba -Matson, M. and Ortega-Rubio, A. (2016b). 'Recovery of vegetation cover and soil after the removal of sheep in Socorro Island, Mexico'. *Forests* 7: 91.

Parkes, J., Fisher, P. and Robinson, S. (2012). *Eradication of Feral Cats on Large Mexican Islands: A Discussion of Options and Feasibility*. Lincoln, New Zealand: Invasive Species International, Landcare Research New Zealand.

Parkes, J., Fisher, P., Robinson, S. and Aguirre-Muñoz, A. (2014). 'Eradication of feral cats from large islands: An assessment of the effort required for success'. *New Zealand Journal of Ecology* 38: 307–314.

Phillips, R.B., Cooke, B.D., Campbell, K., Carrion, V., Marquez, C. and Snell, H.L. (2005). 'Eradicating feral cats to protect Galápagos land iguanas: Methods and strategies'. *Pacific Conservation Biology* 11: 57–66.

Ramsey, D.S.L., Parkes, J.P., Will, D., Hanson, C.C. and Campbell, K.J. (2011). 'Quantifying the success of feral cat eradication, San Nicolas Island, California'. *New Zealand Journal of Ecology* 35: 163–173.

Ratcliffe, N., Bell, M., Pelembe, T., Boyle, D., Benjamin, R., White, R., Godley, B., Stevenson, J. and Sanders, S. (2009). 'The eradication of feral cats from Ascension Island and its subsequent recolonization by seabirds'. *Oryx* 44: 20–29.

Rauzon, M.J. (1985). 'Feral cats on Jarvis Island: Their effects and their eradication'. *Atoll Research Bulletin* 282: 1–30.

Rauzon, M.J., Everett, W.T., Boyle, D., Bell, L. and Gilardi, J. (2008). 'Eradication of feral cats at Wake Atoll'. *Atoll Research Bulletin* 560: 1–21.

Rhea, M. (2000). 'Distribution of the Endangered Socorro Mockingbird, *Mimodes graysoni*, in Relation to Habitat Availability'. University Thesis. Villanova, Pennsylvania: Biology Department, Villanova University.

Richards, A.F. and Brattstrom, B.H. (1959). 'Bibliography, cartography, discovery and exploration of the Islas Revillagigedo'. *Proceedings of the California Academy Sciences* 29: 315–360.

Rodríguez, C., Torres, R. and Drummond, H. (2006). 'Eradicating introduced mammals from a forested tropical island'. *Biological Conservation* 130: 98–105.

Rodriguez-Estrella, R., Rivera, L. and Mata, E. (1994). 'Avifauna Terrestre'. In: A. Ortega-Rubio and A. Castellanos-Vera (eds.). *La Isla Socorro, Reserva de la Biosfera Archipiélago de Revillagigedo, México*, pp. 199–224. La Paz, Mexico: Centro de Investigaciones Biológicas del Noroeste S.C.

Sellés, G., Fereyra, R., Ruíz, R., Ferreyra, R. and Ahumada, R. (2012). *Compactación de Suelos y su Control: Estudio de Casos en el Valle de Aconcagua.* Boletín INIA 234. Santiago, Chile: Instituto de Investigaciones Agropecuarias.

SEMARNAT. (2004). *Programa de Conservación y Manejo, Reserva de la Biósfera Archipiélago de Revillagigedo*. México: Secretaría de Medio Ambiente y Recursos Naturales.

Taylor, D. and Katahira, L. (1988). 'Radiotelemetry as an aid in eradicating remnant feral goats'. *Wildlife Society Bulletin* 16: 297–299.

Tortora, D.F. (1982). 'Understanding electronic dog-training. 1'. *Canine Practice* 9: 17–22.

Twyford, K.L., Humphrey, P.G., Nunn, R.P. and Willoughby, L. (2000). 'Eradication of feral cats (*Felis catus*) from Gabo Island, south-east Victoria'. *Ecological Management and Restoration* 1: 42–49.

Van Vuren, D. (1992). 'Eradication of Feral Goats and Sheep from Island Ecosystems'. In: J.E. Borrecco and R.E. Marsh (eds.) *Proceedings of the 15th Vertebrate Pest Conference*, pp. 377–381. Davis, USA: University of California.

Veitch, C.R. (1989). *The Eradication of Cats and Sheep from Socorro Island: A Report to the Socorro Island Project*. Northern Region Technical Report Series No. 11. Auckland, New Zealand: Department of Conservation.

Veitch, C.R. (2001). 'The eradication of feral cats (*Felis catus*) from Little Barrier Island, New Zealand'. *New Zealand Journal of Zoology* 28: 1–12.

Villa, B. (1960). 'La Isla Socorro. Vertebrados terrestres'. *Monografías Instituto de Geofísica* 2: 203–206.

Walter, H.S. and Levin, G.A. (2008). 'Feral sheep on Socorro Island: Facilitators of alien plant colonization and ecosystem decay'. *Diversity and Distributions* 14: 422–431.

Will, D., Hanson, C.C., Campbell, K.J. Garcelon, D.K. and Keitt, B.S. (2010). 'A Trap Monitoring System to Enhance Efficiency of Feral Cat Eradication and Minimize Adverse Effects on Non-target Endemic Species on San Nicolas Island'. In: R. M. Timm and K. A. Fagerstone (eds.) *Proceedings of the 24th Vertebrate Pest Conference*, pp. 79–85. Davis, CA: University of California.

Wood, B., Tershy, B.R., Hermosillo, M.A., Donlan, C.J., Sánchez, J.A., Keitt, B.S., Croll, D.A., Howard, G.R. and Biavaschi. N. (2002). 'Removing cats from islands in North-west Mexico'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 374–380. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Removing introduced hedgehogs from the Uists

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Abstract Hedgehogs (*Erinaceus europaeus*) are native to Great Britain but were introduced to the island of South Uist in 1974 and gradually colonised South Uist and Benbecula. In 1999 hedgehogs were confirmed in southern areas of North Uist. Hedgehogs eat the eggs and occasionally the chicks of waders, which breed at high densities in the Uists. Initial research by Royal Society for the Protection of Birds (RSPB) in 1998 suggested that predation by hedgehogs was having a significant effect on the wader populations in South Uist. In 2014, remote cameras were used on a sample of wader nests and found hedgehogs responsible for 52% of all predation in South Uist. The Uist Wader Project was set up in 2000 to remove hedgehogs from North Uist initially, but with a long-term aim to remove hedgehogs completely from the Uists. Various methods including lamping, trapping and the use of sniffer dogs were developed, trialled, and improved. We developed an Index of Abundance (IOA) of hedgehogs, using footprint monitoring tunnels. This IOA provides a means of confirming the impact of removal activities on the hedgehog population. In anticipation of scaling up, we carried out a removal trial on a two km² area at Drimore in South Uist. The trial demonstrated the effort required to reduce the abundance of hedgehogs from high density, 30 animals/km², to zero and enabled the project team to estimate the resources required to eradicate hedgehogs from Benbecula, North and South Uist. The North Uist phase should be complete by the beginning of 2018, with only eight hedgehogs caught in 2016 and just one in 2017. Two years of monitoring are planned between 2018 and 2020, to confirm eradication.

Keywords: dunlin, IOA, redshank, ringed plover, sniffer-dogs, translocation, trap, wader

INTRODUCTION

Wader surveys in the early 1980s showed that the Uists, off the west coast of Scotland, held high densities of breeding redshank (*Tringa totanus*), ringed plover (*Charadrius hiaticula*) and dunlin (*Calidris alpina*) (Fuller, et al., 1986). In recognition of the importance of the Uists, 14 Sites of Special Scientific Interest (SSSIs) and two Special Protection Areas (SPAs) for Birds were designated in the late 1990s. Shortly afterwards a decline was found in wader populations on the islands of South Uist and Benbecula that was largely due to egg predation by hedgehogs (Jackson, 2001; Jackson & Green, 2000; Jackson, et al., 2004). Hedgehogs (*Erinaceus europaeus*) are native to Great Britain but were introduced to South Uist in 1974–75 (Angus, 1993). In 1999, hedgehogs were starting to colonise southern areas of North Uist (Jackson & Green, 2000; Jackson, et al., 2004). Declines of waders recorded in South Uist between 1983 and 1998 were: ringed plover by -58%; dunlin by -65%; and redshank by -43% (Fuller & Jackson, 1999). In 2014, remote cameras were used on a sample of wader nests and found hedgehogs responsible for 52% of all predation in South Uist.

The hedgehog population on the United Kingdom mainland has been in decline since the1960s (Noble, et al., 2012). Hedgehogs are protected under Schedule 6 of the Wildlife and Countryside Act 1981 throughout the UK, but are classified as invasive non-natives in the Uists, as they are classified outwith their native range under section 14. Hedgehogs have no natural predators in the Uists, can breed five months out of the year, and can produce at least as many young per year as their population, as measured in the spring (Jackson, 2007). Initial research in South Uist on hedgehog behaviour and methods of locating them was carried out between 1997 and 2001 (Jackson, 2007). This work estimated the density of hedgehogs in different habitats in South Uist at 31.8 animals/km² for machair, 15.4 animals/km² for blackland and two animals/km² for moorland.

The Uist Wader Project was launched in 2000 as a partnership of Scottish Natural Heritage (SNH), Royal Society for the Protection of Birds (RSPB) and the Scottish Executive. The Project's objective was to safeguard the waders of the Uists from introduced hedgehogs. In order to achieve this it would be necessary to remove all the hedgehogs from the Uists, starting in North Uist.

MATERIALS AND METHODS

Study area

The Uists are part of the Outer Hebrides, located off the north-west coast of Scotland (Fig. 1). The Uists include six inhabited, low-lying islands, connected by causeways. The three main islands are North Uist (333 km²), Benbecula (81 km²) and South Uist (315 km²). The climate is wet and windy. Wind-blown shell sand has formed extensive machair habitats on the west side of these islands. These lime-rich coastal grasslands are grazed by livestock and cultivated with arable crops (oats, rye, barley and potatoes) on a traditional rotation (Angus, 2006). There are a few farms but most of the agricultural land is divided into small tenanted units, known as crofts, each with shares in larger common grazings. The other predominant habitat types in the Uists are moorland and blackland (an intermediate zone of mesotrophic grassland between machair and moorland).

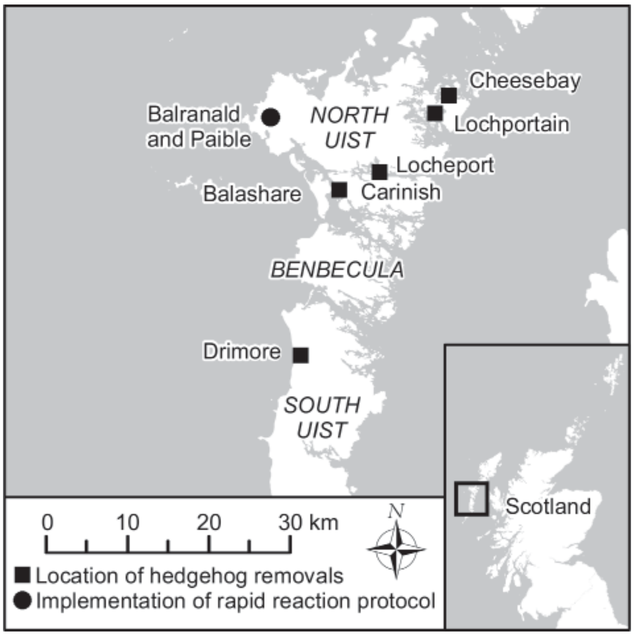

Fig. 1 Location map of the Uists in conjunction with mainland Scotland.

In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) (2019). Island invasives: scaling up to meet the challenge, pp. 274–281. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

In order to carry out any type of fieldwork in the Uists it was essential to have the full support of the local community, land owners, crofters and residents. Project staff spent time working with these groups to secure access to land and receive information relating to hedgehogs. Although access permission was always granted, there were constraints placed on some of the removal methods described below.

Population model

Based on initial research by Jackson (2007), modellers at Newcastle University developed a hedgehog population model in two phases (Shirley, et al., pers. comm. 2007; Shirley, et al., 2010). These individual-based, simulation models of the hedgehog population indicated that trapping and lamping would achieve total eradication of hedgehogs from the Uists within 30 years, at best, which represents eradication by 2040. The hedgehog population in the Uists was estimated from a combination of field data and the model to be around 3,900 in 2010, whereas this was estimated to be about 3000 in 2007, (95% confidence limits ±800). This highlighted the shortcomings of our initial methods and led to a new approach, using sniffer dogs and all year round removal of hedgehogs.

Hedgehog removal

There are three key methods of removing hedgehogs; lamping (spot-lighting), live cage trapping and searching with sniffer dogs. Lamping at night was very effective on short cropped machair turf. It involves three to five people transecting areas of land operating in a straight line about five to 10 metres apart, each using a 10–50 watt, 12 volt halogen spot-lamp to survey the ground for hedgehogs. Lamping was not effective in longer vegetation and the night work caused disturbance to local residents. Lamping as a method of hedgehog removal was gradually phased out due to its intrusive nature with regard to light and noise disturbance at night.

Live cage trapping was tried in 2004 and proved very effective at removing a large proportion of the population: 80–90% of hedgehogs over an eight week period. Trapping worked well in all types of habitat and replaced lamping as the main method of removing hedgehogs. The live cage traps used for hedgehogs are 180 × 150 × 480 mm, with a spring-loaded door, activated by a treadle plate. Traps are installed in large trap grids, designed to intersect the home range of each potential hedgehog. Two different trapping densities are used. Low density trapping (30 traps/km^2) is used when initially establishing a trapping route, where the underlying hedgehog population is expected to be zero (i.e. monitoring suggests no hedgehogs) or when the underlying habitat is not particularly suitable for hedgehogs, such as moorland and bog. Higher density trapping (50 traps/km^2) is used where a known hedgehog population exists and the underlying habitat is suitable. The traps are baited with fish, which is placed behind the treadle plate, but not obstructing it. Once operational, traps are checked every day. Throughout the project, trap placement was continuously improved through experience and research with habitat, location, cover, bait and trap sensitivity being the most important factors. Trapping proved to be an effective means of capturing a large proportion of a population, but not every animal, suggesting that some were trap shy.

Sniffer dogs are also used to remove hedgehogs. The dogs are trained to indicate the location of a hedgehog without harming it and are rewarded with a short period of play time with a favourite toy when successful. A specialist trainer was brought in for six days each year to guide the training process, encourage best practice and work with each dog handler on a one to one basis. Sniffer dogs can work effectively for periods of three to four hours and an experienced sniffer dog and handler can cover up to two km^2 per week in most weather conditions and across diverse vegetation. Dense vegetation and calm conditions result in narrower, more condensed search transects, while wind speeds between eight and 55 kph and short vegetation allow wider more expansive transects and hence greater area covered per unit of time. Wind speeds in excess of 55 kph progressively reduce the efficiency of dog searching due to the scent being dispersed too widely. Sniffer dogs and trapping complement each other as hedgehog removal methods, because dogs are more effective in boggy ground where traps simply can't be set and traps are more effective in areas where dens are located deep underground and hedgehogs only re-emerge at night. There were sometimes more restrictions on using dogs than traps in fields at lambing time but sniffer dogs could locate hedgehogs during the winter, when trapping is ineffective. The use of dogs was suspended, early on in the project, following the introduction of legislation banning the hunting of wild mammals with dogs, The Protection of Wild Mammals (Scotland) Act 2002. This greatly reduced the efficiency of removing hedgehogs at lower densities and added additional time and cost to the Project. Following careful legal interpretation of how dogs could be used to locate and 'flush' hedgehogs, the use of sniffer dogs was reinstated in 2010.

Between 2003 and 2006 all captured hedgehogs were euthanised, based on the best information available at that time. Advice from the animal welfare organisation, the Scottish Society for the Prevention of Cruelty to Animals (SSPCA), rejected translocation on welfare grounds and advocated hedgehogs were euthanised. The SNH board in 2002 stated that there was no scientific evidence or overriding conservation imperative to justify translocation of hedgehogs from the Uists to the mainland. During this time, the Project came under increasing pressure from animal rights groups and special interest conservation groups to stop killing hedgehogs and consider moving them to the Scottish mainland instead. The British public perceives hedgehogs as an iconic species, which is the gardener's friend, and there was strong media and public pressure against the cull.

New research carried out at Bristol University (Molony, et al., 2006) showed that translocation of hedgehogs resulted in low mortality if certain levels of veterinary care, feeding and general welfare were provided. Based on this work, the SSPCA advised that the hedgehogs' welfare would not be adversely affected by being translocated to the Scottish mainland. SNH then entered into a partnership with the animal care sector to translocate hedgehogs. Fieldworkers pass hedgehogs onto a 'carer', based in South Uist, for onward transport to an animal rescue centre on the mainland for release under established protocols. In response to improvements in the ability to identify and care for pregnant females and to locate dependant young, it became possible in 2012 to remove hedgehogs throughout the season, rather than only during the non-breeding season of three and a half months as done previously.

Monitoring

Monitoring between 2009 and 2010 simply involved checking traps and lamping, which equates to extending the removal methods until a period of two years has elapsed where no capture of hedgehog has occurred.

From 2011 onwards, three monitoring techniques were deployed: footprint monitoring tunnels, sniffer dogs and motion-activated cameras.

The footprint monitoring tunnels were made out of 150 mm plastic drainage pipe, cut to 560 mm lengths. A rectangular section 100 × 190 mm was cut out of the middle of the pipe to accommodate a plastic tray, 110 × 50 × 200 mm. The tray was then filled with one of three different substrates; clay, sand or carbon plate. The tunnels were dug into the ground and covered with turf to make the tunnel as much like a natural burrow as possible. The inside of the tunnel was fashioned to allow a natural walk through for an animal over the tray. These tunnels were dug into the monitoring area at a density of five tunnels/km² and their positions recorded using GPS.

Trained sniffer dogs were deployed to search at least 25% of the monitoring area following methods similar to their use for hedgehog removal. Hedgehogs located in North Uist were removed as re-release was not an option, whereas hedgehogs located in South Uist were released, since their removal would have no real impact on the overall population, which had reached its maximum carrying capacity.

Motion-activated cameras (model: Bushnell Trophycam HD max) were deployed at a density of 1.25 cameras/km². We set them to record 60 second video clips (1280 × 720 px) onto a 32 GB SD card. The camera was focussed on a 120 g 'tuna tin' with perforations in its top, filled with fish and dug into the ground so the surface of the can was level with the ground. This acts as an attractant to hedgehogs and a host of other animals, yet prevents them from removing the fish. The SD card needs to be changed every two weeks and the rechargeable batteries have a variable lifespan, of two to three weeks, depending on the rate of triggering.

Six sample areas representing the whole of North Uist were monitored between 2013 and 2014 using at least two different monitoring methods. Monitoring highlighted the areas where hedgehogs were present and allowed a more strategic and selective approach to checking the total area of North Uist.

Occupancy model

In the early part of the Project, progress was measured as 'number of hedgehogs caught per 1000 trap nights'. When trapping effort was applied over time, this measure generally showed a decline. However, we were unsure if this measure reflected the actual impact of removal activities on the hedgehog population, or if a significant number of animals remained undetected due to trap avoidance. In 2013, a two-year monitoring trial was established to estimate occupancy and the relative index of abundance (IOA) of hedgehogs across the Uists, and evaluate the effectiveness of the removal methods.

Between 2013 and 2014, hedgehog populations were assessed in 19 locations in the key areas for breeding waders, using footprint monitoring tunnels. Attempts were also made to assess populations using motion-activated cameras and sniffer dogs, but insufficient cameras were available and the sniffer dog data proved too difficult to interpret due to a number of factors including experience of dog and handler, wind speed, and topology of land.

Each plot (route) covered an area of four km² with a minimum of five monitoring tunnels/km² and was checked twice per week.

Various occupancy models were tested, and the Royle-Nichols single season, abundance-induced heterogeneity model (Royle & Nichols, 2003) was chosen as the most appropriate single season occupancy model. This is a two-parameter model that derives occupancy (ψ) from estimates of detectability r (the probability of detection per tunnel) and population density λ (the mean of the Poisson distribution), thus estimating occupancy in a way that accounts for hedgehogs being easier to detect when there are more of them. The following formulas represent the Royle-Nichols model:

$$1)\ L(w) = \prod_{i=1}^{R} \left\{ \sum_{k=0}^{K} \binom{T}{w_i} p_k^{w_i} (1-p_k)^{T-w_i} f_k \right\}$$

$$2)\ p_k = 1 - (1-r)^k$$

$$3)\ f_k = \frac{e^{-\lambda}\lambda^k}{k!}$$

Formula (1) represents the likelihood of detections, where W represents detections, R represents sites, T represents (route) locations. Formula (2) represents the site detection probability and (3) represents the probability density formula for a Poisson distribution, where both (2) and (3) substitute into (1). Note also how r and λ are incorporated into this model.

The plots were grouped together by year and modelled with a constrained detectability and unconstrained population density. Detectability was estimated, along with individual population density, for each location and year.

Footprint monitoring results were used in preference to camera monitoring results due to the limited data sample from the cameras compared to tunnels (Paul Ross, pers. comm. 2014).

Hedgehog removal trial

In 2014, we undertook a hedgehog removal trial to evaluate the effectiveness of the hedgehog removal methods. A research area of 1.78 km² was selected on Drimore farm in South Uist, which represented typical machair habitat with a probable high population of hedgehogs. A perimeter area of 1.3 km² surrounding this research area was also created to reduce the effects of dispersion and migration of hedgehogs following removal from the research area.

The research area was monitored using footprint monitoring tunnels, motion-activated cameras and sniffer dogs for a four week period to establish an IOA. The monitoring tunnels were evenly distributed at a density of five tunnels/km², motion-activated cameras at a density of five cameras/km² and sniffer dogs were operated at a rate of two km² per week. Hedgehogs were then removed from both areas using 50 traps/km² on the research area only, and sniffer dogs on both areas, for an eight week period. The research area was monitored for a further four weeks in the same design as the pre-removal monitoring, to establish whether all hedgehogs had been removed.

Scaling up to North Uist

Recent hedgehog removal efforts in North Uist were guided by the results of the monitoring work; areas that showed presence of hedgehogs were searched using a combination of trapping and sniffer dogs. The search effort was set using the results of the Drimore trial.

The removal phase is expected to be completed by spring 2018, and will be followed by a further two years of monitoring to confirm absence of hedgehogs. If a hedgehog is encountered during the monitoring phase a rapid-reaction protocol will be initiated.

Rapid-reaction protocol

A one km radius buffer around the sighting of a hedgehog will be searched for four weeks with sniffer dogs and 50 traps/km². If further hedgehogs are found, this process will be repeated.

RESULTS

Activites implemented in 2003 to 2008

Sniffer dogs were used only in 2003 during this period. Lamping and trapping were used as the main methods of hedgehog removal. Monitoring between 2009 and 2010 confirmed successful eradication.

Initial hedgehog removal: 2003–2008

Hedgehog removal started in 2003 in Locheport and Carinish in the southern area of North Uist (129 km²) and was completed by 2008. A further two years of monitoring were carried out to verify a successful eradication, which was declared in 2010. Fig. 2 shows the removal of hedgehogs and effort applied in Carinish and Locheport. Believing that North Uist was clear, the Project expanded the removal methods into Benbecula to continue working southwards. Good progress was made initially, but further hedgehogs were reported from new areas of North Uist; from Balranald in 2009 and Lochportain in 2012. Work in Benbecula was postponed whilst the trapping team was re-deployed to eradicate hedgehogs from these new areas.

Monitoring results – occupancy estimates 2013–2014

As expected, the lowest occupancy (ψ) estimates were in North Uist and the highest ones in South Uist.

The North Uist IOA monitoring results for 2013 and 2014 are shown in Table 1 and Table 2, respectively. Note that the route names do not correspond to the same areas between the two years. In 2013, Baleshare (represented by H1 & H2) showed no occupancy of hedgehogs, and Balranald (F1) showed a low level of occupancy. In 2014, Balranald (F1, M1, andG2) showed further dispersal of hedgehogs.

The South Uist and Benbecula occupancy results for 2013 and 2014 are shown in Table 3 and Table 4, respectively. Note that the routes K1 and B2 correspond between Tables 3 and 4. Fig. 3 shows these results spatially. The occupancy estimate of hedgehogs is relatively high for almost all parts sampled in South Uist and Benbecula.

All other parts of North Uist, Benbecula and South Uist were monitored by sniffer dog but due to a range of confounding factors it proved impossible to convert these data into a meaningful occupancy estimate. However, the sniffer dog monitoring did give a good overview of the distribution of hedgehogs across the Uists to complement the formal occupancy estimate results.

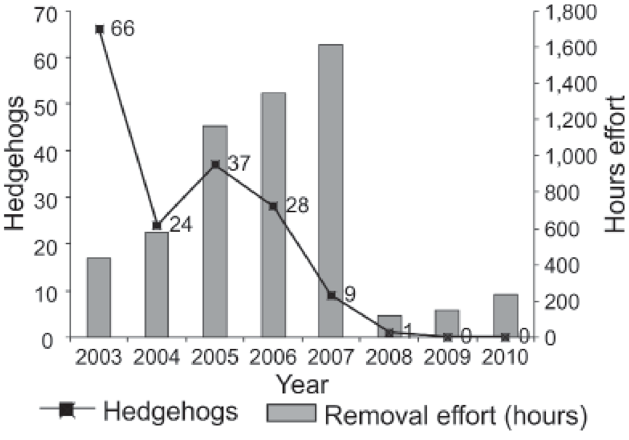

Fig. 2 Carinish and Locheport hedgehog removal and effort.

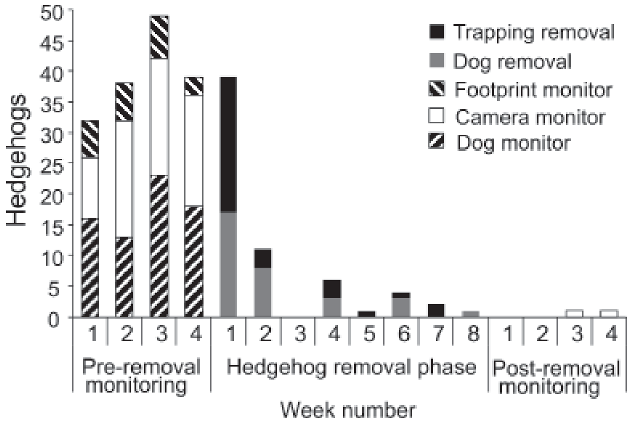

Fig. 3 Hedgehogs removed or detected, by method, for the Drimore trial.

Removal trial results (Drimore, 2014)

Pre-removal monitoring phase

Monitoring was carried out across the research area for four weeks between 7 April and 4 May. Table 5 shows the numbers of hedgehogs detected each week by footprint monitoring, camera monitoring and sniffer dogs. Using only the footprint monitoring data, it was possible to derive an occupancy estimate for this area of land during the four week monitoring phase, which is shown in row B2 in Table 3. In comparison to other sites monitored in the Uists, the Drimore site represented a high population of hedgehogs.

Removal of hedgehogs

This phase of operation involved removing hedgehogs from the research and perimeter areas using live cage traps and sniffer dogs over an eight week period between 5 May and 29 June. Two fieldworkers searched the area using sniffer dogs and operated 89 live cage traps. Table 6 shows that the same numbers of hedgehogs were removed from the research area by sniffer dogs as by trapping. Over the same period, hedgehogs were removed from the perimeter area by sniffer dogs alone, as shown in Table 7.

Post-removal monitoring

The final phase of the trial involved repeating the monitoring over another four week period between 30 June and 28 July to measure the IOA of the hedgehogs after the removal operation. Table 8 shows that only two hedgehogs showed up on camera during this period, both in the perimeter area. No hedgehogs were detected within the research area, providing an acceptable level of confidence that all of the hedgehogs had been removed. Fig. 3 summarises the numbers of hedgehogs detected by each monitoring method at all three stages.

Recent hedgehog removals in North Uist – scaling up to North Uist

Lochportain hedgehog removal

Lochportain, along with the neighbouring townships, is located on a peninsula on the east side of North Uist. In 2012 a hedgehog was found by a member of the public on the road close to Cheesebay (adjacent to Lochportain). Due to other commitments and limitations on staff resources the Project was only able to respond to this potential hedgehog population with a very limited removal effort in 2013, which yielded no hedgehog captures. The team returned to this area in 2015, with a concerted removal effort covering some 75 km^2, followed by monitoring in 2016. Fig. 4 shows the hedgehog removal and relative effort appliedin Lochportain.

Table 1 2013 – Royle-Nichols parameter estimates for hedgehogs in North Uist. Note: ψ represents the probability of occupancy, derived from (1 – f_k), r represents the probability of detection per hedgehog / tunnel, and l represents population density as the mean of the Poisson distribution. Route name refers to four km² plot areas.

Route name	Naïve occupancy	Occupancy		Detectability		Population density	
		ψ	SE	r	SE	λ	SE
F1	0.046	0.073	0.322	0.142	0.027	0.076	0.077
F2	0.000	0.000	-	0.142	0.027	0.000	-
G1	0.000	0.000	-	0.142	0.027	0.000	-
G2	0.000	0.000	-	0.142	0.027	0.000	-
H1	0.000	0.000	-	0.142	0.027	0.000	-
H2	0.000	0.000	-	0.142	0.027	0.000	-

Table 2 2014 – Royle-Nichols parameter estimates for hedgehogs in North Uist.

Route name	Naïve occupancy	Occupancy		Detectability		Population density	
		ψ	SE	r	SE	λ	SE
F1	0.100	0.154	0.294	0.142	0.027	0.167	0.086
M1	0.025	0.041	0.333	0.142	0.027	0.042	0.042
G2	0.075	0.124	0.304	0.142	0.027	0.133	0.079
J1	0.000	0.000	-	0.142	0.027	0.000	-

Table 3 2013 – Royle-Nichols parameter estimates for hedgehogs in South Uist and Benbecula.

Route name	Naïve occupancy	Occupancy		Detectability		Population density	
		ψ	SE	r	SE	λ	SE
A1	0.286	0.582	0.145	0.142	0.027	0.873	0.347
A2	0.400	0.531	0.163	0.142	0.027	0.758	0.288
B1	0.130	0.241	0.264	0.142	0.027	0.275	0.164
B2	0.400	0.632	0.128	0.142	0.027	1.000	0.366
C1	0.286	0.405	0.206	0.142	0.027	0.519	0.223
C2	0.100	0.182	0.284	0.142	0.027	0.201	0.145
D1	0.000	0.000	-	0.142	0.027	0.000	-
D2	0.250	0.458	0.188	0.142	0.027	0.613	0.273
E1	0.300	0.446	0.192	0.142	0.027	0.591	0.255
E2	0.100	0.171	0.288	0.142	0.027	0.188	0.135
K1	0.050	0.080	0.319	0.142	0.027	0.084	0.085
B2	0.400	0.628	0.129	0.142	0.027	0.988	0.332

Table 4 2014 – Royle-Nichols parameter estimates for hedgehogs at Drimore in South Uist and Benbecula.

Route name	Naïve occupancy	Occupancy		Detectability		Population density	
		ψ	SE	r	SE	λ	SE
K1	0.050	0.105	0.311	0.142	0.027	0.110	0.111
B2	0.000	0.000	-	0.142	0.027	0.000	-

Lochportain was effectively cleared of hedgehogs over a 10 week period, which matched very closely with the Drimore removal trial. Migration to and from Lochportain was minimised by being located on a peninsula with a narrow isthmus.

Fig. 5 compares the Drimore trial and Lochportain removal. There is a strong similarity in the pattern of hedgehog removal even though the starting populations of hedgehogs and the area of land covered are very different. Both locations represent declining sequences of weekly captures ending at one or less over eight to 10 weeks. Subsequent monitoring on both sites demonstrated that no further hedgehogs were immediately present.

Balranald & Paible hedgehog removal

In 2009 hedgehogs were sighted in Balranald and Paible in the west of North Uist by members of the public. Trapping began in 2009, and sniffer dogs were introduced gradually from 2010, so that by 2013 all fieldwork staff operated a dog.

Fig. 6 shows that the bulk of the hedgehog population was removed between 2013 and 2015, with just a small number of hedgehogs removed in 2016. It also shows the relationship between trapping effort and the number of hedgehogs removed for Balranald and Paible.

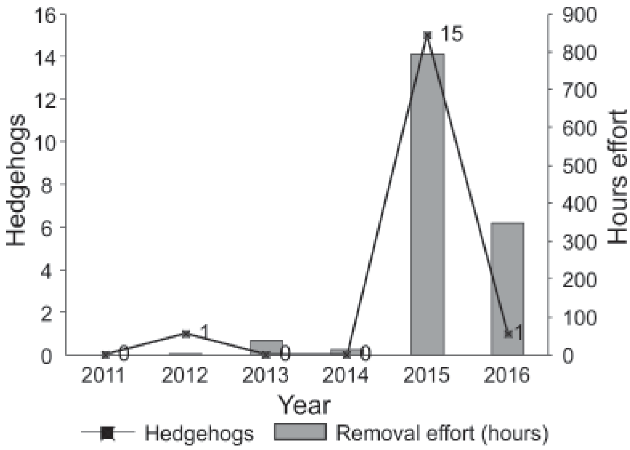

Fig. 4 Lochportain hedgehog removal and effort.

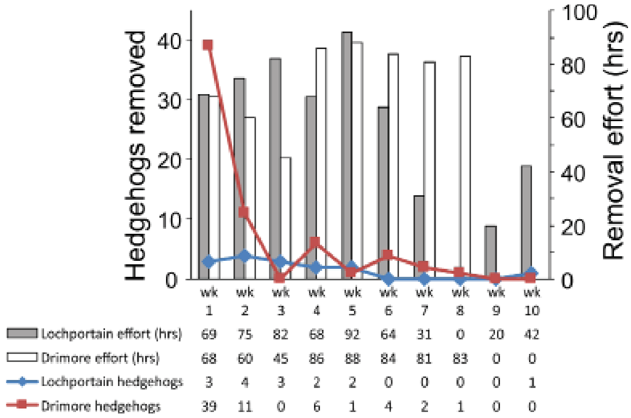

Fig. 5 Comparison of hedgehogs removed between Lochportain and Drimore.

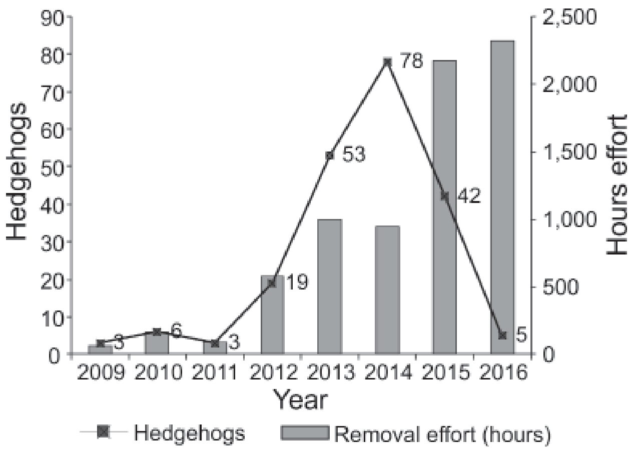

Fig. 6 Balranald and Paible removal and effort.

Table 5 Hedgehogs detected during the pre-removal monitoring phase at Drimore. Monitoring effort: two footprint monitoring checks per week per tunnel over 10 tunnels, five cameras running continuously and two sniffer dogs checking two km² per week.

Monitoring method	Week				Total
	1	2	3	4	
Footprint	6	6	7	3	22
Camera	10	19	19	18	66
Sniffer dog	16	13	23	18	70

Implementation of rapid-reaction protocol

The rapid-reaction protocol has been used only once. One hedgehog was located by a monitoring camera and then located and removed by a dog handler and sniffer dog in the area of east Balranald during April 2017. A search zone was established using a buffer of a radius of 1 km from the location of the hedgehog, as shown in Fig. 7. Four weeks searching using sniffer dogs and trap checks were carried out, but no further hedgehogs were located.

DISCUSSION

It is essential to have the support of the local community, not just to report sightings but also to persuade people not to move hedgehogs to new areas. Hedgehogs were clearly moved to discrete unconnected areas in North Uist, including Carinish, Locheport, Balranald and Lochportain. We had support from most land managers but we failed to reach all individuals within the wider community. Some people moved hedgehogs as they thought they would provide a helpful service such as controlling garden slugs or snails that host sheep fluke. Once we were able to discuss these introductions and the potential impacts with the individuals involved, they usually became more supportive. Any future removal project should include an education and promotion resource to assist with community engagement. There is also a need to secure full support and commitment right from the start of the project all the way through until eradication is confirmed.

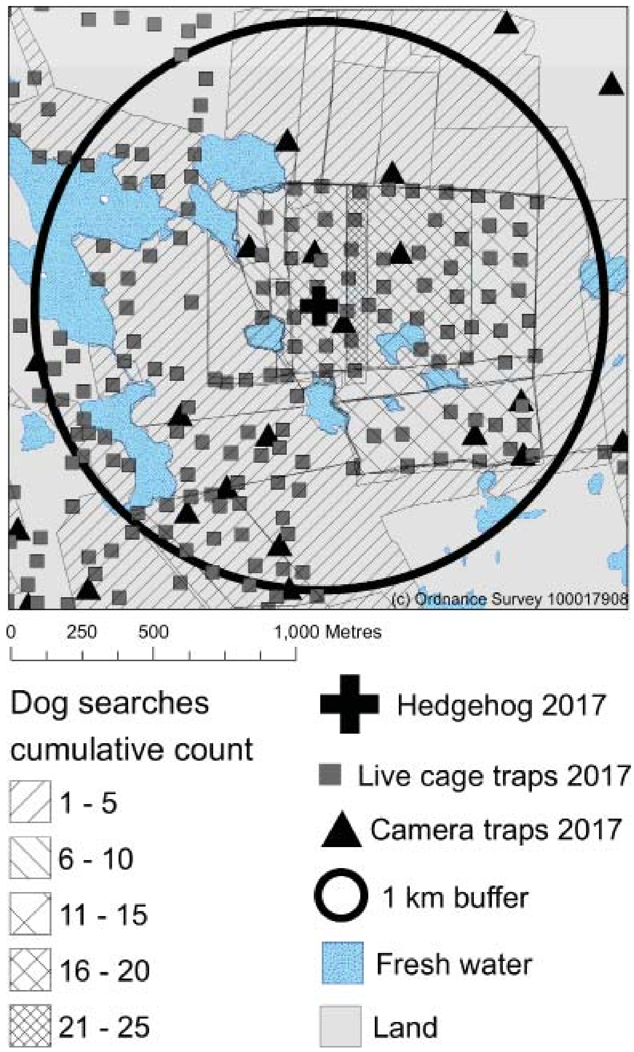

Fig. 7 Rapid-reaction protocol in response to hedgehog capture at Balranald.

Table 6 Hedgehogs removed from research area at Drimore during removal phase.

	Week								
Removal method	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**Total**
Trapping	22	3	0	3	1	1	2	0	32
Sniffer dog	17	8	0	3	0	3	0	1	32
Total hedgehogs	39	11	0	6	1	4	2	1	64
Effort: trapping (hrs)	44	43	44	44	41	44	44	44	348
Effort: dog (hrs)	24	17	1	42	47	40	37	39	247

Table 7 Hedgehogs removed from perimeter area at Drimore during removal phase.

	Week								
Removal method	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**Total**
Trapping	-	-	-	-	-	-	-	-	-
Sniffer dog	0	2	12	4	1	1	1	0	21
Total hedgehogs	0	2	12	4	1	1	1	0	21
Effort: trapping (hrs)	-	-	-	-	-	-	-	-	-
Effort: dog (hrs)	0	2	14	2	2	6	3	3	32

The methods used in eradicating hedgehogs from Carinish and Locheport were limited by the absence of sniffer dogs and a lack of clarity on the abundance of hedgehogs in any given area. For animal welfare reasons, hedgehog removal was restricted to the three and a half month non-reproductive period. These limitations meant it took approximately eight years to clear the area and verify it as clear. Balranald and Paible were also initially limited to the non-reproductive season and sporadic, exploratory efforts prior to 2013. However from 2013 onwards Balranald and Paible had a fully operational team of sniffer dogs and hedgehog removal progressed relatively quickly, with captures tailing off by 2016. Lochportain also benefitted from the use of dogs and from being on a peninsula. The introduction of monitoring, refined control methods and strategies meant that removing the Lochportain hedgehog population took just two years, compared to eight at Carinish. If there are obstacles or barriers to removal activities then it will reduce the effectiveness of removal and it will take longer to reduce the population to zero. Being able to work all year round made the Project much more efficient, reducing the predicted minimum time required for eradication of hedgehogs from the Uists from 30 to five years.

The Drimore trial demonstrated that hedgehog population density within a discrete area can be effectively reduced to zero by trapping and sniffer dogs over a relatively short period of time. The removal phase reduced the IOA from a high level to zero. The two hedgehogs detected on camera in the latter weeks of the post-removal monitoring were located in the perimeter area and it is assumed these were migrating into the research area. Comparing the Drimore trial results to the Lochportain eradication shows that it took roughly the same effort to remove 64 hedgehogs as it did 14 hedgehogs from an equivalent area. This suggests that eradication effort is determined by area of suitable habitat more than hedgehog density.

Table 8 Hedgehogs monitored during the post-removal monitoring phase at Drimore.

Monitoring method	Week				
	1	2	3	4	Total
Footprint	0	0	0	0	0
Camera	0	0	1	1	2
Sniffer dog	0	0	0	0	0

The Project needed to estimate the effort required to reduce the hedgehog population to zero over a given area of land and prevent re-colonisation from surrounding areas. The Drimore removal trial enabled us to assess whether the resource had been sufficient on every bit of land at Balranald and where to put in additional resource.

The near complete removal of hedgehogs from North Uist was achieved using an agreed strategy with proven methods of removal, which were shown to be effective. Being able to measure the effectiveness of the hedgehog removal methods used, and the effort required to clear a given area of land, enables a fairly accurate estimation of what timescale would be required to clear a specific area of land. There also needs to be a method of confirming that the population has been reduced to zero (Russell, et al., 2016). The IOA has been extremely valuable in that respect, particularly on areas such as Balranald, with complicated land tenure and constraints on using dogs whilst livestock are in fields at certain times of year.

In the early days of the Project we coloured maps by hand and filled in paper data sheets, whereas now we use graphic GPS, integrated to GIS systems, connected to relational databases. This increased data flow has facilitated a more adaptive approach to managing project activity. Scientific advice from a wide range of sources has been extremely helpful but needs to be combined with practical considerations.

Ideally it would have been desirable to have cleared the hedgehogs from South Uist to allow the waders to recover faster, but clearing North Uist first and then moving south made more strategic sense. Having successfully removed all of the hedgehogs from North Uist, the next step is to continue southwards and remove hedgehogs from Benbecula and South Uist. This will require clearing an area of almost 400 km². Using the results from the Drimore trial and the current removal methods, we estimated that this will take between five and 10 years and will require a team of 18 staff. It is estimated that this will cost between £3.5 and £5.0 million and, at the time of writing, SNH is exploring funding options with partners.

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REFERENCES

Angus, S. (1993). 'Hedgehogs in the Western Isles in 1991'. *Hebridean Naturalist* 11: 73–76.

Angus, S. (2006). 'De tha machair? Towards a machair definition'. Sand dune machair 4, pp. 7–22. Aberdeen Institute for Coastal Science & Management, Aberdeen.

Fuller, R.J., Reed, T.M., Buxton, A., Williams, T.D. and Pienkowski, M.W. (1986). 'Populations of breeding waders (Charadrii) and their habitats on the crofting lands of the Outer Hebrides, Scotland'. *Biological Conservation* 37: 333–361.

Fuller, R.J. and Jackson, D.B. (1999). 'Changes in populations of breeding waders on the machair of North Uist, Scotland, 1983–1998'. *Wader Study Group Bulletin* 90: 47–55.

Jackson, D.B. and Green, R.E. (2000). 'The importance of the introduced hedgehog (*Erinaceus europaeus)* as a predator of the eggs of waders (Charadrii) on the machair in South Uist, Scotland'. *Biological Conservation* 93: 333–348.

Jackson, D.B. (2001). 'Experimental removal of introduced hedgehogs improves wader nest success'. *Journal of Applied Ecology* 38: 802–812.

Jackson, D.B., Fuller, R.J. and Campbell, S.T. (2004). 'Long-term population changes among breeding shorebirds in the Outer Hebrides, Scotland, in relation to introduced hedgehogs'. *Biological Conservation* 117: 151–166.

Jackson, D.B. (2007). 'Factors affecting the abundance of introduced hedgehogs (*Erinaceus europaeus*) on the Hebridean island of South Uist in the absence of natural predators, and the implications for ground-nesting birds'. *Journal of Zoology* 271: 210–217.

Molony, S.E., Dowding, C.V., Baker, P.J., Cuthill, I.C. and Harris, S. (2006). 'The effect of translocation and temporary captivity on wildlife rehabilitation success: an experimental study using European hedgehogs (*Erinaceus europaeus*)'. *Biological Conservation* 130: 530–537.

Noble, D.G., Aebischer, N.J., Newson, S.E., Ewald, J.A. and Dadam, D. (2012). *A Comparison of Trends and Geographical Variation in Mammal Abundance in the Breeding Bird Survey and the National Gamebag Census*. JNCC Report No. 468.

Royle, J.A. and Nichols, J.D. (2003). 'Estimating abundance from repeated presence-absence data or point counts'. *Ecology* 84: 777–790.

Russell, J.C., Binnie, H.R., Oh, J., Anderson, D.P., Samaniego-Herrera, A. (2016) Optimizing confirmation of invasive species eradication with rapid eradication assessment. *Applied Ecology* 54(1): 160–169.

Shirley, M., Lurz, P. and Rushton, S. (2010). *Development of a Population Model for the Management of Hedgehogs on the Uists and Benbecula*. Newcastle University, SNH Commissioned Report: 372

Five eradications, three species, three islands: overview, insights and recommendations from invasive bird eradications in the Seychelles

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Abstract Management and eradication techniques for invasive alien birds remain in their infancy compared to invasive mammal control methods, and there are still relatively few examples of successful avian eradications. Since 2011, five separate eradication programmes for invasive birds have been conducted on three islands by the Seychelles Islands Foundation (SIF). Target species were prioritised according to their threat level to the native biodiversity of the UNESCO World Heritage Sites of the Seychelles, Aldabra Atoll and Vallée de Mai, which SIF is responsible for managing and protecting. Red-whiskered bulbuls (*Pycnonotus jocosus*) and Madagascar fodies (*Foudia madagascariensis*) occurred on Assumption, the closest island to Aldabra, which, at the time, had no known introduced bird species. The growing population of ring-necked parakeets (*Psittacula krameri*) on Mahé posed a threat to endemic Seychelles black parrots (*Coracopsis barklyi*) on Praslin where the Vallée de Mai forms their core breeding habitat. In 2012, red-whiskered bulbuls and Madagascar fodies were detected on Aldabra, so an additional eradication was started. All eradications used a combination of mist-netting and shooting. The intensive part of each eradication lasted three years or less. On Assumption, 5,279 red-whiskered bulbuls and 3,291 Madagascar fodies were culled; on Mahé, 545 parakeets were culled; and on Aldabra 262 Madagascar fodies and one red-whiskered bulbul were culled. Each programme underwent 1–2 years of follow-up monitoring before eradication was confirmed, and four of the five eradications have been successful so far. None of these species had previously been eradicated in large numbers from other islands so the successes substantially advance this field of invasive species management. The challenges and insights of these eradications also provide unique learning opportunities for other invasive avian eradications.

Keywords: Aldabra, Indian Ocean islands, invasive alien bird management and control, mist-netting, parakeets, passerines, shooting

INTRODUCTION

Birds are currently one of the least represented groups of terrestrial vertebrates in the field of invasive alien species research and management, and the development of successful eradication strategies for introduced birds remains in its infancy, especially when compared to well-established invasive mammal control techniques (see: Blackburn, et al., 2009; Feare, 2010; Bauer & Woog, 2011; Strubbe, et al., 2011; Baker, et al., 2014; and Menchetti & Mori, 2014; for potential reasons for the discrepancy). The relatively few examples of successful large-scale avian eradications include rock pigeons (*Columba livia*) from the Galápagos Islands (Brand Phillips, et al., 2012), which at the time was the largest successful eradication of an alien bird from an island system (with 1,477 birds removed), and several eradications of the common myna (*Acridotheres tristis*) (e.g. Saavedra, 2010; Canning, 2011; Feare, et al., 2017). There has, however, been little development of best practices or compilation of lessons learnt so far. Furthermore, we are not aware of any examples of pre-emptive invasive bird eradications from islands to protect native biodiversity on nearby islands.

Since 2011, five separate eradication programmes for invasive alien birds have been conducted on three islands in the Seychelles by the Seychelles Islands Foundation with the aim of protecting endemic biodiversity on Aldabra Atoll and Praslin from the potential impacts of these invasive bird species, should they become established. These eradications targeted: (1) red-whiskered bulbuls (*Pycnonotus jocosus*) on the island of Assumption; (2) Madagascar fodies (*Foudia madagascariensis*) on Assumption; (3) red-whiskered bulbuls on Aldabra; (4) Madagascar fodies on Aldabra; and (5) ring-necked parakeets (*Psittacula krameria*) on the main Seychelles island of Mahé. Red-whiskered bulbuls have a broad introduced range covering 15 countries (Global Invasive Species Database (GISD), 2017), and their impacts on native ecosystems and biodiversity (Clergeau & Mandon-Dalger, 2001; Linnebjerg, et al., 2010; GISD, 2017) have prompted control efforts and even small-scale eradications, but these efforts have not been upscaled in most places. Madagascar fodies are widely introduced across the Western Indian Ocean islands including many of the Seychelles islands, where they threaten native avifauna through hybridisation (Lucking, 1997), and transmission of pathogens (de Sales Lima, et al., 2015). Ring-necked parakeets have been introduced to over 35 countries outside their native range, making them one of the most successful avian invaders in the world, and are known to cause detrimental impacts on native wildlife (Strubbe & Matthysen, 2007; Strubbe & Matthysen, 2009; Strubbe, et al., 2010; GISD, 2017), but have not yet been eradicated or substantially reduced in numbers from any of them.

In this paper, we present a general overview of each eradication including: (i) the main methods applied in each phase; (ii) the relative success and numbers of birds culled with each method; and (iii) the difficulties encountered. Finally, we suggest 10 key insights and recommendations that can be applied to further eradication attempts and adopted for best practice, and offer a positive outlook for the future of introduced bird management.

METHODS

Location and background of project

The Seychelles archipelago consists of 115 islands across the Western Indian Ocean region (Fig. 1). The country has two UNESCO World Heritage sites; Aldabra Atoll, which was inscribed on the World Heritage list in 1982, and the Vallee de Mai, inscribed in 1983. Aldabra (15,250 ha; 9°24' S, 46°20' E; Fig. 1), one of the largest raised coral atolls in the world, is famous for its remarkable

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biodiversity, including the largest giant tortoise population in the world, huge seabird colonies, pristine marine ecology and its relative lack of ecological disturbance. The Vallée de Mai (4°19′ S 55°44′ E), a 20 ha site on the island of Praslin (Fig. 1), is a mature palm forest dominated by the endangered endemic giant palm, the coco de mer (*Lodoicea maldivica*). A public trust, the Seychelles Islands Foundation (SIF), is responsible for managing and protecting both sites. The sites form crucial strongholds for many endemic and/or endangered species, and both sites host endemic bird species that face increasing threats from the invasive birds present on nearby islands. This context prompted SIF to consider and initiate pre-emptive management action of the introduced species in 2010, to ensure protection of the endemic species.

In the case of Aldabra, Assumption Island (1,171 ha, 9°44' S, 46°30' E; Fig. 1), only 27 km away, had populations of red-whiskered bulbuls and Madagascar fodies which were introduced in the 1970s. Aldabra's native avifauna, including the endemic Aldabra fody (*Foudia aldabrana*) and a native sub-species of Madagascar bulbul (*Hypsipetes madagascariensis rostratus*), had long been considered threatened by the proximity (*sensu* propagule pressure, Simberloff, 2009) of these introduced birds on Assumption (Roberts, 1988). The main threats posed by the potential spread of these introduced species to Aldabra were considered to be competition, hybridisation and transmission of novel pathogens. When the Assumption eradication of red-whiskered bulbuls and Madagascar fodies was being planned in 2010/2011, Aldabra was not known to have any introduced bird species and may have been the largest tropical island to be free of invasive birds. Unfortunately however, both of the introduced species from Assumption were identified on Aldabra in early 2012, soon after the start of the Assumption eradications. This was thought to be due to the increasing populations of both species on Assumption, so an additional eradication operation for these new populations on Aldabra was quickly planned.

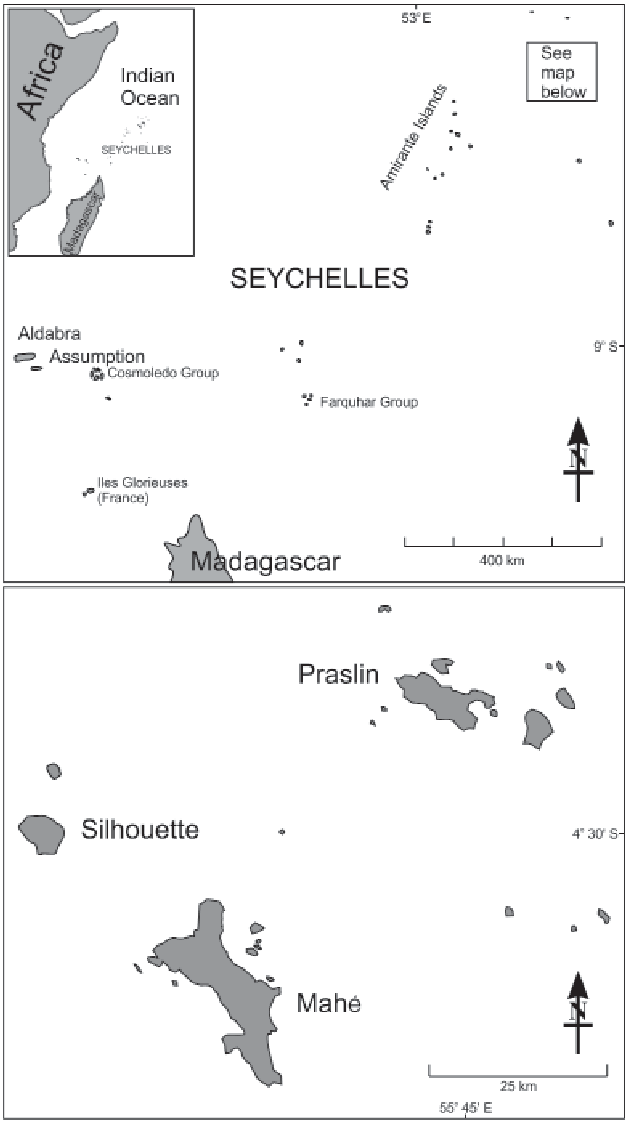

Fig. 1 Location of the Seychelles archipelago in the Indian Ocean (top, inset), the main islands and island groups of the Seychelles, including Aldabra and Assumption (top), and the inner Seychelles islands (bottom), showing Mahé, Silhouette and Praslin.

In the case of the Vallée de Mai, the mature coco de mer palm forest at this site forms the main breeding area for the Seychelles black parrot (*Coracopsis barklyi*), which is endemic to the island of Praslin and a flagship species for this island. The main Seychelles island of Mahé (Fig. 1), ca. 37 km away from Praslin, had a rapidly growing population of introduced ring-necked parakeets since the 1990s. The increasing probability of their establishment on Praslin, was accompanied by threats to the black parrot through competition and pathogen transmission. The presence of the parakeets on Mahé was thus considered the most pressing threat to these endemic parrots, which number only 520–900 birds on one island in the wild (Reuleaux, et al., 2013). In addition, long-term conservation plans for the black parrot include possible translocations of the species to other islands (Rocamora & Laboudallon, 2009) and such interventions could not be considered while ring-necked parakeets remained on Mahé.

Eradication time-frames and methods

All of the eradications were initiated in 2011/2012 and started with a 2–6 month initial phase, which included surveys to estimate the population size and distribution of the introduced bird populations, and trials to identify the most effective eradication methods.

Population estimates were carried out by island-wide distance sampling for Madagascar fodies and red-whiskered bulbuls on Assumption, grid-based surveys on Aldabra, and standardised roost counts for ring-necked parakeets on Mahé.

The choice of eradication methods trialled in the first phase of each project (see Table 1) was based on literature research, staff experience with the species, advice from experts, and experimentation. The trialled methods included trapping (using a number of types of trap, bait, trapping locations, decoys and playback), ground mist-netting, shooting and poisoning, as well as manual methods such as location and hand-capture of birds at nests and roosts. For the ring-necked parakeets, high-level (canopy) mist-netting was also trialled, which involved mist-nets set up in the canopy at 8–15 m from the ground using either bamboo poles or tree branches. The outcomes of these initial trials in terms of capture rates, efficiency, cost and labour intensiveness were then assessed and informed the choice of focal method(s) for the main phase of each eradication (Table 1). Thereby, the methods used for each eradication varied by island, species and phase of the project. Nevertheless, amendments needed to be made throughout the main eradication phase as the situation changed, so flexibility and adaptability in approach was essential.

The initial phase was followed by a second phase of intensive eradication efforts which lasted about three years for all of the eradications. During this phase the focus was on reducing the target bird population numbers to zero

Table 1 Trialled methods in phase 1 and focal methods in phase 2 for each of the eradications.

Island	Species	Methods trialled in phase 1	Phase 2 focal method(s)
Assumption	Red-whiskered bulbul	Trapping, shooting, poisoning, hand-capture at nests/roosts	Mist-netting, then shooting
Assumption	Madagascar fody	Trapping, shooting, poisoning, hand-capture at nests/roosts	Mist-netting, then shooting
Aldabra	Red-whiskered bulbul	Mist-netting, shooting	Mist-netting
Aldabra	Madagascar fody	Mist-netting, shooting, hand capture	Mist-netting, supplemented by shooting with air rifle & hand capture of fledglings
Mahé	Ring-necked parakeet	Trapping, canopy mist-netting, ground mist-netting, nest cavity targeting, shooting along flight lines and feeding areas	Shooting along flight lines and feeding areas with shotgun

as quickly and efficiently as possible using the methods identified in the trial phase. The second phase started in 2012 for Madagascar fodies and red-whiskered bulbuls on both islands, and in 2013 for the ring-necked parakeets after approval to use firearms was granted. It ended when no more birds could be detected.

Outreach was an important part of the ring-necked parakeet eradication in particular and efforts were made at the start of the intensive phase of this project to reach as many people as possible to encourage them to call the team with any information on sightings. We initially used all means available (including radio, TV, talks and presentations, newspaper and magazine articles, social media posts, website, newsletters, posters, stickers) to spread the message, and fine-tuned this according to responses over time.

The third and final phase consisted of monitoring (direct observations at all sites, island-wide point counts on Assumption; grid-based surveys in and surrounding the invaded area of Aldabra; roost and feeding tree checks at all known sites on Mahé) to confirm that no individuals of the target species remained. The monitoring was implemented in four 2–3 week periods with a team of 2–4 local scientific staff who had experience in one or more bird eradications, every 3–6 months.

RESULTS

Bird removal

Table 2 summarises pre-eradication population estimates and the total number of birds culled in each eradication, with estimates of the size of the introduced bird populations ranging from two to 4,300.

To date, four of the five eradications have been successful, with only the ring-necked parakeet eradication still in the monitoring phase. On Assumption and Aldabra, there were no sightings of either introduced bird species in two years of monitoring so both islands are again considered free of invasive birds.

Efficiency of control methods

The proportion of birds culled using different methods varied in each eradication (Fig. 2) and only a summary is provided here. The predominant and most effective methods for all campaigns were shooting and mist-netting (Table 1; Fig. 2).

For the ring-necked parakeets on Mahé, mist-netting caught 25 birds in the trials and first two months of the intensive phase of the campaign, but quickly became unfeasible as the birds learnt to avoid the nets even when set up in different places. Trapping caught no birds. The

Table 2 Summary of the pre-eradication population estimate and the number of birds culled for each of the target populations.

Island	Species	Pre-eradication population estimate	Number of birds culled	Population estimation method and reference
Assumption	Red-whiskered bulbul	4,300	5,279	Distance sampling; Feare & Fries-Linnebjerg, 2012
Assumption	Madagascar fody	1,600	3,291	Distance sampling; Feare & Fries-Linnebjerg, 2012
Aldabra	Red-whiskered bulbul	2–3	1	Direct observations; van de Crommenacker, 2012
Aldabra	Madagascar fody	150–200	262 (incl. hybrids)	Point counts; van de Crommenacker, 2012
Mahé	Ring-necked parakeet	288	545*	Roost counts; Birch, et al., 2012

* The 545 ring-necked parakeets included 543 from Mahé, one bird from Silhouette and one bird from Praslin. The single ring-necked parakeets culled on Praslin and Silhouette were assumed to have flown there from the Mahé population as there were no records of captive birds on either island.

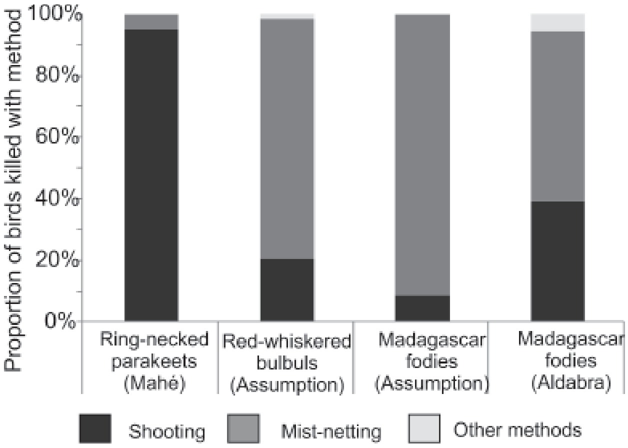

Fig. 2 The proportion (%) of birds culled on each introduced bird eradication (ring-necked parakeets [*Psittacula krameri*] from Mahé, red-whiskered bulbuls [*Pycnonotus jocosus*] and Madagascar fodies [*Foudia madagascariensis*] from Assumption; and Madagascar fodies from Aldabra) using the main eradication methods of shooting and mist-netting. 'Other methods' include trapping and manual capture. The red-whiskered bulbul eradication on Aldabra is not included in this figure because there was only one bird (mist-netted).

parakeet on Silhouette was culled by a member of the public using a catapult – a method not trialled on Mahé. The bird on Praslin was hand-caught.

Ring-necked parakeet eradication outreach

Television adverts were found to have prompted ca. 70% of all callers with information on sightings of the parakeets, with less than 10% of responders prompted by newspaper and magazine articles and the remainder from presentations, social media and having seen the posters.

DISCUSSION

Difficulties encountered

For each island and species, there was a particular set of challenges to overcome. On Aldabra, Madagascar fodies very closely resembled the endemic Aldabra fodies. This caused problems with capture of non-target species, and identification of introduced vs endemic species at a distance. The two species also quickly hybridised (van de Crommenacker, et al., 2015), making the eradication decisions more complex. Most birds targeted therefore needed to be identified at very close range to ensure that no Aldabra fodies were culled. Aldabra's physical challenges also included impenetrable vegetation, treacherous terrain and extremely demanding logistics. The invaded area was in the most remote part of Aldabra, initially had no field station, freshwater or facilities, and is only accessible via boat on a high tide, followed by a one-hour hike. Establishing basic infrastructure was therefore an essential pre-requisite for this eradication to proceed.

On Assumption the main challenges were higher than estimated population sizes of both target species, and the fact that neither species behaved as predicted from previous observations elsewhere. For example, trapping was initially anticipated to be an important and relatively simple capture method throughout the eradication, but this method appeared almost completely ineffective in extensive trials of the first phase. The failure of birds to accept bait (without traps) or to enter traps, combined with the high densities of both species, meant that mist-netting was by far the most effective capture method in the early part of the intensive eradication phase. This was labour-intensive and most successful when targeted at flight lines to and from nocturnal communal roosts. The propensity for communal roosting varied seasonally and the location of flight lines required constant monitoring to maximise mist-net captures. As numbers of target birds fell and mist-netting became less effective, shooting became the dominant method in the last year of the eradication. Both bird species on Assumption also appeared to be extremely wary of humans, even before the start of the project, and this became more marked as the eradication progressed. The originally planned methods therefore had to be re-assessed early in the project and underwent continual assessment as the eradication progressed.

On Mahé, mist-netting and trapping of ring-necked parakeets proved ineffective or inefficient – the birds were found to fly and roost usually too high for mist-netting, and several specialised trap designs (including the use of decoys) were unsuccessful. Poisoning could not be considered on Mahé because of possible effects on humans and non-target species. This left shooting as the only viable alternative, which was a politically and socially difficult method to adopt. Mahé is an inhabited island, with a population of ca. 80,000 people, and eradication activities had to occur in inhabited areas as the birds were predominantly observed in agricultural and cultivated areas with crops and fruit trees. The Seychelles is, for historical and security reasons, highly sensitive about the use of firearms and this resulted in a delay of two years before firearms were approved for use on the project. Shooting was then permitted to external hunters, provided they were accompanied by a military escort at all times and used only shot-gun and air rifle. Ring-necked parakeets are also highly intelligent birds and became 'educated' and wary very quickly. For example, we think they learnt to recognise and avoid the project car and staff uniforms. Shooting therefore had to be done with extreme caution (e.g. from cryptic locations, wearing camouflage gear, only shooting at groups of one or two birds, and shooting only when the hunter was very confident of a strike). A final critical issue with working on an inhabited island was public perceptions concerning the project, especially with such a charismatic target species and because the success of any eradication partly depends on public support and contribution (Mack, et al., 2000). SIF tackled this potential problem from the outset by conducting intensive outreach campaigns to try to ensure that as many people on Mahé as possible were aware of the eradication and the reasons for it. Lack of support did cause occasional problems with access to private land and misinformation. Fortunately, the parakeets were a known pest and commonly viewed as a threat to farming and endemic wildlife, so the majority of people encountered were in favour of the project and very supportive.

Ten key insights and recommendations

Here is a list of 10 key insights from these eradications, which can serve as a basis for recommendations for practitioners who are considering invasive alien bird eradications. The eradications presented here cover islands from both ends of the ecological disturbance spectrum, from the most ecologically depauperate (Assumption), to the least disturbed and most biodiversity-rich (Aldabra) making the lessons relevant to a broad suite of islands.

1. Large-scale invasive bird eradications are feasible

Red-whiskered bulbuls in the Seychelles occurred only on Assumption, plus the single bird on Aldabra, so the outcome of these eradications has been national elimination of the species. With the parakeet's range on the Seychelles encompassing only Mahé, if this eradication is successful, it will mark a second national eradication of an invasive bird species of high concern. Madagascar fodies remain established in high numbers on many islands of the Seychelles, but their eradication from two very different islands confirms the feasibility of this approach, should there be a need to consider their eradication elsewhere. Therefore, our first key message is that eradications of invasive alien birds from islands are feasible, even if the population of the target species exceeds 5,000 birds.

2. Pre-emptive action should be considered as a means to remove perceived threats

The three initial eradications of red-whiskered bulbuls and Madagascar fodies on Assumption, and ring-necked parakeets on Mahé, were based on the precautionary principle, i.e. the aim was to protect threatened endemic biodiversity pre-emptively based on perceived threats. This was justified in the case of the red-whiskered bulbul and the ring-necked parakeet, which have known detrimental impacts in their introduced ranges. However, even in the case of the Madagascar fody, the impacts of which on endemic birds have been questioned (Garrett, et al., 2007), the perceived threats were verified during the course of the eradications: (i) all three target species reached the islands of concern and at least one of these species established a breeding population (Madagascar fodies on Aldabra); (ii) hybridisation was confirmed to occur between introduced and endemic fodies on Aldabra (van de Crommenacker, et al., 2015); and (iii) several potentially novel pathogens were identified in the invasive species (SIF, unpubl. data).

3. Don't assume what you know of a species from other locations will apply in a new area – plan to conduct initial trials

Based on experience of the same species elsewhere, we expected a main method for catching Madagascar fodies and red-whiskered bulbuls to be trapping, and planned accordingly with respect to equipment and logistics. Trapping can be an effective capture method elsewhere for these species (N.B. & P.H., pers. obs.; C.G. Jones, pers. comm., all in Mauritius), but was found to be almost completely ineffective on Assumption for reasons that are unclear, and the birds never became accustomed to baited areas. This was despite several members of staff working on the project who had extensive experience successfully trapping these species in other locations. This caused delays at the beginning of the eradication while methods were re-assessed and other equipment sourced. A similar problem was encountered with the ring-necked parakeets, which have been successfully trapped in other countries (e.g. Bashir, 1979; Hussain, et al., 1992), but could not be trapped using the same or similar trap designs on Mahé, although these problems were less significant, as trapping parakeets had not been assumed as a main method of capture. It is important to note that we are not ruling out any particular method for targeting these species elsewhere. Trapping may still be a highly effective capture technique in other places for these species, so our advice here is simply that initial small-scale trials should be conducted to determine the feasibility of several different methods and save time and funding.

4. One size doesn't fit all birds

Shooting was by far the most effective method for ring-necked parakeets in the Seychelles, while mist-netting proved to be generally more effective for passerines (although this depended on the phase of the eradication). However, a flexible approach and willingness to modify the strategy was critical for the success of these eradications. Even within the same species and island, our techniques needed to be assessed and 'tweaked' frequently (and often substantially) to maintain efficient capture rates. For example, on Assumption, there was a switch in the final year of the intensive phase of the eradication, from using mist-nets as the main method of capture to firearms (this switch also applies to mynas; Feare, et al., 2017). This was decided when catch rates in mist-nets (i.e. the density of target population) had dropped too low for continued progress with the eradication (i.e. population recruitment rates were thought likely to be equal to or higher than capture rates).

5. Don't count your eggs before they hatch

For all three species, there were more birds present than had been estimated by survey methods. This was the case regardless of which estimation method was used. The higher numbers are likely to have been primarily due to recruitment of young birds into the populations since distance sampling is based on classical closed population sampling (Cassey & McArdle, 1999) but the survey methods (roost counts, distance sampling) could also have produced underestimates. The higher figures had implications for the planning and especially the costs of completing the eradications.

6. Identify the weak points of your target species

Each target species was found to have at least one trait or habit which either increased their vulnerability at certain times or to certain methods, or could be used to improve eradication effectiveness. The communal roosting sites of ring-necked parakeets enabled regular standardised counts to be conducted, which initially provided a valuable way to monitor the population numbers, flight lines and the impacts of the eradication efforts and later formed an essential location for targeting the remaining birds. These sites proved so useful that parakeets were not targeted at roosts until close to the end of the project to ensure that the roost sites were not disturbed or compromised. Red-whiskered bulbuls also roosted communally in the early stages of the project and could be targeted with mist-nets along their flight lines towards roosts, which maximised the mist-net catch. Later in the eradication, their habit of vocalising from prominent perches meant that they could be reliably located from several hundred metres away, which greatly helped in the search for and targeting of the last few birds. Madagascar fodies were found to have a tendency to form large foraging groups, especially in the non-breeding season, which, when spotted, provided key areas for mist-netting.

7. Use research to aid management decisions on the ground

A scientific and research-based approach was an important aspect of the eradications and greatly facilitated management decisions on the ground. This included collecting comprehensive data and samples from all birds caught, regularly analysing the effectiveness of methods and approaches, and setting up external research partnerships for analysis which could not be done on site. The strongest example of this was the case of the Madagascar fody introduction to Aldabra, for which SIF was able to quickly establish a collaboration with university researchers, ensuring that the samples and data collected could be rapidly and effectively analysed. The resulting research outputs included analysis of origin (Assumption) and timing (recent, but probably pre-dating the start of the eradication) of the invasion, as well as confirmation of hybridisation between the endemic and introduced species and more insights into this process (van de Crommenacker, et al., 2015). A collaboration was also established for disease-screening of ring-necked parakeet and black parrot blood samples to provide information on the pathogen status of each species.

8. Training of local staff is essential for project success

Few people with the necessary technical skills needed for the eradications existed in the Seychelles when the project started, so more than 30 local staff were intensively trained on the job throughout the eradications. Five of these staff members subsequently led parts of the eradications and were crucial to their success. Several of the staff members have subsequently been recruited in other invasive species management positions within SIF and elsewhere, so the eradications have increased in-country capacity in this field. Indeed, local staff training is seen as one of the biggest achievements of the eradications and has had the additional benefit of providing a strong sense of national ownership to the eradications.

9. Assess effectiveness of publicity and focus on the most appropriate means

Outreach activities are important in any eradication but in some, they are an essential means of achieving success. For the ring-necked parakeet eradication, on assessing where callers had heard about the project we found that the vast majority were prompted by the TV advert so we were able to focus on this for the rest of the project, which reduced costs and time without compromising the information received. In addition to public outreach, we found it was essential to liaise with other stakeholders in the environmental sector about the importance of the eradications. We noticed that the eradications tended to bring out strong feelings either for or against the project, and most people appreciated an opportunity to ask questions and understand the reasons for it. Our impression was that the outreach and education carried out for these projects went a long way to increase public support although we have no way of quantifying this

10. The early bird catches the worm

In the case of these eradications, we are certain that pre-emptive action has been a more effective and cost-efficient strategy to protect endemic species than would have been the case had we waited for the introduced species to spread to Aldabra and the Vallée de Mai (or other islands in the Seychelles) and establish populations. Indeed, this had already started to happen with all three species and, had we waited much longer, eradication may have proved an impossible task. Finally, at least one and potentially two of these invasive bird species are now nationally eradicated from the Seychelles and there is minimal risk of them being reintroduced to the sites in the future. We therefore consider the biodiversity and ecological integrity of the Seychelles World Heritage sites to have been safeguarded from these particular threats by these eradications.

CONCLUSION

Although all three species targeted here are known invasive species, and control efforts have been made or are underway in several places, there were no previous records of them being removed in such large numbers, or their complete eradication from any other islands or countries. The challenges and successes of these eradications provide a unique learning opportunity and offer a positive outlook for the future of introduced bird eradications. The fact that these eradication successes (or near successes) in the Seychelles are the first of their kind suggests that a change in approach and mindset to invasive bird eradications is timely. We believe that insights gained from these programmes can be used as a basis to significantly advance the field of invasive bird management and to initiate the development of best practices for eradication attempts.

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REFERENCES

Baker, J., Harvey, K.J. and French, K. (2014). 'Threats from introduced birds to native birds'. *Emu* 114: 1–12.

Bashir, W.A. (1979). 'A new "PAROTRAP" adapted from the Mac trap for capturing live parakeets in the field.' *Wildlife Damage Management, Internet Center for Bird Control Seminars Proceedings* 23, pp. 167–171. Lincoln, US: University of Nebraska.

Bauer, H-G. and Woog, F. (2011). 'On the 'invasiveness' of non-native bird species'. *Ibis* 153: 204–206.

Birch, D., Haverson, P., Simeon, J. and Constance, A. (2012). *Phase 1 Report: Inception Report and Plan to Eradicate the Ring-necked Parakeet (Psittacula krameri) from Mahé, Seychelles (Project Objective 2b).* Unpublished report. Seychelles. Seychelles Islands Foundation.

Blackburn, T.M., Lockwood, J.L. and Cassey, P. (2009). *Avian Invasions: The Ecology and Evolution of Exotic Birds*. Oxford, UK: Oxford University Press.

Brand Phillips, R., Cooke, B.D., Carrión, V. and Snell, H.L. (2012). 'Eradication of rock pigeons, *Columba livia*, from the Galápagos Islands'. *Biological Conservation* 147: 264–269.

Canning, G. (2011). 'Eradication of the invasive common myna, *Acridotheres tristis*, from Fregate Island, Seychelles'. *Phelsuma* 19: 43–53.

Cassey, P. and McArdle, B. H. (1999). 'An assessment of distance sampling techniques for estimating animal abundance'. *Environmetrics* 10: 261–278.

Clergeau, P. and Mandon-Dalger, I. (2001). 'Fast colonization of an introduced bird: the case of *Pycnonotus jocosus* on the Mascarene Islands'. *Biotropica* 33: 542–546.

Feare, C.J. (2010). 'Invasive bird eradication from tropical oceanic islands'. *Aliens: The Invasive Species Bulletin* 30: 12–19.

Feare, C.J. and Fries-Linnebjerg, J. (2012). *Eradication of Avian Invasive Alien Species on Assumption Island Including Feasibility Studies, and Development and Implementation of Eradication Plan (Project Objective 1a). Phase 1 Report: Inception Report and Eradication Plan*. Unpublished report. Seychelles. Seychelles Islands Foundation,

Feare, C.J., van der Woude, J., Greenwell, P., Edwards, H.A., Taylor, J.A., Larose, C.S., Ahlen, P-A., West, J., Chadwick, W., Pandey, S., Raines, K., Garcia, F., Komdeur, J. and de Groene, A. (2017). 'Eradication of common mynas *Acridotheres tristis* from Denis Island, Seychelles'. *Pest Management Science* 73: 295–304.

Garrett, L.J., Jones, C.G., Cristinacce, A., and Bell, D.J. (2007). 'Competition or co-existence of reintroduced, critically endangered Mauritius fodies and invasive Madagascar fodies in lowland Mauritius?' *Biological Conservation* 140: 19–28.

GISD (2017). 'Global Invasive Species Database' <http://www.iucngisd.org/gisd/speciesname/Psittacula+krameri> and <http://www.iucngisd.org/gisd/speciesname/Pycnonotus+jocosus>. Accessed 18 August 2017.

Hussain, I., Khan, A.A. and Munir, S. (1992). 'Trapping success of PAROTRAP.' *Pakistan Journal of Zoology* 24: 170–172.

Linnebjerg, J., Hansen, D.M., Bunbury, N. and Olesen, J.M. (2010). 'Diet composition of the invasive red-whiskered bulbul *Pycnonotus jocosus* in Mauritius'. *Journal of Tropical Ecology* 26: 347–350.

Lucking, R.S. (1997). 'Hybridization between Madagascan red fody *Foudia madagascariensis* and Seychelles fody *Foudia sechellarum* on Aride Island, Seychelles'. *Bird Conservation International* 7: 1–6.

Mack, R.N., Simberloff, D., Lonsdale, W.M., Evans, H., Clout, M.N. and Bazzaz, F.A. (2000). 'Biotic invasions: causes, epidemiology, global consequences, and control.' *Ecological Applications 10*: 689–710.

Menchetti, M. and Mori, E. (2014). 'Worldwide impact of alien parrots (Aves Psittaciformes) on native biodiversity and environment: A review'. *Ethology Ecology and Evolution* 26: 172–194.

Reuleaux, A., Bunbury, N., Villard, P. and Waltert, M. (2013). 'Status, distribution and recommendations for monitoring of the Seychelles black parrot *Coracopsis* (*nigra*) *barklyi*'. *Oryx* 47: 561–568.

Roberts, P. (1988). 'Introduced birds on Assumption Island—a threat to Aldabra'. *Oryx* 22: 15–17.

Rocamora, G. and Laboudallon, V. (2009). *Seychelles Black Parrot* Coracopsis (nigra) barklyi *Conservation Assessment and Action Plan. 2009–2013. FFEM Project 'Réhabilitation des Ecosystèmes Insulaires'*. Seychelles: Island Conservation Society and MENRT.

Saavedra, S. (2010). 'Eradication of invasive mynas from islands: Is it possible?' *Aliens: The Invasive Species Bulletin*: 29: 40–47.

de Sales Lima, F.E., Gil, P., Pedrono, M., Minet, C., Kwiatek, O., Campos, F.S., Spilki, F.R., Roehe, P.M., Franco, A.C., Maminiaina, O.F., Albina, E. and de Almeida, R.S. (2015). 'Diverse gammacoronaviruses detected in wild birds from Madagascar'. *European Journal of Wildlife Research* 61: 635–639.

Simberloff, D. (2009). 'The role of propagule pressure in biological invasions'. *Annual Review of Ecology, Evolution, and Systematics* 40: 81–102.

Strubbe, D. and Matthysen, E. (2007). 'Invasive ring-necked parakeets *Psittacula krameri* in Belgium: Habitat selection and impact on native birds'. *Ecography* 30: 578–588.

Strubbe, D. and Matthysen, E. (2009). 'Experimental evidence for nest-site competition between invasive ring-necked parakeets (*Psittacula krameri*) and native nuthatches (*Sitta europaea*)'. *Biological Conservation* 142: 1588–1594.

Strubbe, D., Matthysen, E. and Graham, C.H. (2010). 'Assessing the potential impact of invasive ring-necked parakeets *Psittacula krameri* on native nuthatches *Sitta europaea* in Belgium'. *Journal of Applied Ecology* 47: 549–557.

Strubbe, D., Shwartz, A. and Chiron, F. (2011). 'Concerns regarding the scientific evidence informing impact risk assessment and management recommendations for invasive birds'. *Biological Conservation* 144: 2112–2118.

van de Crommenacker, J. (2012). *First Annual Report: Takamaka Invasive Bird Project 2012*. Unpublished report. Seychelles. Seychelles Islands Foundation,

van de Crommenacker, J., Bourgeois, Y.X.C., Warren, B.H., Jackson, H., Fleischer-Dogley, F., Groombridge, J. and Bunbury, N. (2015). 'Using molecular tools to guide management of invasive alien species: Assessing the genetic impact of a recently introduced island bird population'. *Diversity and Distributions* 21: 1414–1427.

House sparrow eradication attempt on Robinson Crusoe Island, Juan Fernández Archipelago, Chile

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Abstract House sparrows (*Passer domesticus*) compete with native bird species, consume crops, and are vectors for diseases in areas where they have been introduced. Sparrow eradication attempts aimed at eliminating these negative effects highlight the importance of deploying multiple alternative methods to remove individuals while maintaining the remaining population naïve to techniques. House sparrow eradication was attempted from Robinson Crusoe Island, Chile, in the austral winter of 2012 using an experimental approach sequencing passive multi-catch traps, passive single-catch traps, and then active multi-catch methods, and finally active single-catch methods. In parallel, multiple detection methods were employed and local stakeholders were engaged. The majority of removals were via passive trapping, and individuals were successfully targeted with active methods (mist nets and shooting). Automated acoustic recording, point counts and camera traps declined in power to detect individual sparrows as the population size decreased; however, we continued to detect sparrows at all population densities using visual observations, underscoring the importance of local residents' participation in monitoring. Four surviving sparrows were known to persist at the conclusion of efforts in 2012. Given the lack of formal biosecurity measures within the Juan Fernández archipelago, reinvasion is possible. A local network of citizen observers is the best tool available to detect house sparrows at low density, however ongoing, dedicated eradication funding does not exist. Opportunistic removals via shooting have been possible from 2013–2016, but elusive individual sparrows were seen during a small number of days each year suggesting remnant group(s) exist in yet unknown forest locations.

Keywords: invasive bird, management, *Passer domesticus*, trapping

INTRODUCTION

House sparrows (*Passer domesticus*) have a wide range of negative impacts in areas where they have been introduced. They affect native bird species, pushing eggs and nestlings from nests and chasing adults (McGillivray, 1980; Gowaty, 1984); they consume crops and ornamental plants; and are vectors of at least 29 diseases affecting people, livestock and wildlife (Clergeau, et al., 2004; Fagerstone, 2007). This species is an effective invader owing to its generalist diet; rapid rate of increase, facilitated by colonial-communal nesting, large clutch sizes and extended breeding seasons; effective range expansion in human-altered landscapes; and aggression against similar and smaller sized birds (MacGregor-Fors, et al., 2010). The risks of house sparrows are often underestimated and delays in rapid responses to incipient or small localised populations can result in much more complex and costly future actions for their management once population growth and negative impacts on native species are documented (Clergeau, et al., 2004). Complete removal of invasive house sparrow populations should be considered to eliminate risk for negative impacts arising from the species' invasion.

House sparrow eradication attempts on other islands have demonstrated that the effectiveness of some methods may decline over time, if sparrows learn to avoid them (Bednarczuk, et al., 2010) emphasising the importance of using a variety of techniques in an adaptive management approach. Campaigns for house sparrow eradication should employ multiple methods and aim to remove the entire population within as short a time as possible. Otherwise, given the species' reproductive potential, there is a risk that house sparrows will breed faster than they are removed. To maintain naïveté of the population to methods for as long as possible and reduce the likelihood of house sparrows dispersing in response, methods should be implemented strategically. The detection and removal of the last individuals must be considered in planning the deployment of the multiple alternative methods available (Morrison, et al., 2007). To increase likelihood of successful eradication, some methods should be deployed consecutively and others sequentially with attention to maintaining sparrows naïve to methods.

The Juan Fernández Archipelago in Chile is comprised of three islands (Robinson Crusoe (4,790 ha), Alexander Selkirk (4,950 ha),) and Santa Clara (220 ha)) with globally significant biodiversity and endemism due to its isolation and topographic variation. However, invasive species continue to drive catastrophic changes to these unique natural values including species extinctions and massive erosion, as well as precipitous declines in plant and animal species and loss of native vegetation cover (Sanders, et al., 1982; Bourne, et al., 1992; Arroyo, 1999; Hahn & Römer, 2002). Feasibility of the complete removal of invasive species has been explored and participatory planning with the islands' inhabitants and varied stakeholders continues to advance as benefits of invasive species removals and restoration are prioritised (Saunders, et al., 2011; Glen, et al., 2013; Ministerio del Medioambiente, 2017).

House sparrows have been present on Robinson Crusoe Island (RC) since 1943 as a wild population (Hahn, et al., 2006) and none are kept as pets. The population appeared stable at around 80 individuals and to be restricted to the island's only human settlement of San Juan Bautista (Hahn, et al., 2006; Hagen pers. obs.); however, observations in 2011–2012 indicated population expansion within San Juan Bautista into new home construction areas following a tsunami in February 2010. The potential increased risk from this species to single-island endemic birds and local food production prompted a review of control and eradication options within a local multi-stakeholder group focused on animal issues related to conservation and local development.

The study reports on an attempt to eradicate the local house sparrow population within an experimental framework to examine the efficacy of methods for house sparrow eradications and protect local biodiversity. The

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objectives of the study were to keep house sparrows naïve and eliminate the potential for survivors to learn to avoid methods (e.g. escape from traps).

METHODS

A range of potential methods for use in house sparrow eradication from RC were considered (see Table 22 of Saunders, et al., 2011). Removal techniques were evaluated and prioritised based on previous success in bird removals, permissibility in this urban setting, and likelihood to contribute to sparrow learning. Toxicants were assessed, but none were considered suitable for house sparrow eradication (Fisher, et al., 2012). Trapping was identified as having the greatest potential to provide a large reduction in the house sparrow population on RC while minimising risks to native birds and poultry. Pre-baiting was initiated one month before removals began (15 June 2012) at 10 sites to allow house sparrows to become accustomed to feeding at a given location on provided crushed maize (1.6–3.2 mm diameter) and to confirm minimal attraction of non-target species to these sites.

Passive removal techniques were employed in the first phase of this trial, to minimise education of house sparrows to future methods (10 July 2012–14 September 2012). Active removal techniques were added to the trial beginning 27 July 2012.

Passive removal techniques

To minimise education of house sparrows in the population, passive traps were employed in the initial phase of removals.

Elevator multi-catch traps have demonstrated good capture and low escape rates (Fitzwater, 1981). House sparrows enter a compartment alone to feed on bait, their body weight causes an "elevator" to lower the individual to its "escape" into a closed cage. Without the bird's weight, the counterbalanced "elevator" springs back into the original position ready for another passenger. Birds trapped in the closed cage act as live decoys. We purchased traps without the central mesh body for ease of transport, and then assembled the mesh over a plywood base forming the holding cage once on the island. Trap dimensions were 60 × 40 × 20 cm (<http://www.sparrowtraps.net/index.htm>). Elevator traps were placed on an elevated platform, approximately 2 m in height, to reduce the potential for trap interference by domestic animals and private citizens. We added a covered plywood compartment with a perch within each elevator trap's holding cage to provide protection from the elements for live decoys. Decoys had primary flight feathers on one wing clipped so that they couldn't fly in the event of escape. Food and water were provided.

Trio multi-catch traps are comprised of two compartments which each function as a single-catch trap, whose sprung doors must be manually reset after each catch (Nature-House ST1 Trio house sparrow trap http://www.amazon.com/Nature-House-ST1-Trio-Sparrow/dp/B001GIP2MG). The bird drops into the compartment, onto a perch over the feed tray which triggers the compartment door to close. Captured individuals can freely move into the third compartment, where they act as live decoys. Three trio traps were deployed, mounted at least 1.5 m above the ground to reduce potential for trap interference by domestic animals and private citizens. We provided flooring in each compartment to increase bait retention and partial roofs to decrease interference from natural elements.

Modified Australian crow (MAC) traps function when birds drop into the MAC trap to access bait and are unable to fly through the trap entrance to escape as their wingspan exceeds the diameter of the entrance. Captured individuals alight on perches in the higher parts of this trap (Clark & Hygnstrom, 1994). Exclusive use of a 'mini' MAC trap has enabled local populations of house sparrows to be entirely removed (McGregor & McGregor, 2008). We constructed two mini MAC traps, retaining traditional width of slats and height of centre board to avoid birds jumping to escape, reducing overall length (82 × 137 × 71 cm). MAC traps were placed on the ground given their robust size.

Nest box traps were made from nest boxes which were converted into single-catch traps (http://www.vanerttraps.com/urban.htm) to capture house sparrows investigating nest cavities. In areas where house sparrows were seen entering and exiting cavities, known cavities were covered to exclude sparrows and nest box traps were deployed with small feathers and fine nesting material added to the entrances to encourage investigation.

Traps were placed within open areas where birds could easily see them, and near frequented flyways, perches and feeding areas. For 2–3 days before arming traps, wired-open traps were placed at pre-baiting sites, with crushed maize on and around the open traps, to permit birds to explore them without risk of capture. When birds were trapped, the trap would be covered with a bed sheet to assist calming the birds during transport and reducing visibility to the general public. Covered traps were then transported to a room where any escapees could be recaptured, prohibiting escape. Within this facility birds were removed from traps and either selected for use as live decoys, or euthanised. Euthanasia was via cervical dislocation; possibly the easiest means for this species and a practical means for mass euthanasia (Sharp & Saunders, 2005; AVMA, 2007).

Active removal techniques

As capture rates declined with passive traps, active removal techniques were added to the trial. We continued using passive traps simultaneously with active removal techniques.

Walk in cage traps were used to target individual sparrows unable to be trapped in other trap types. A wooden box with mesh sides was set up as a walk-in cage trap by propping open a door that opens from the bottom. When the prop is pulled out by a nearby observer (Sharp & Saunders, 2005), bungee cords add to downward force to close the door quickly.

Clap traps utilise a spring-loaded throw net triggered remotely by the trapper, which is placed on the ground and pre-baited with crushed maize (<http://pestbarrier.com/store/itemdesc.asp?xCc=8u4u3>). The trap was not triggered unless all birds in a flock were able to be captured.

Mist nets are a common ornithological capture technique for small birds and were deployed on flyways to capture house sparrows that had avoided traps. Continual monitoring was required to quickly remove any house sparrows or non-target species.

Nest destruction can be used during the breeding season to slow or halt recruitment, and may make adult birds more susceptible to other techniques such as clap traps baited with nest material (Fitzwater, 1994). Eggs are crushed and nestlings euthanised (Sharp & Saunders, 2005). Nest destruction, although planned, was not needed in our trial.

Shooting was employed in specific scenarios where traps were proving ineffective. A 0.177 caliber air rifle with 4–12 times magnification scope (Beeman R9, Weirauch, Germany) was utilised, targeting only individuals alone or in pairs, to avoid wariness. Adult females were targeted first, to limit potential growth of the local population. After 2012, shooting was employed opportunistically.

Detection techniques

Eradication campaigns rely on effective detection of the target species to indicate when individuals of the target species no longer exist and the campaign can conclude. We assessed potential detection techniques for house sparrows throughout the trial, to examine their efficacy at varying house sparrow population densities. We anticipated that some detection methods may become innefective at low population densities as changes in flocking, calling and movements may result from individuals. Therefore we deployed multiple detection techniques simultaneously in order to ensure at least one technique was effective at even low population densities.

Autonomous recording units (ARUs) were deployed at 15 sites within San Juan Bautista. Ten ARUs were co-located with pre-baiting locations while the remaining units were in locations without pre-baiting. We programmed ARUs to record every other day for a 4-hour period around dawn (starting 30 minutes before sunrise) when house sparrows are known to be acoustically active. In addition, each sensor was programmed to record one of every 10 minutes throughout the rest of the day until 30 minutes after sundown. Data from these recordings was available only after post-processing in a sound laboratory. Automated analysis of all field recordings was carried out with the eXtensible BioAcoustic Tool (XBAT, <http://www.xbat.org>) using an image processing technique known as spectrogram cross correlation to detect and classify sounds on our field recordings that were correlated with the spectral qualities of typical house sparrow calls. Sensitivity in the detection analysis was increased to improve the probability of detecting house sparrow calls when few individuals remained, which led to manual review of all events to confirm accuracy of detecting true house sparrow calls (McKown, 2013).

Visual observations were conducted over the same period to provide alternative detection methods in the case that a given method failed to detect individuals even though a population remains present. Fixed radius point counts (Bibby, 2000; Buckland, et al., 2001) were conducted weekly beginning 15 June 2012. Project personnel conducted point counts 14 times throughout the trial period at 21 locations throughout San Juan Bautista, 15 of these locations were co-located with ARU deployment sites and six of which were not located with acoustic sensors or pre-baiting locations. Point counts were analysed using the fixed-radius point count equation as detailed by Buckland, et al. (2001), generating density estimates by habitat type, based on the estimated total surface area of coverage class occupied by sparrows (settlement and cultivated *Eucalyptus, Cupressus* and *Pinus* per Greimler, et al., 2002). Point count density estimates were compared to recorded call rates and sparrow removals each week.

In addition to point counts, citizens were encouraged to report opportunistic sightings of house sparrows, which were all investigated by project staff. Multiple reports of the same individuals, as well as uncorroborated reports prevented clear calculations of the number of individuals remaining.

Camera traps (Reconyx, Holmen WI) were deployed opportunistically at pre-baiting and passive trapping locations. Camera traps were used as an additional technique for visual confirmation of surviving individuals.

After the intensive 2012 campaign, an early observer's network attempting to harness the interest and participation of island residents was developed. This network has grown, and has become a formalised early detection network for invasive species, with individuals' observations of invasive species combined with a common smartphone application (WhatsApp) which allows researchers to capture reports within a database.

Stakeholder communications

Throughout the project, a communications campaign was undertaken to highlight the threats that house sparrows pose to local endemic species. Announcements via radio, signs, fliers, and a booth at a children's day event, were complemented with active participation in the local conservation committee, opportunistic presentations for local institutions and a nest box design contest for local endemic bird species. We also promoted the needs for biosecurity and a municipal ordinance to be established to regulate entry and possession of invasive species.

RESULTS

Methods to maximise personnel efficiency were deployed while reducing the risk of educating animals. Passive multi-catch traps (elevator, trio and mini-MAC traps) were deployed first. As nest-building behaviour was observed, passive single-catch nest box traps were deployed. As the number of individual sparrows was reduced, specific individuals were targeted with more time-intensive active multi-catch traps (mist net, clap trap and walk-in trap). Shooting (active, single-catch technique) was reserved for specific scenarios once other methods appeared ineffective.

Personnel contributed a total of 2,600 person hours across two months of sustained effort. A total of 814 trap days were conducted during the trial, resulting in 89 house sparrows removed. The majority of removals resulted from elevator traps (46 individuals, 275 trap days), followed by mist nets (22 individuals, 22 trap days) and trio traps (15 individuals, 70 trap days). Additional methods did not capture birds (modified MAC, walk-in cage, and clap traps) or were used in specific situations, after the population was reduced, and thus removed fewer birds (nest box trap, 1; shooting, 5). At the conclusion of the trial, four house sparrows were known to remain on the island (two males and two females).

Mist nets and shooting were the most effective removal techniques when effectiveness is assessed as the number of individual sparrows removed as a function of the days the technique was deployed. However, both of these active methods can educate individuals in the target population and require much higher personnel effort as compared to passive trap deployment (for example elevator traps and trio traps), demonstrating that this calculation of effectiveness is incomplete. Also, house sparrows captured in traps appeared to be useful as decoys; however, data specific to differential capture rates is not available.

Detection techniques

Both point counts and automated surveys detected a decline of house sparrows after house sparrow removals occurred. Point count density estimates showed abrupt declines after 60 individuals had been removed from the population, while call rates estimated from ARUs varied more gradually over the trial period (McKown, 2013; Fig. 1). Point count observers did not detect house sparrows

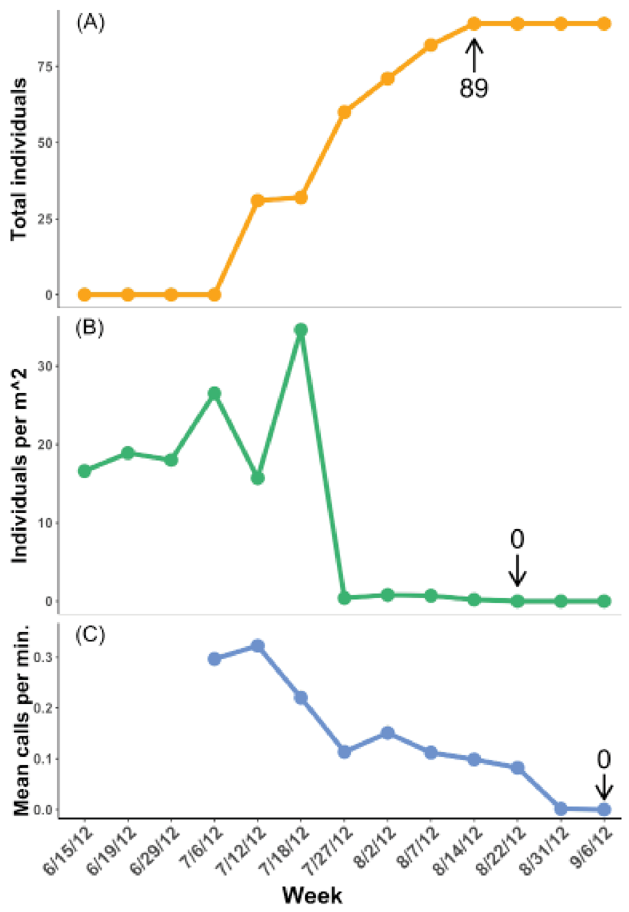

Fig. 1 Results of house sparrow removals over time (month/day/year), as well as detections from point count estimates and acoustic recordings. The cumulative total of house sparrows removed is presented (A) over the same time period that weekly density estimates were calculated from point count observations (B) and mean call rates by house sparrows (C), reported as averages over the previous survey week (McKown, 2013).

after 15 August 2012, while ARUs continued to detect house sparrow activity for 10 additional days. Both point counts and automated surveys failed to detect individual house sparrows known to be present by opportunistic observations on the island in early September 2012; neither point counts nor automated acoustic surveys were effective detection methods at low sparrow densities. Reports and observations made by community members were initiated in July 2012 and continue to date. These observations are a critical component of visual observations as they increase the effective coverage of the dedicated eradication team in area as well as time. In 2014 observations were also being made through the smartphone network, as well as through personal communications.

A total of 1,179 hours of acoustic recordings were collected and analysed from July to September 2012. All 79,822 events detected as potential house sparrow vocalisations were manually reviewed to confirm accuracy. Mean house sparrow acoustic activity, at all surveyed sites with data, declined from an average of 0.3 calls per minute in July 2012 to no calls by the end of August when a low number of individual house sparrows remained on the island (McKown, 2013).

Camera traps effectively captured images of house sparrows visiting known food sources. Given the trial setting in San Juan Bautista, some sites were inefficient for house sparrow detections via camera given that domestic animals, people, and objects moving in the wind would trigger the camera traps resulting in a significant number of images without the target species present. Camera traps did not capture images of individuals when population density was lowered by removals (after 15 August 2012), demonstrating ineffectiveness as a detection method for sparrows at low population densities.

The remnant house sparrows were infrequently detected within the town area between 2012 and 2016 and were reported by residents. Observations were limited to isolated localities and dates (20–23 June 2013, one individual detected and removed; 14 and 23 November 2015, one individual detected; 1 November 2016, five individuals detected; 19–30 October 2016, six individuals detected, three removed). Remaining house sparrows successfully avoided removal techniques and, based on inability to detect them, are thought to spend most of the year outside of the town area. It is uncertain whether or not house sparrows have continued to arrive via cargo ships from mainland Chile.

In addition to house sparrow detections, shiny cowbirds (*Molothrus bonariensis*) have been detected through the citizen observers network (15 March 2016, two individuals detected and removed; 20–24 April 2017, two individuals detected, one removed; Hagen, unpublished data).

Stakeholder communications

Dedicated efforts for regular, personalised and transparent communications about the trial and its goal to benefit native biodiversity were invested before, during and after the trial. Emphasis was given towards communications with homeowners at or near removal sites, as well as broad community-wide communications to minimise misinformation. Project personnel questioned while working always provided community members their attention, answering questions and continuing conversations as needed. A dedicated outreach coordinator led interactions with site owners and local institutions, served as primary point of contact for stakeholder concerns and provided regular updates to stakeholders regarding trial status and advances.

DISCUSSION

The house sparrow has aggressive foraging and nesting behaviour towards native bird species and is one of the most widespread invasive bird species throughout the world (Anderson, 2006). The house sparrow population expansion on Robinson Crusoe Island caused concerns for impacting vulnerable island endemic birds such as the Juan Fernández firecrown (*Sephanoides fernandensis*) and the Juan Fernández tit-tyrant (*Anairetes fernandezianus*), species which already co-occur with house sparrows (Hahn, et al., 2005). Given that house sparrows were pro-actively eliminated from neighbouring Alejandro Selkirk Island in 1994 (Hahn, et al., 2009), there was local interest in their removal from Robinson Crusoe Island while they were still restricted to one area of the island.

Worldwide, invasive bird eradications have received criticism for perhaps not being the highest need or having substantial evidence related to their impacts (Strubbe, et al., 2011). The precautionary principle may be invoked in decisions of eradicating potential threats before ecological damage is documented and the invasive bird establishes a population; in fact, this early action may be the only option for removing highly mobile bird species in some places and can definitely be the most economical option (IUCN, 2000; Baker, et al., 2014; Martin-Albarracin, et al., 2015). On Robinson Crusoe Island, house sparrow eradication and related activities as a community engagement and invasive species awareness-building technique for a broader invasive species programme (Glen, et al., 2013) were used. By working within the island's only town and with dedicated transparent communications focused on native species conservation, a coalition of homeowners was built that not only actively asked questions about invasive species management, but also contributed observations regularly to an early observer's network. This network has grown and today is formalised as an early detection network, continuing to rely on individuals' observations of invasive species by word of mouth, phone and smartphone application as a critical part of invasive species management (Ministerio del Medioambiente, 2017).

At the conclusion of the trial in late 2012, local decision-makers were interested in completing the house sparrow eradication, however the only detection techniques effective at low population densities are opportunistic visual observations. A wide network of citizen observers has successfully indicated presence and locations of house sparrows on Robinson Crusoe in following years; however detailed observations that lead to successful removals require effort-intensive follow-up by specialized personnel. Follow-up trapping has not been successful, however removals by shooting have occurred. It is unclear how many individuals remain, however they tend to be reported in the period from October to January. Multiple methods were ineffective at detecting the presence of remaining house sparrows at low population density, complicating the ability to assess eradication success probability without considerable observer effort across the island. Statstical frameworks developed to assess the probability of eradication confirmation success for other species may lend themselves to adjustments for invasive bird eradications and should continue to be explored (Ramsey, et al., 2011; Samaniego-Herrera, et al., 2013). There is no local institution able to dedicate staff to responding to observations, and so reported sparrow sightings and opportunistic removals are recorded in an exotic species database, including detections and removals from Alejandro Selkirk Island in 2016. The complete removal of house sparrows from the Juan Fernández Archipelago is possible with continued observations and removals; however, the arrival of additional individuals from continental sources via cargo boats is likely as no formal biosecurity measures exist and established municipal ordinances cannot restrict these movements. Persistent threats to native avifauna from introduced species continue to exist in the absence of formal biosecurity and environmental protection legislation.

Worldwide we are aware of at least 23 documented bird eradication attempts (DIISE, 2016, using data classified as good or satisfactory quality, and whole island eradications only). Bird eradication projects are more challenging compared to mammal eradications because volant birds fly more readily between adjacent islands, leading to higher rates of reinvasion (thus necessitating definition of eradication units for eradication planning, e.g. Robertson & Gemmell, 2004; Abdelkrim, et al., 2005) and it is often harder to define if treatment of the whole island or only part of the island is required. Recently, six successful bird eradications in the Seychelles were implemented (Bunbury, et al., 2019) adding to the global knowledge pool for planning and implementing invasive bird eradications. We are aware of only two other attempts to eradicate invasive house sparrows from island habitats, an unsuccessful attempt on Round Island in Mauritius (Bednarczuk, et al., 2010), and a successful attempt of a restricted range population on Mahe in the Seychelles, where repeated invasions (due to international ship traffic) are treated on an ongoing basis (Beaver & Mougal, 2009).

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REFERENCES

Abdelkrim, J., Pascal, M. and Samadi, S. (2005). 'Island colonization and founder effects: The invasion of the Guadeloupe Islands by ship rats (*Rattus rattus*)'. *Molecular Ecology* 14: 2923–2931.

Anderson, T.R. (2006). *Biology of the Ubiquitous House Sparrow: From Genes to Populations,* Oxford: Oxford University Press.

Arroyo, M.T.K. (1999). 'The Juan Fernández Islands'. In*:* R.A. Mittermeier, N. Myers, G.P. Robles and C.G. Mittermeier (eds.) *Hotspots: Earth's Biologically Richest and Most Endangered Terrestrial Ecoregions,* p. 173. Mexico City: CEMEX.

AVMA (2007). 'AVMA Guidelines on Euthanasia'. *Journal of the American Veterinary Medical Association,* p. 14. American Veterinary Medical Association.

Baker, J., Harvey, K.J. and French, K. (2014). 'Threats from introduced birds to native birds'. *Emu* 114: 1–12.

Beaver, K. and Mougal, J. (2009). *Review of IAS Control and Eradication Programmes in Seychelles*. Seychelles: Plant Conservation Action Group. Unpublished report. <http://www.pcusey.sc/index.php/component/remository/func-startdown/107/?Itemid=>

Bednarczuk, E., Feare, C., Lovibond, S., Tatayah, V. and Jones, C. (2010). 'Attempted eradication of house sparrows *Passer domesticus* from Round Island (Mauritius), Indian Ocean'. *Conservation Evidence* 7: 75–86.

Bibby, C.J., Burgess, N.D., Hill, D.A., and Mustoe, S.H. (2000). *Bird Census Techniques*. London: Academic Press of Elsevier.

Bourne, W.R.P., Brooke, M.d.L., Clark, G.S. and Stone, T. (1992). 'Wildlife conservation problems in the Juan Fernández Archipelago, Chile'. *Oryx* 26: 43–51.

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L. and Thomas, L. (2001). *Introduction to Distance Sampling Estimating Abundance of Biological Populations*. Oxford, UK: Oxford University Press.

Bunbury, N., Haverson, P., Page, N., Agricole, J., Angell, G., Banville, P., Constance, A., Friedlander, J., Leite, L., Mahoune, T., Melton-Durup, E., Moumou, J., Raines, K., van de Crommenacker, J. and Fleischer-Dogley, F. (2019) 'Five eradications, three new species, three islands: overview, insights and recommendations from invasive bird eradications in the Seychelles'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 282–288. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Clark, J.P. and Hygnstrom, S.E. (1994). 'Crowned Sparrows'. In*:* S. Hyngstrom, R.M. Timm and G. Larson (eds.) *Prevention and Control of Wildlife Damage,* pp. E97-E100. Lincoln, NE: University of Nebraska Cooperative Extension Service.

Clergeau, P., Levesque, A. and Lorvelec, O. (2004). 'The precautionary principle and biological invasion: The case of the house sparrow on the Lesser Antilles'. *International Journal of Pest Management* 50: 83–89.

DIISE. (2016). *The Database of Island Invasive Species Eradications.*<http://diise.islandconservation.org>. Accessed 16 February, 2016.

Fagerstone, K.A. (2007). 'Mitigating Impacts of Terrestrial Invasive Species'. In*:* D. Pimentel (ed.) *Encyclopedia of Pest Management, Volume II,* pp. 347–349. Boca Raton: CRC Press.

Fisher, P., Booth, L. and Campion, M. (2012). *Review of Avicides for Use in Eradication of Introduced Sparrows and Pigeons from the Juan Fernández Islands, Chile*. Unpublished report. Lincoln: Landcare Research.

Fitzwater, W. (1981). 'Outwitting the House Sparrow [*Passer domesticus* (Linnaeus)]'. *Great Plains Wildlife Damage Control Workshop Proceedings*, pp. 244–251. Lincoln, NE: University of Nebraska.

Fitzwater, W. (1994). 'House Sparrows'. In*:* S. Hyngstrom, R.M. Timm and G. Larson (eds.) *Prevention and Control of Wildlife Damage,* pp. E101-E108. Lincoln, NE: University of Nebraska Cooperative Extension Service.

Glen, A.S., Atkinson, R., Campbell, K.J., Hagen, E., Holmes, N.D., Keitt, B.S., Parkes, J.P., Saunders, A., Sawyer, J. and Torres, H. (2013). 'Eradicating multiple invasive species on inhabited islands: The next big step in island restoration?' *Biological Invasions* 15: 2589–2603.

Gowaty, P.A. (1984). 'House sparrows kill eastern bluebirds'. *Journal of Field Ornithology* 55: 378–380.

Greimler, J., Lopez S., P., Stuessy, T.F. and Dirnböck, T. (2002). 'The vegetation of Robinson Crusoe Island (Isla Masatierra), Juan Fernández Archipelago, Chile'. *Pacific Science* 55: 263–284.

Hahn, I. and Römer, U. (2002). 'Threatened avifauna of the Juan Fernández Archipelago, Chile: The impact of introduced mammals and conservation priorities'. *Cotinga* 17: 66–72.

Hahn, I., Römer, U. and Schlatter, R. (2005). 'Distribution, habitat use, and abundance patterns of landbird communities on the Juan Fernández Islands, Chile'. *Ornitología Neotropical* 16: 371–385.

Hahn, I., Römer, U. and Schlatter, R. (2006). 'Population numbers and status of land birds of the Juan Fernandez Archipelago, Chile: (Aves: Falconiformes, Columbiformes, Strigiformes, Caprimulgiformes, Passeriformes)'. *Senckenbergiana Biologica* 86: 109–125.

Hahn, I., Römer, U., Vergara, P. and Walter, H. (2009). 'Biogeography, diversity, and conservation of the birds of the Juan Fernández Islands, Chile'. *Vertebrate Zoology* 59: 103–114.

IUCN. (2000). 'IUCN Guidelines for the Prevention of Biodiversity Loss Caused by Alien Invasive Species'. < http://www.issg.org/pdf/guidelines_iucn.pdf>. Accessed 10 May, 2017.

MacGregor-Fors, I., Morales-Perez, L., Quesada, J. and Schondube, J.E. (2010). 'Relationship between the presence of house sparrows (*Passer domesticus*) and neotropical bird community structure and diversity'. *Biological Invasions* 12: 87–96.

Martin-Albarracin, V.L., Amico, G.C., Simberloff, D. and Nuñez, M.A. (2015). 'Impact of non-native birds on native ecosystems: A global analysis'. *PLOS ONE* 10: e0143070.

McGillivray, W.B. (1980). 'Communal nesting in the house sparrow'. *Journal of Field Ornithology* 51: 371–372.

McGregor, D.B. and McGregor, B.A. (2008). 'Eliminating an avian pest (house sparrow *Passer domesticus*) population: The role of trapping at a homestead scale'. *Victorian Naturalist* 125: 4–10.

McKown, M. (2013). *Acoustic Monitoring of House Sparrows Eradication Trials on Isla Robinson Crusoe, Chile Final Report*. Unpublished report. Santa Cruz, California: Conservation Metrics.

Ministerio del Medioambiente (2017). *Plan de Acción del Archipiélago Juan Fernández para la Prevención, Control y/o Erradicación de Especies Exóticas Invasoras*. Santiago, Chile: Imprenta Grafic Suisse.

Morrison, S.A., Macdonald, N., Walker, K., Lozier, L. and Shaw, M.R. (2007). 'Facing the dilemma at eradication's end: Uncertainty of absence and the Lazarus effect'. *Frontiers in Ecology and the Environment* 5: 271–276.

Ramsey, D.S.L., Parks, J.P., Will, D., Hanson, C.C., and Campbell, K.J. (2011). 'Quantifying the success of feral cat eradication, San Nicolas Island, California'. *New Zealand Journal of Ecology* 35: 163–173.

Robertson, B.C. and Gemmell, N.J. (2004). 'Defining eradication units to control invasive pests'. *Ecology* 41: 1042–1048.

Samaniego-Herrera, A., Anderson, D.P., Parkes, J.P. and Aguirre-Muñoz, A. (2013). 'Rapid assessment of rat eradication after aerial baiting'. *Journal of Applied Ecology* 50: 1415–1421.

Sanders, R.W., Stuessy, T.F. and Marticorena, C. (1982). 'Recent changes in the flora of the Juan Fernandez Islands, Chile'. *Taxon* 31: 284–289.

Saunders, A., Glen, A., Campbell, K., Atkinson, R., Sawyer, J., Hagen, E. and Torres, H. (2011). *Feasibility of Managing Invasive Species in the Juan Fernández Archipelago, Chile*. Unpublished report. Lincoln, New Zealand: Invasive Species International.

Sharp, T. and Saunders, G. (2005). 'BIR002: Trapping of Pest Birds'. In*:* Sharp, T. and Saunders, G. (eds.) *Humane Pest Animal Control: Codes of Practice and Standard Operating Procedures,* p. 11. New South Wales, Australia: NSW Department of Primary Industries.

Strubbe, D., Shwartz, A. and Chiron, F. (2011). 'Concerns regarding the scientific evidence informing impact risk assessment and management recommendations for invasive birds'. *Biological Conservation* 144: 2112–2118.

Predation pressures on sooty terns by cats, rats and common mynas on Ascension Island in the South Atlantic

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Abstract Despite the presence of invasive black rats (*Rattus rattus*), common mynas (*Acridotheres tristis*), and feral domestic cats (*Felis catus*), sooty terns (*Onychoprion fuscatus*) breed in large numbers on Ascension Island in the tropical South Atlantic Ocean. These introduced predators impact the terns by destroying eggs or interrupting incubation (mynas), eating eggs (mynas and rats), eating chicks (rats and cats), or eating adults (cats). Between 1990 and 2015, 26 censuses of sooty terns and five of mynas were completed and myna predation was monitored on 10 occasions. Rat relative abundance indices were determined through trapping around the tern colonies and rat predation was monitored by counting chick carcasses. Cat predation was quantified by recording freshly killed terns. Prior to their eradication in 2003, cats had the greatest impact on sooty terns and were depredating 5,800 adults and 3,600 near-fledging chicks (equivalent to the loss of 71,000 eggs) each breeding season. We estimated that 26,000 sooty tern eggs (13% of all those laid) were depredated by approximately 1,000 mynas. Rats were not known to depredate sooty terns prior to cat eradication but in 2005, 131 of 596 ringed (monitored) chicks (22%) were depredated by rats. In 2009 chick carcass density was 0.16 per m^2. Predation by rats hugely increased in the absence of cats and was the equivalent of 69,000 eggs. Care is needed when applying our findings to seabirds globally. The scarcity of alternative food sources and seasonally high density of easily available prey in the sooty tern colony may have magnified predation by cats, rats and mynas.

Keywords: Non-native species, population size, predation rate, United Kingdom Overseas Territory (UKOT)

INTRODUCTION

Comparative studies of global declines in faunal biodiversity have concluded that harvesting, habitat loss and introduced invasive species are leading causes (see refs in Young, et al., 2016). Of extinction events for which causes have been investigated, 54% have been attributed in part to invasive species (Clavero & Garcia-Berthou, 2005). Globally, terrestrial invertebrate invaders have reduced faunal diversity by 29% (Cameron, et al., 2016). Lowe, et al. (2000) compared the severity of alien species on animal and plant diversity by compiling a list entitled "100 of the world's worst invasive alien species". The list includes invasive predators that are commensal with man; they pose major threats to seabirds and they persist following anthropogenic introduction to 90% of all island archipelagos (Towns, et al., 2006). Global seabird population size has declined by 70% between 1950 and 2010 (Paleczny, et al., 2015) with introduced commensal predators being one of the major proposed causes of such declines (Moors & Atkinson, 1984).

Of introduced predators, feral domestic cats (*Felis catus*) (Medina, et al., 2011) and black rats (*Rattus rattus*) (Jones, et al., 2008) inflict the most severe impacts on native avifauna. Common mynas (*Acridotheres tristis*) (hereafter referred to as 'mynas') also have significant negative impacts on native avifauna through competition for food and nest sites (Grarock, et al., 2012). When cats, rats and mynas invade islands on which seabirds are breeding, cats have a direct effect on the size of the seabird population through predation of adults (van Aarde, 1983) while rats and mynas have a less immediate, but a more indirect, effect through predation of eggs or chicks (Jones, et al., 2008). Therefore, rats and mynas reduce breeding success and inflict downstream impacts on seabird demography through reduced recruitment to the breeding population (Harper & Bunbury, 2015). The direct impacts of cats on breeding seabirds are more readily observed than the indirect effects of rats and mynas that are more difficult to quantify because rat and myna predation is less obvious and is confounded by rats scavenging on chicks that have died from causes other than direct predation (e.g. starvation).

Alien invasive predators are the potential cause of precipitous declines in the population size of breeding sooty terns (*Onychoprion fuscatus*) on Ascension Island during the 20th century (Hughes, et al., 2017a). Sooty terns are the most numerous avian species in tropical waters and Ascension Island accommodates the largest breeding population in the Atlantic (Schreiber, et al., 2002). Three of the world's 'worst' invasive predators are found in the seabird colonies on Ascension Island. Black rats probably arrived when HMS Roebuck was abandoned close to the island in 1701 (Ashmole & Ashmole, 2000), and by 1725 rats were so numerous that a castaway on the island lived in fear of being eaten alive (Ritsema, 2006). In 1815 domestic cats were introduced to control the rat population. Mynas were introduced in the 1880s to reduce damage to crops by black cutworms (*Agrotis ipsilon*) (Duffey, 1964). Common mynas in their home range (i.e. India) are regarded as a beneficial species (BirdLife International, 2015) because typically more than 80% of their food mass comprises insects regarded as pests (e.g. cutworms – larvae of Noctuidae). Since the arrival of these invasive species on the island, the once vast colonies of seabirds, estimated to contain > 10 million birds (Ashmole & Ashmole, 2000), have dwindled to less than half a million birds (Bell & Ashmole, 1995). Of the 11 seabird species that breed on Ascension Island, only sooty terns now breed in large numbers on the main island. Numerically, 97% of all seabirds breeding on the main island are sooty terns (Hughes, 2014). Remnant populations of other seabird species nest on cat- and rat-free offshore stacks and Boatswainbird Islet (Ratcliffe, et al., 2009).

In 1958 and 1959, cats were the only non-native predatory species known to depredate seabirds and an aspiration for their eradication was conceived (Ashmole, 1963). During a feasibility study for cat eradication in 1992, rats were also considered a major threat to seabirds (Ashmole, et al., 1992) but the threat that mynas posed was not recognised at that time. More recently, in the Seychelles, Feare, et al. (2015) recorded mynas inflicting intense predation on seabird eggs. On Ascension Island cats were eradicated in 2003 (Bell & Boyle, 2004) and

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rat control measures (Pickup, 1999) were implemented. The eradication of apex predators is generally associated with an increase in the abundance of smaller predators with this trophic interaction referred to as 'mesopredator release' (Prugh, et al., 2009). However, Russell, et al. (2009) modelled the effects of mesopredator release and concluded that the negative impact of more mesopredators is outweighed by the benefit of apex predator removal, allowing recovery of prey populations. If we apply their conclusions to Ascension Island then cat eradication should have resulted in an increase in the population size of sooty terns but, to date, no such effect has been detected (Hughes, et al., 2017a).

Here, we have collated data from published outputs and from a 25-year Army Ornithological Society (AOS) dataset on introduced species to calculate the relative impacts of cat, rat and myna predation on the sooty tern breeding population.

METHODS

Study area and period

Ascension (07º57′S, 14º24′W, 97 km²) is one of the volcanic islands that make up the UK Overseas Territory (UKOT) of St Helena, Ascension and Tristan da Cunha, and is isolated in the tropical South Atlantic Ocean midway between South America and Africa (Fig. 1; Hughes, et al., 2010). Its nearest neighbour is the island of St Helena some 1,300 km to the south-east. The territory is an Important Bird Area (IBA reference number SH009; BirdLife International, 2017). More than half of its surface consists of cinder plains, ash cones and basaltic lava flows. The average annual rainfall is 144.0 mm (Anon., 1998) and plant species richness on the plain is < 11 species (Duffey, 1964). The dry coastal plain is the traditional nesting site for seabirds and sooty terns nest at Mars Bay and Waterside in the south-west corner of the island (Fig. 1).

Fieldwork lasted two weeks per breeding season and was timed to coincide with the peak of the sooty tern breeding season (see further details in Reynolds, et al., 2014). Time in the field amounted to 1,691 person-days.

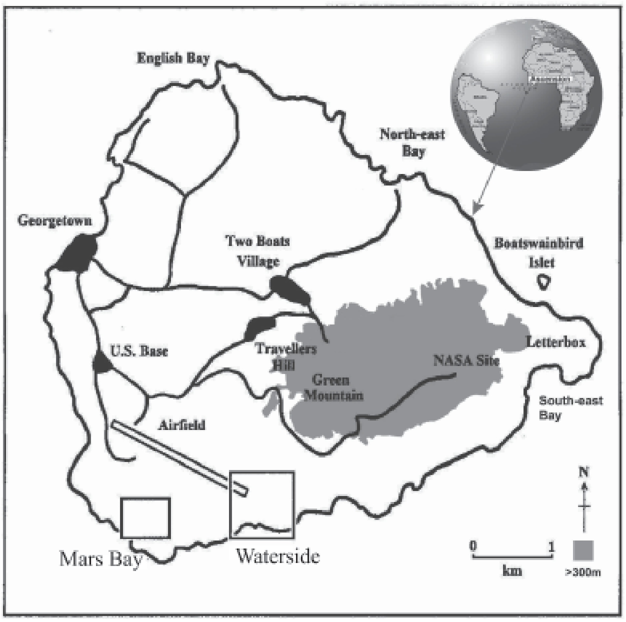

Fig. 1 Map of Ascension Island in the South Atlantic showing sites of human habitation and ground above 300 m (shaded). Sooty terns nest in the south-west corner of the island in the areas marked as 'Mars Bay' and 'Waterside'.

Prey population size

Sooty terns, the primary avian prey species of rats, cats and mynas, are a migratory species and are absent from Ascension Island for approximately three in every 9.6 months that constitute the sub-annual cycle of the species (Reynolds, et al., 2014). Modal clutch size of sooty terns is one (Schreiber, et al., 2002). The population was censused on 26 occasions between 1990 and 2015. We calculated the number of eggs laid by measuring the area of the two breeding colonies using conventional land survey techniques and determined egg density by counting eggs in quadrats (see full details in Hughes, et al., 2008).

Myna population size and their predation pressure on sooty terns

Censuses of the myna population were obtained from a consolidation of counts in 1994, 2004, 2005, 2006 and 2015 and included counts of birds feeding on the two rubbish tips, in 116 1-km grid squares covering the whole island, and at night-roosts.

Rates of predation were estimated by marking focal sooty tern eggs and following their fates. In each sooty tern breeding season, egg predation by mynas was measured for approximately seven days (i.e. for 25% of the incubation period of 28.8 days; Ashmole, 1963) and mean egg failure rates for the core and periphery of the colony (Hughes, et al., 2008) were calculated using the Mayfield method (Johnson & Shaffer, 1990). Causes of egg failure were categorised according to egg damage likely caused by mynas: 'consumption' was defined as the opening of a viable egg and feeding on some (usually < 10%) of the contents, and 'puncturing' was defined as the creation of a single small hole that destroyed egg integrity. The ratio of consumed:punctured sooty tern eggs was obtained from quadrat counts of depredated eggs. To establish causation of egg desertion, sets of focal eggs that contained deserted eggs were separated into two categories: those containing eggs consumed or punctured by mynas, and those that did not. We had previously found that the apparent association between these egg fates was significant (see full details in Hughes, et al., 2017b).

Cat population size and their predation pressure on sooty terns

On Ascension Island the cat population size in 1958 was estimated to be in the hundreds (Ashmole, 1963). Of the 1,100 feral cats that were removed from the island in the eradication programme of 2002, approximately 50 were removed from the tern colonies (Bell & Boyle, 2004).

Predation of adult sooty terns was monitored by removing all corpses of terns from the breeding area and then re-visiting the colonies to record the number of freshly killed birds. The mortality data gathered over two weeks may sometimes under-estimate the level of predation. Ashmole (1963) found that towards the end of each season cats began to take large chicks as well as adults. To compensate for this unknown level of chick predation, cats were assumed to take equal numbers of adults and chicks for 110 days (i.e. the period when some adults incubate while others feed chicks close to fledging; see full details in Hughes, et al., 2008). Because cats have been observed consuming seabird eggs elsewhere, albeit on rare occasions (Plantinga, et al., 2011), we also assessed this source of egg loss by inspecting cat middens for cat-predated eggs.

Rat population size and their predation pressure on sooty terns

The size of the rat population on Ascension Island has not been estimated but anecdotal data indicate that it has been (and remains) large. For example, 70,148 rats were killed between 1878 and 1887 (Hart-Davis, 1972). Relative abundance of rats in the tern colonies was estimated using a simple index calculated as the number of rat captures per 100 trap-nights (C/100TN) corrected for traps tripped (after Cunningham & Moors, 1983). During field seasons 50 'Victor' break-back rat traps baited with peanut butter and cornflakes were set out in pairs along the edge of both tern breeding colonies. Nest density was too high to allow traps to be set within the colonies without significantly disturbing breeding birds. Trapping occurred over two consecutive nights.

We studied the rate of chick predation by rats by counting chick carcasses (Towns, et al., 2006). We eliminated the possibility that starvation was the ultimate cause of chick mortality by recording the muscle score and body mass of live chicks in the same parts of the colonies as carcass surveys. The shape of the pectoral muscles was scored between 0 and 2 according to the prominence of the keel as described in Gosler (1991). A muscle score of 0 on this scale is indicative of low body condition most likely caused by malnourishment. Body mass of live chicks was recorded to the nearest 1 g with a Pesola spring balance. Chicks aged 28–30 days that were underweight weighed approximately 80 g and those that were in higher condition were > 150 g (Ashmole, 1963).

Prior to cat eradication in 2002 we found two cavities in rocks on the perimeter of the tern colonies that contained many broken sooty tern eggs but only later did we attribute the find to rat predation. Rats will roll eggs away from avian nests to a place of safety where they can open them (Zarzoso-Lacoste, et al., 2011). After cat eradication, we studied the rate of egg predation by rats by marking focal eggs and recording their losses. The rate of egg losses to rats was calculated as for that to mynas. We calculated the level of egg predation by rats prior to cat eradication by scaling up our findings from the above focal study. We used rat indices to generate relative rat abundance estimates before and after cat eradication.

Comparison of the three predation pressures

To evaluate the impact of chick and egg losses on the size of the breeding population of sooty terns, ratios of adults to chicks, and adults to eggs were required. In other words, on average, how many eggs need to hatch, and thus how many chicks need to survive until recruitment, to replace one adult in the breeding population? Furthermore, cats depredate near-fledging chicks while rats take half-grown chicks and thus we also required a ratio of eggs to both cat- and rat-depredated chicks.

The ratio of near-fledging chicks to adults was obtained from demographic data and estimates of adult and juvenile survival rates were calculated from ringing-re-capture data of adults and near-fledging chicks that were ringed during the same breeding seasons and re-captured in subsequent seasons (see further details in Reynolds, et al., 2014). Adult and juvenile survival rates, age at first breeding and mean age of birds in the breeding population were determined each breeding season by the re-capture from each cohort of adults and new recruits, and a mean with a 95% confidence limit (CL) calculated using the program MARK (White & Burnham, 1999).

The ratio of eggs to near-fledging chicks (i.e. those depredated by cats) was calculated from density counts of eggs and near-fledging chicks in quadrats (Bibby, et al., 2000; Schreiber & Burger, 2002). The ratio of eggs to half-grown chicks (i.e. those depredated by rats) was calculated by taking the average of near-fledging chick survival (see above) and nestling survival rates. The age at which nestlings leave the nest was approximately five days (Schreiber, et al., 2002). We calculated nestling survival rate for the five days by applying a hatchability rate of 0.91 (i.e. the number of eggs that hatched at the end of the incubation period; after Koenig, 1982) and a predation rate from Ascension frigatebirds (*Fregata aquila*) of 0.98 (i.e. the number of nestlings that escape frigatebird predation; BJH, unpubl. data) to the incubation success rate.

RESULTS

Sooty tern population size

Each season between 1990 and 2015, sooty terns laid on average 180,000 ± 8,000 (1 standard error [SE]) eggs (range: 70,000–270,000 eggs, n = 26 censuses). The mean number of nestlings in the tern colony each season was 94,000 ± 14,000 (n = 12 breeding seasons). The mean size of the breeding population was 360,000 ± 14,000 (95% CL) birds (Fig. 2).

Myna population size and their predation pressure on sooty terns

Between 1992 and 2015 the mean size of the myna population was 935 ± 265 (95% CL) birds (Fig. 3a). We found no evidence to suggest that mynas killed tern chicks. Mynas were recorded every field season in the tern colonies. Between 2000 and 2008 we monitored 1,238 eggs (935 on the periphery and 303 in the core). Of the 331 nest failures at the periphery of the colonies, 87 (26.3%) failed as a direct result of mynas. We calculated the mean egg failure rate at the periphery of the colonies as being 0.35 ± 0.07 (± 1 SE) eggs per season (n = 10 breeding seasons). The core of each colony appeared largely free from egg predation by mynas. The mean rate of egg loss to mynas in the two colonies was 0.19 ± 0.04 eggs per pair of terns (range: 0.02–0.37 eggs per pair, n = 1,238 breeding pairs over 10 breeding seasons). The ratio of consumed:punctured sooty tern eggs was 1:1.83 (n > 500 eggs in five sample quadrats across three breeding seasons). In summary, of all sooty tern eggs lost to mynas, 21% were consumed, 39% were punctured and 40% were deserted. We calculated that sooty tern mean egg losses to myna predation per season amounted to 26,000 ± 12,000 eggs (range: 4,000–50,000 eggs) that represented an average of 13% of all eggs laid (n = 10 breeding seasons).

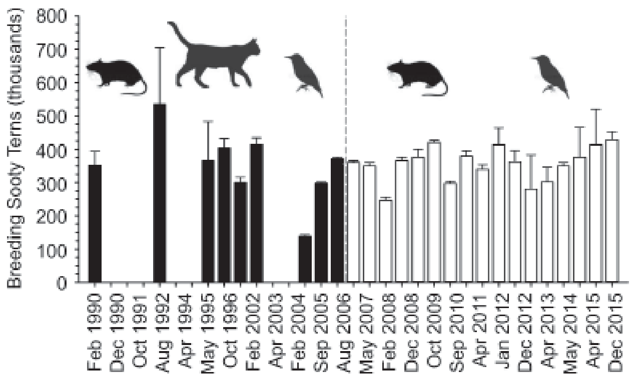

Fig. 2 Estimated size of the sooty tern breeding population (mean + 95% confidence limits) on Ascension Island between 1990 and 2015. Filled columns are censuses carried out during the cat-rat-myna (three) predator regime while open columns are those conducted during the rat-myna (dual) predator regime. Note that the sub-annual breeding cycle results in birds breeding twice in 1996, 2004, 2008, 2012 and 2015.

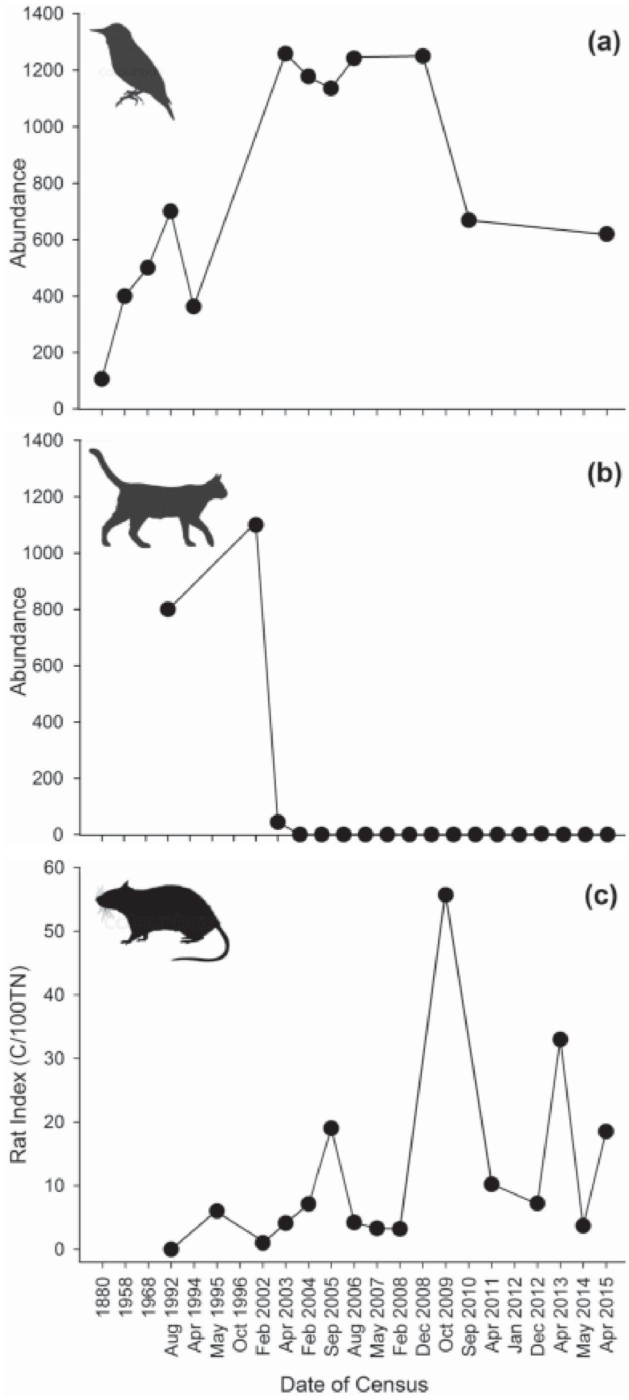

Fig. 3 Population sizes of the three introduced species that prey on various life stages of sooty terns breeding on Ascension Island in the South Atlantic: (a) common mynas, (b) domestic cats, and (c) black rats. Note that in (a) and (b) population estimates are based upon whole-island counts while (c) provides a relative abundance index calculated as the mean number of rats captured (C) per 100 trap-nights (TN) at both colonies (see further details in the Methods).

Cat population size and their predation pressure on sooty terns

We were unable to quantify cat numbers in the sooty tern colonies. Estimates of the size of the feral cat population across the whole island are shown in Fig. 3b. The number of adult sooty terns killed by cats from collection of corpses in 1990, 1992 and 1994 amounted to 2,996, 340 and 310, respectively. The mean number of adults killed by cats in the two colonies was 29 per night (n = 32 nights over three breeding seasons). Towards the end of the sooty tern breeding season cats were killing near-fledging chicks as well as adults. If cat predation continued at the same intensity in the second half of the season as in the first, the overall percentage of the adult population depredated by cats would have been 1.8% (or 5,800 birds on average, n = 3 breeding seasons). Predation of chicks was not monitored but we estimated that the overall percentage of chicks that were depredated or died of starvation because a parent was killed by cats, was 3,600 chicks (i.e. 29 cat kills per night for the four months that chicks were in the colony, yielding a total of 3,600 chicks, 3.8% of the chick population of 94,000). We found no evidence that feral cats were taking any sooty tern eggs.

Rat population size and their predation pressure on sooty terns

During 473 days of fieldwork prior to the eradication of cats, no rat predation of tern chicks was observed. Between 1992 and 2002 the mean relative abundance of rats pre-cat eradication on the dry coastal plain close to the two tern colonies was 1.3 ± 1.0 (± 1 SE) C/100TN (range: 0–6.0 C/100TN, n = 6 trap-lines over three breeding seasons). Between 2005 and 2015 after the cat eradication the mean relative abundance of rats was 15.2 ± 3.8 C/100TN (range: 0–74.5 C/100TN, n = 25 trap-lines over 12 breeding seasons) (Fig. 3c).

Carcasses of chicks depredated by rats were first observed in 2005 when 131 of 596 ringed chicks (22.0%) were depredated. In 2009 mean carcass density in quadrats was 0.16 (range: 0–0.9 per m^2, n = 68 quadrats). The area of the colony was 12.21 ha and thus it contained an estimated 19,500 ± 27,000 carcasses (20%) of the chick population. We found no evidence of mass starvation as live chicks had a mean muscle score of 1.05 ± 0.31 (range: 0–2, n = 998 chicks) and a mean body mass of 157.5 ± 29.2 g (range: 54.8–220.0 g, n = 946 chicks). In 2005 and 2009 the mean number of chicks depredated by rats was 20,000 (21% of the chick population).

During nine sooty tern breeding seasons between 2003 and 2012, we monitored 1,067 single egg clutches (792 on the periphery and 275 in the core) for rat predation. Of the 327 nest failures, 314 were on the periphery and 13 were in the core of the colonies. Of the 327 that failed, 51 (15.6%) were missing eggs and these were attributed to rat predation. The mean rate of egg loss to rats was 0.17 ± 0.06 (± 1 SE) eggs per pair of terns (range: 0.00–0.49 eggs per pair, n = 1,067 breeding pairs over nine breeding seasons). Assuming rats only depredated eggs at the periphery of the colonies as so few eggs failed in the core, the overall percentage of the eggs depredated in the tern colony by rats was 4.8% representing an egg total of 9,000 (n = 9 breeding seasons).

From a comparison of mean relative abundances of rats pre- (1.3 C/100TN, n = 3 breeding seasons) and post-cat eradication (15.2 C/100TN, n = 12 breeding seasons), we estimated that the rat population was only 8.6% as large prior to, compared with after, the cat eradication. We also estimated that rats depredated 800 eggs per season prior to cat eradication.

Comparison of the three predation pressures

A summary of the comparative predation pressures in terms of egg losses (i.e. the lowest currency to represent all tern life stages) is provided in Fig. 4. We generated ratios to eggs laid of survival estimates at various life stages using life-history data in Hughes (2014). The ratios generated were eggs preyed upon by mynas (1:1); chicks succumbing to rat predation (2.99:1); chicks preyed upon by cats (4.13:1); and adults preyed upon by cats (9.67:1).

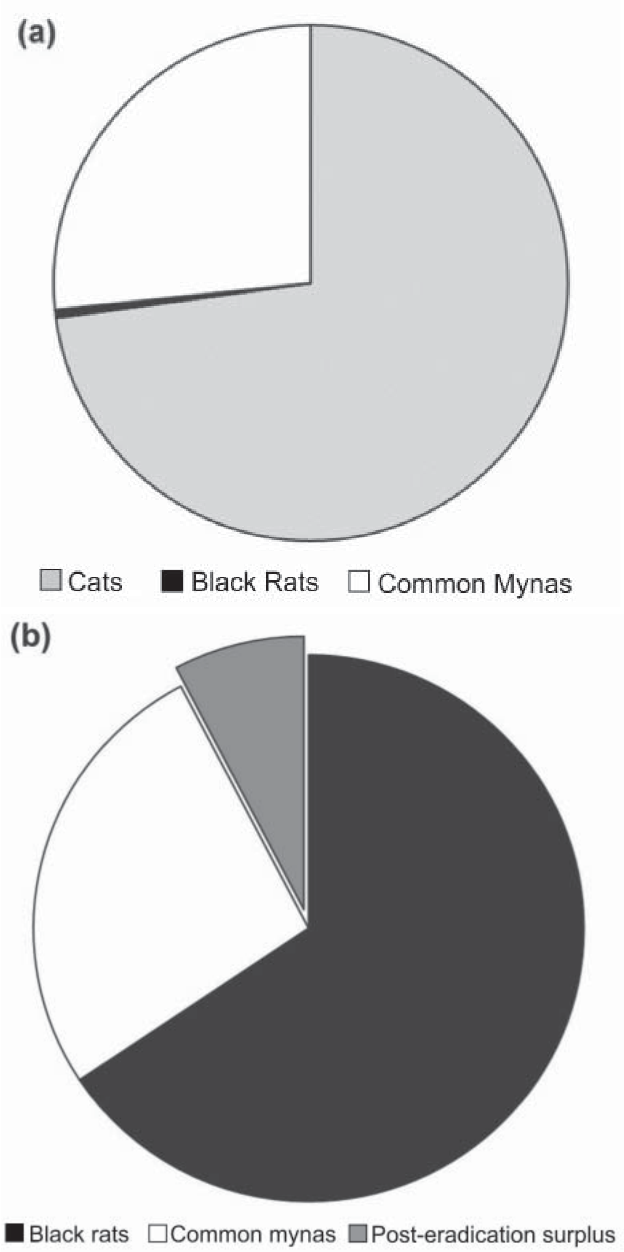

Fig. 4 (a) Pre- and (b) post-cat eradication losses of sooty tern egg equivalents (i.e. eggs and post-hatching chick and adult life stages) to three introduced predator species on Ascension Island in the South Atlantic. The average number of eggs laid was 180,000. Note that in (b) there are only two predators contributing to losses of egg equivalents while the third category represents the benefit to the tern population of cat eradication, equating to 2,000 eggs (see further details in the Results).

The mean number of eggs lost to myna predation per sooty tern breeding season was 26,000. We calculated that losses of chicks to rats translated into the 'loss' of 60,000 eggs (i.e. 20,000 chicks × 2.99). Rats also directly removed 9,000 eggs from the breeding colony. The sum of rat predation translated into the 'loss' of 69,000 eggs. Cats depredated 5,800 adults and 3,600 near-fledging chicks translating into egg losses of 56,000 (i.e. 5,800 adults × 9.67) and 15,000 (i.e. 3,600 near-fledging chicks × 4.13), respectively. Therefore, the sum of cat predation translated into the 'loss' of 71,000 eggs.

DISCUSSION

We compared carcass counts of adults killed by cats with those of chicks killed by rats to assess their relative impacts on the tern population. Carcass density of chicks killed by rats was likely to provide an under-estimate of chick losses as decomposition of corpses was rapid and rough ground made it easy to overlook them. Carcass counts of adults and near-fledging chicks killed by cats (and adjusted by the adult:chick ratio), and those of chicks solely killed by rats, were similar to each other, varying by 18%. If we take into account that rats also depredated 9,000 eggs then the variation between rat and cat predation is just 3%. Of the three sources of predation on the tern population, cats had the greatest impact on the tern population but following their eradication, rats replaced them as the primary source of predation pressure (Fig. 4).

The third source of predation on the island was mynas that depredated 26,000 eggs every sooty tern breeding season but their overall impact on the population size of sooty terns was less than half that of cats or rats. Mynas depredated more tern eggs than did rats (i.e. 26,000 *versus* 9,000) and very many more than did cats. Bell and Boyle (2004) found egg remains in stomachs of one of five cats culled close to the tern colony. We found no evidence that mynas depredated chicks or adults. Mynas depredated more tern eggs than rats or cats depredated chicks and before cat eradication (i.e. pre-2002) mynas had a greater detrimental impact on the size of the breeding tern population than did rats. There were large variations between sooty tern breeding seasons in the relative abundance of rats in the tern colonies (Fig. 3c) and in the extent of egg losses to mynas (i.e. 0.02–0.37 eggs per pair) suggesting that sooty terns were not the main driver of the population dynamics of these two omnivorous predators (Towns, et al., 2006).

Other comparative studies

A meta-analysis by Baker, et al. (2013) of threats to native avian species posed by introduced ones concluded that introduced invasive avian species are not a major threat. However, we found that mynas posed a major threat to native sooty terns on the island (Hughes, et al., 2017b). For every egg that mynas consumed, they punctured or caused desertion of four others. The only quantitative comparative study of seabird egg predation by mynas was of 350 wedge-tailed shearwaters (*Puffinis pacificus cuneatus*) on Hawaii where mynas punctured 74 (21%) of all eggs laid during one season (Byrd, 1979).

Ashmole (1963) estimated that on Ascension Island cats were killing approximately 10,000–20,000 sooty tern adults (i.e. 0.5 to 1.0% of the adult population) and up to 40% of chicks in 1958 and 1959. On Juan de Nova Island in the Mozambique Channel in the western Indian Ocean where predator/prey constituent members were similar to those on Ascension Island, Peck, et al. (2008) found that cats were killing 2,205 sooty terns per week (0.1% of the breeding population).

Prior to cat eradication, we saw no live rats in the tern colony and we did not suspect any rat predation of tern life stages. Similarly, Ashmole (1963) saw no such incidents of predation by rats during numerous day and night visits to the tern colonies in 1958 and 1959. On Juan de Nova Island where black rats co-exist with cats, and both depredate sooty terns, Ringler, et al. (2015) reported that losses of sooty terns to rats were relatively low. On Ascension Island losses of sooty terns to rats increased dramatically when cats were eradicated (Fig. 4). The severity of the predation post-eradication was similar to that found by Jones, et al. (2008) in their meta-analysis of the severity of rat predation.

Mesopredation

We found that the intensity of predation by rats varied depending on whether rats were the apex predator. Under the cat-rat-myna predator regime, rats exerted the least predation pressure on the tern population (Fig. 4a) but following apex predator (cat) removal, they exerted the greater predation pressure in the dual predator regime (Fig. 4b). Cats were eradicated from the tern colony in 2002 (Bell & Boyle, 2004) and the rat population increased seven-fold in size following their eradication as determined from the relative abundance index (Fig. 3c). The eradication of cats is seen as particularly beneficial to seabirds (Nogales, et al., 2013) but, to the best of our knowledge, this only applies to islands without rats in the first place (e.g. Natividad Island, Marion Island in the sub-Antarctic and Baker Island in the Pacific). We found clear evidence that when black rats are 'released' by apex predator removal the size of the rat population increased and rats started to depredate tern chicks. Rats as apex predators exerted a predation pressure on terns that was 97% of that in the regime of cats and rats. Our findings are at odds with those of Ringler, et al. (2015) who predicted that cat eradication would be beneficial to sooty terns. They also oppose McCreless, et al. (2016) who found that the potential for extirpation of seabird populations was greater in the twin predator regime of cats and rats and they also disagree with Ratcliffe, et al. (2009) who reported that on Ascension Island five seabird species had re-colonized the mainland following the eradication of cats. There are three possible explanations for this disparity: 1) despite major changes in predator population sizes (Fig. 3), there has been little fluctuation in that of breeding sooty terns on the island (Fig. 2) which suggests that predation may not be the primary driver of the tern's population size; 2) a change in the habitat on the tern colonies on Ascension Island occurred concurrently with cat eradication which rats, with their catholic diets, took advantage of by switching to alternative food sources such as seeds of the invasive plant mesquite (*Prosopis juliflora*) (Pickup, 1999); and 3) the sub-annual breeding cycle of sooty terns on Ascension Island may provide rats with more opportunities to breed than if sooty terns were breeding annually as they do elsewhere in their range (Reynolds, et al., 2014).

CONCLUSIONS

Care is needed when applying our findings related to predation pressures on Ascension Island sooty terns to other seabird species on the island and to other places in the world. When sooty terns are present, the super-abundance of prey as represented by eggs, chicks and adults may magnify predation pressures. As far as we are aware, our study is the first to provide a comparison of predation pressures by cats, rats and mynas on seabirds. Such empirical evidence of invasive species' impacts on native avifauna is critical for the prioritization of management options directed towards introduced species (Jeschke, et al., 2014; McCreless, et al., 2016). Here, we present strong evidence that mynas can be major egg predators of seabirds. We have quantified changes in predation pressures resulting from the eradication of cats and we have highlighted that rats in the absence of cats have impacted upon breeding success of sooty terns sufficiently to bring into serious question the benefits of cat eradication to the recovery of the sooty tern breeding population on Ascension Island. Conversely, pressures on sooty terns from predators have declined by 3% following the removal of cats. How rats have largely replaced the predation pressure posed by cats following their removal and why the population of sooty terns on Ascension Island has not recovered in response to seabird conservation efforts to date are questions that require considerable future investigation.

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REFERENCES

Anon. (1998). *Metchat – The Ascension Island Climate*. Ascension Island: Meteorological Office.

Ashmole, N.P. (1963). 'The biology of the wideawake or sooty tern *Sterna fuscata* on Ascension Island'. *Ibis* 103b: 297–364.

Ashmole, P., Ashmole, M. and Simmons, K.E.L. (1992). *Seabird Conservation and Feral Cats on Ascension Island, South Atlantic. ICBP Technical Publication No.1*, pp. 94–121. Cambridge, UK: ICBP.

Ashmole, N.P. and Ashmole, M. (2000). *St Helena and Ascension Island: A Natural History*. Oswestry, UK: Antony Nelson.

Baker, J., Harvey, K.J. and French, K. (2013). 'Threats from introduced birds to native birds'. *Emu* 114: 1–12.

Bell, B.D. and Ashmole, N.P. (1995). *The Feasibility of Eradication of Feral Cats and Rats from Ascension Island*. Wellington, New Zealand: Wildlife Management International Ltd.

Bell, M. and Boyle, D. (2004). *The Eradication of Feral Cats from Ascension Island*. Wellington, New Zealand: Wildlife Management International Ltd.

Bibby, C.J., Burgess, N.D., Hill, D.A. and Mustoe, S.H. (2000). *Bird Census Techniques*. London, UK: Academic Press.

BirdLife International (2015). 'Species Factsheet: *Acridotheres tristis*'. BirdLife International Data Zone [website]. <www.birdlife.org>. Accessed 9 May 2015.

Birdlife International (2017). 'Important bird areas factsheet: Wideawake fairs'. *BirdLife International Data Zone* [website]. <www.birdlife.org>. Accessed 5 July 2017.

Byrd, V.G. (1979). 'Common myna predation on wedge-tailed shearwater eggs'. *Elepaio* 39: 69–70.

Cameron, E.K., Vilà, M. and Cabeza, M. (2016). 'Global meta-analysis of the impacts of terrestrial invertebrate invaders on species, communities and ecosystems'. *Global Ecology and Biogeography* 25: 596–606.

Clavero, M. and Garcia-Berthou, E. (2005). 'Invasive species are a leading cause of animal extinctions'. *Trends in Ecology and Evolution* 20: 110.

Cunningham, D.M. and Moors, P.J. (1983). *A Guide to the Identification and Collection of New Zealand Rodents*. Wellington, New Zealand: Department of Internal Affairs.

Duffey, E. (1964). 'The terrestrial ecology of Ascension Island'. *Journal of Applied Ecology* 1: 219–247.

Feare, C.J., Lebarbenchon, C., Dietrich, M. and Larose, S. (2015). 'Predation of seabird eggs by common mynas *Acridotheres tristis* on Bird Island, Seychelles, and its broader implications'. *Bulletin of the African Bird Club* 22: 162–170.

Gosler, A.G. (1991). 'On the use of greater covert moult and pectoral muscle as measures of condition in passerines with data for the great tit *Parus major*'. *Bird Study* 38: 1–9.

Grarock, K., Tidemann, C.R., Wood, J. and Lindenmayer, D.B. (2012). 'Is it benign or is it a pariah? Empirical evidence for the impact of the common myna on Australian birds'. *PLOS ONE* 7(7): e40622.

Harper, G.A. and Bunbury, N. (2015). 'Invasive rats on tropical islands: Their population biology and impacts on native species'. *Global Ecology and Conservation* 3: 607–627.

Hart-Davis, D. (1972). *Ascension: The Story of a South Atlantic Island.* London, UK: Constable.

Hughes, B.J., Martin, G.R. and Reynolds, S.J. (2008). 'Cats and seabirds: Effects of feral domestic cat *Felis silvestris catus* eradication on the population of sooty terns *Onychoprion fuscata* on Ascension Island, South Atlantic'. *Ibis* 150(Suppl. 1): 121–129.

Hughes, B.J., Martin, G.R. and Reynolds, S.J. (2010). 'Sooty terns *Onychoprion fuscata* on Ascension Island in the South Atlantic are a reproductively isolated population'. *Revista Brasileira Ornitologia* 18: 194–198.

Hughes, B.J. (2014). 'Breeding and Population Ecology of Sooty Terns on Ascension Island'. PhD thesis. Birmingham, UK: University of Birmingham.

Hughes, B.J., Martin, G.R., Giles, A.G. and Reynolds, S.J. (2017a). 'Long-term population trends of sooty terns *Onychoprion fuscatus*: Implications for conservation'. *Population Ecology* 59: 213–224.

Hughes, B.J., Martin, G.R. and Reynolds, S.J. (2017b). 'Estimating the extent of seabird egg depredation by introduced common mynas on Ascension Island in the South Atlantic.' *Biological Invasions* 19: 843–857.

Jeschke, J.M., Bacher, S., Blackburn, T.M., Dick, J.T.A., Essl, F., Evans, T., Gaertner, M., Hulme, P.E., Kühn, I., Mrugała, A., Pergl, J., Pyšek, P., Rabitsch, W., Ricciardi, A., Richardson, D.M., Sendek, A., Vilŕ, M., Winter, M. and Kumschick, S. (2014). 'Defining the impact of non-native species'. *Conservation Biology* 28: 1188–1194.

Johnson, D.H. and Shaffer, T.L. (1990). 'Estimating nest success: When Mayfield wins'. *The Auk* 107: 595–600.

Jones, H.P., Tershy, B.R., Zavaleta, E.S., Croll, D.A., Keitt, B.S., Finkelstein, M.E. and Howald, G.R. (2008). 'Severity of the effects of invasive rats on seabirds: A global review'. *Conservation Biology* 22: 16–26.

Koenig, W.D. (1982). 'Ecological and social factors affecting hatchability of eggs'. *The Auk* 99: 526–536.

Lowe, S., Browne, M., Boudjelas, S. and De Poorter, M. (2000). *100 of the World's Worst Invasive Alien Species: A Selection from the Global Invasive Species Database.* Auckland, New Zealand: The Invasive Species Specialist Group (ISSG) of the World Conservation Union (IUCN).

McCreless, E.E., Huff, D.D., Croll, D.A., Tershy, B.R., Spatz, D.R., Holmes, N.D., Butchart, S.H.M. and Wilcox, C. (2016). 'Past and estimated future impact of invasive alien mammals on insular threatened vertebrate populations'. *Nature Communications* 7: 12488.

Medina, F.M., Bonnaud, E., Vidal, E., Tershy, B.R., Zavaleta, E.S., Donlan, C.J., Keitt, B.S., Le Corre, M., Horwath, S.V. and Nogales, M. (2011). 'A global review of the impacts of invasive cats on island endangered vertebrates'. *Global Change Biology* 17: 3503–3510.

Moors, P.J. and Atkinson, I.A.E. (1984). 'Predation on seabirds by introduced animals, and factors affecting its severity'. In: J.P. Croxall, P.G.H. Evans and R.W. Schreiber (eds.) *Status and Conservation of the World's Seabirds ICBP Technical Publication No. 2*, pp. 667–690. Cambridge, UK: ICBP.

Nogales, M., Vidal, E., Medina, F.M., Bonnaud, E., Tershy, B.R., Campbell, K.J. and Zavaleta, E.S. (2013). 'Feral cats and biodiversity conservation: The urgent prioritization of island management'. *Bioscience* 63: 804–810.

Paleczny, M., Hammill, E., Karpouzi, V. and Pauly, D. (2015). 'Population trend of the world's monitored seabirds, 1950-2010'. *PLOS ONE* 10(6): e0129342.

Peck, D.R., Faulquier, L., Pinet, P., Jaquemet, S. and Le Corre, M. (2008). 'Feral cat diet and impact on sooty terns at Juan de Nova Island, Mozambique Channel'. *Animal Conservation* 11: 65–74.

Pickup, A.R. (1999). *Ascension Island Management Plan*. Cambridge, UK: BirdLife International.

Plantinga, E., Bosch, G. and Hendriks, W. (2011). 'Estimation of the dietary nutrient profile of free-roaming feral cats: Possible implications for nutrition of domestic cats'. *British Journal of Nutrition* 106: 35–48.

Prugh, L.R., Stoner, C.J., Epps, C.W., Bean, W.T., Ripple, W.J., Laliberte, A.S. and Brashares, J.S. (2009). 'The rise of the mesopredator'. *Bioscience* 59: 779–791.

Ratcliffe, N., Bell, M., Pelembe, T., Boyle, D., Benjamin, R., White, R., Godley, B., Stevenson, J. and Sanders, S.M. (2009). 'The eradication of feral cats from Ascension Island and its subsequent recolonization by seabirds'. *Oryx* 44: 20–29.

Reynolds, S.J., Martin, G.R., Dawson, A., Wearn, C.P. and Hughes, B.J. (2014). 'The sub-annual breeding cycle of a tropical seabird'. *PLOS ONE* 9(4): e93582.

Ringler, D., Russell, J.C. and Le Corre, M. (2015). 'Trophic roles of black rats and seabird impacts on tropical islands: Mesopredator release or hyperpredation?' *Biological Conservation* 185: 75–84.

Ritsema, A. (2006). *A Dutch Castaway on Ascension Island in 1725*. Deventer, The Netherlands: Alex Ritsema.

Russell, J.C., Lecomte, V., Dumont, Y. and Le Corre, M. (2009). 'Intraguild predation and mesopredator release effect on long-lived prey'. *Ecological Modelling* 220: 1098–1104.

Schreiber, E.A. and Burger, J. (2002). *Biology of Marine Birds*. Boca Raton, FL, USA: CRC Press.

Schreiber, E.A., Feare, C.F., Harrington, B.J., Murray, B.G. Jr, Robertson, W.B. Jr, Robertson, M.J. and Woolfenden, G.E. (2002). 'Sooty tern (*Sterna fuscata*)'. In: A. Poole and F. Gill (eds.) *Birds of North America No. 665*, pp. 1–32. Philadelphia, PA, USA: American Ornithologists' Union.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?' *Biological Invasions* 8: 863–891.

Van Aarde, R.J. (1983). 'Demographic parameters of the feral cat *Felis catus* population at Marion Island'. *South African Journal of Wildlife Research* 13: 12–16.

White, G.C. and Burnham, K.P. (1999). 'Program MARK: Survival estimation from populations of marked animals'. *Bird Study* 46(suppl.): S120–S139.

Young, H.S., McCauley, D.J., Galetti, M. and Dirzo, R. (2016). 'Patterns, causes, and consequences of Anthropocene defaunation'. *Annual Review of Ecology, Evolution, and Systematics* 47: 333–358.

Zarzoso-Lacoste, D., Ruffino, L. and Vidal, E. (2011). 'Limited predatory capacity of introduced black rats on bird eggs: An experimental approach'. *Journal of Zoology* 285: 188–193.

Eradication and control programmes for invasive mynas (*Acridotheres* spp.) and bulbuls (*Pycnonotus* spp.): defining best practice in managing invasive bird populations on oceanic islands

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Abstract Invasive plants and animals inflict much damage on native species and this is particularly the case on isolated oceanic islands with high degrees of endemism. Such islands commonly are important refugia for species of high conservation value. Some of the most pervasive and potent of invasive animal species are birds of the myna (*Acridotheres*) and bulbul (*Pycnonotus*) genera that historically were introduced to isolated islands as biological control agents for the management of insect pest species that can cause considerable economic damage to agricultural crops and wider ecosystems. In this paper we consider a number of 'successful' eradication and control programmes targeting mynas and bulbuls. We review the locations and taxa where 17 such programmes took place and report that the common myna *(Acridotheres tristis)* has been the most heavily targeted species in eradication efforts followed by the red-whiskered bulbul *(Pycnonotus jocosus)*. Common mynas were also at the focus of control programmes as were jungle mynas *(Acridotheres fuscus)* and red-vented bulbuls *(Pycnonotus cafer)*. By far the most favoured method of eradication and control was trapping whereas mist-netting was employed rarely. We discuss 'best practice' in planning and executing such eradication and control programmes on oceanic islands so as to maximise their benefits to local human communities. We outline measures that must be adopted pre-, during and post-intervention in both programme types. They include adequate resourcing, local engagement and the integration of both traditional ecological knowledge and established conservation theory.

Keywords: engagement, ethno-ornithology, invasive bird management, local community, multispecies approach

INTRODUCTION

The modern world is experiencing unprecedented anthropogenic inputs that are resulting in the sixth global wildlife extinction (Foley, et al., 2013) with concomitant phenomena such as accelerating climate change (Crowley, 2000) and increased frequencies of invasions of alien species (Vitousek, et al., 1997; Dukes & Mooney, 1999), resulting in losses of biodiversity (Lowe, et al., 2000).

The establishment of early human societies resulted in the trade of goods and services (Zeder, 2008), and the accompanying development of transport modes and infrastructure, such as roads and other trading routes, resulted in commodities traded over greater distances (Earle, 1994). Inevitably, this resulted in the movement of species out of their native ranges into areas where they were alien (exotic). Today, we continue to trade goods and services internationally and in so doing we move thousands of exotic species, approximately 10% of which will become established as invasive (Williamson & Fitter, 1996; Westphal, et al., 2008). By definition, an alien species occurs outside of its natural (past or present) range and it has dispersal potential, including any part of it (e.g. propagules, gametes) surviving and subsequently reproducing (Lowe, et al., 2000). Invasive species are targeted for conservation actions because they are alien species that become established in natural or semi-natural ecosystems or habitats where they present problems to native species (Colautti & MacIsaac, 2004). The impacts of avian invasive species on ecosystems are pervasive and enduring; they include, for example, competitive exclusion and predation of native species, disease transmission and dilution of native gene pools through hybridisation (reviewed in Blackburn, et al., 2009).

Oceanic islands are known to be more susceptible to negative impacts of exotic species compared with continental land masses because of their increased endemism as a result of their geographical isolation (Coblentz, 1990; Reaser, et al., 2007; Feare, 2017). Furthermore, their ecological fragility is magnified on smaller islands that accommodate more simple native ecological communities than larger ones (Donlan & Wilcox, 2008). Therefore, conservation priorities for insular environments are often defined by the need for effective eradication and/or control programmes of invasive species (e.g. Dulloo, et al., 2002; Donlan & Wilcox, 2008).

In this study we focus on two invasive avian genera (i.e. mynas *Acridotheres* and bulbuls *Pycnonotus*), consisting of six different species. These two genera are both represented on the list of '100 of the World's Worst Invasive Alien Species' (Lowe, et al., 2000), a subset of the Global Invasive Species Database, by common mynas (*Acridotheres tristis*) and red-whiskered bulbuls (*Pycnonotus jocosus*). The 18th and 19th centuries saw a series of introductions of mynas to oceanic islands as biocontrol agents to counter insect pests that threatened agricultural production. They were also transported to oceanic islands as cage birds. On Tutuila in American Samoa, for example, common mynas arrived in 1980 and jungle mynas (*Acridotheres fuscus*) in 1985 (SSC, unpubl. data) while on Ascension Island (Hughes, et al., 2017) and St Helena (Burns, 2011) they were introduced in the 19th century. Bulbuls were kept widely as caged birds but escaped captivity on islands such as Tahiti in 1925, Assumption in the 1970s and Tenerife and Fuerteventura at the turn of the 21st century.

The aim of our study is to provide an account of the characteristics of successful eradication and control programmes and then to discuss how they can be used to define 'best practice'. Through material presented in the discussion, we indicate how the conservation community can take effective measures to combat avian invasive species on remote oceanic islands.

METHODS

We used the following keywords – 'common myna', 'bulbul', '*Acridotheres*', '*Pycnonotus*', 'trapping', 'control', 'shooting', 'island*', and 'eradication' – in searches of several bibliographic databases including Webspire, Web

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of Knowledge, Ovid SP, Inist, Blackwell Publishing and Science Direct to identify primary scientific literature about management programmes for avian invasive species on islands. References in literature cited/bibliography sections of the resulting sources were also considered for inclusion in our study. Whether programmes were considered further as successful eradications or controls, or rejected followed correspondence with programme managers to obtain further details about the interventions (Table 1). At the time of writing all programme managers have been contacted and we have received responses from all but one of them. Following inclusion in the study, the information obtained from these programme managers combined with that in publications was examined to assess a number of factors determining the effectiveness of programmes: the target species; and the numbers of birds of each species and the methods used as part of the intervention.

RESULTS

Literature searches

We did not consider every programme where eradication or control of avian invasive species had been attempted because after apparently successful removal of invasive birds, they reappeared on some islands (Table 1). The publications that were not considered further are detailed in Table 2. Following exclusions of these published studies we were left with 17 programmes (Table 3); their locations are shown in Fig. 1.

Invasive species targeted

Common mynas have been successfully eradicated from eight islands (Tenerife, Gran Canaria, Mallorca, Fuerteventura, Fregate, Denis, Tarawa and Atiu), and red-whiskered bulbuls from two islands (Tenerife and Assumption; Table 4). Control efforts targeting common mynas are ongoing on North Island in Seychelles (Table 4). Short-term isolated control programmes targeting common mynas were carried out twice on Ascension Island and once on St Helena, each being conducted in late 2009 (Table 4). An ongoing project on Tahiti is carrying out long-term control of common mynas and red-vented bulbuls (*Pycnonotus cafer*). In the Tarawa eradication two *Acridotheres* species were targeted and they were the common myna and the jungle myna. The only multi-species long-term control programme running today is in American Samoa where the two aforementioned myna species and the red-vented bulbul are being successfully targeted through trapping, with approximately 9,600 birds being captured in two consecutive trapping campaigns.

Total numbers of birds of each species by island

Table 4 provides details of the numbers of birds of each species that have been targets of population eradication and control programmes in the 17 projects (see also Table 3). In total, over 57,000 invasive birds have been captured. With ongoing projects such as the work on, for example, Tahiti and Tutuila (long-term control programmes) and North Island (an eradication programme), numbers are predicted to climb steeply in the near future. The vast majority of birds were common mynas and most were captured in Atiu in the Cook Islands (officially 'eradicated' but one remaining bird currently being tracked; SSC, unpubl. data), and on Tahiti in the control programme (Table 4). All but one of the 4,606 jungle mynas were caught in the ongoing control programme on Tutuila. The majority of red-whiskered bulbuls were caught as part of the eradication programme on Assumption, while most red-vented bulbuls were caught in the control programme on Tahiti (Table 4).

Table 1 Inclusive sets of categorisation criteria that allowed us to exclude (reject) or include published studies as successful eradication and control programmes targeting avian invasive species on oceanic islands as a result of questioning programme managers (see Methods for further details).

Reject	Eradication	Control
Birds were present as of April 2017	Birds were absent as of April 2017	Birds were present as of April 2017
No post-intervention monitoring	Post-intervention monitoring found no birds	Post-intervention monitoring found reproductive birds
No defined milestones during intervention	Defined milestones during intervention	Defined milestones during intervention
Pathways of invasion remain open	Pathways of invasion closed	Pathways of invasion remain open
No defined period of quarantine	Defined period of quarantine	No defined period of quarantine

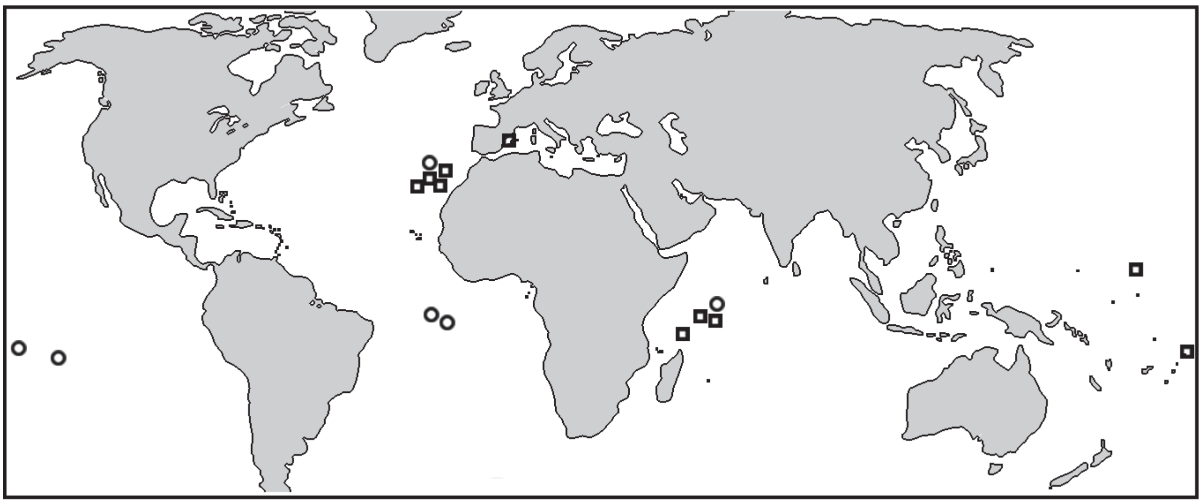

Fig. 1 Locations of islands where eradication (square symbols), and control (round symbols) programmes have been carried out to address problems of invasive myna and bulbul genera (see Tables 3 & 4 for further details).

Table 2 Details of eradication and control programmes on islands involving myna and bulbul genera, including the location, focal species, the number of birds eradicated/controlled (where known), and notes (including references) to explain why programmes were excluded from further consideration in our study.

Location	Species	No of birds	Notes (including references where available)
Ascension Island, Atlantic Ocean	Common myna *Acridotheres tristis*	40	Trapped birds were non-target species during feral domestic cat (*Felis silvestris catus*) removal in 2004 (Hughes, et al., 2008)
Seychelles, Indian Ocean	Common myna	-	Some birds remained after the eradication project ended (Canning, 2011). Eradication was abandoned when rats were discovered on site on Denis (Millett, et al., 2005)
Fakaofo (Tokelau), Pacific Ocean	Common myna	40	Birds targeted in 2006 with their egg and nest destruction resulting in no further sightings in 2011, but in early 2012 birds were seen on Nukunonu Atoll, 64 km north of Fakaofo (Parkes, 2012)
Western Samoa, Pacific Ocean	Bulbul *Pycnonotus* spp. and myna *Acridotheres* spp.	6,000	Feeding of ©DCR-1339 (3-chloro-p-toluidine hydrochloride) has been effective but to date no strategy to control avian invasive species has been formalised and consistently implemented island-wide
Mainland Australia	Common myna	>69,000	The Canberra Indian Myna Action Group (CIMAG) work, removing birds over 11 years using volunteer trappers, has taken place on a continental land mass and is of limited applicability to oceanic islands
Moturoa Island, Bay of Islands, New Zealand	Common myna	45	No detailed results were reported from trapping which has been criticised as an inappropriate method to control this species (Parkes, 2012)

Table 3 Island groups and islands where eradication and control programmes targeting myna and bulbul genera were carried out and the year when they ended and started, respectively.

Eradication			Control		
Island group	Island	Year	Island group	Island	Year
Balearic Islands	Mallorca	2007	American Samoa	Tutuila	2016
Canary Islands	Fuerteventura	2008	Canary Islands	Fuerteventura	2010
Canary Islands	Gran Canaria	2006	French Polynesia	Tahiti	2012
Canary Islands	Tenerife	2000	Seychelles	North	2016
Canary Islands	Tenerife	2007	UK Overseas Territories	Ascension[b]	Sept. 2009
Cook Islands	Atiu	2016	UK Overseas Territories	Ascension[c]	Nov. 2009
Kiribati	Tarawa	2015			
Seychelles	Assumption[a]	2014			
Seychelles	Denis	2015			
Seychelles	Fregate	2011			

[a]Eradication of one target genus (i.e. red-whiskered bulbul) and one non-target genus (i.e. red fody *Foudia madagascariensis*) was achieved
[b]A control programme carried out by SSC by trapping
[c]A separate one carried out by C.J. Feare by poisoning, in the same year

Methods employed on projects

Methods used in eradication and control programmes included trapping, shooting, poisoning and mist-netting (Table 5). The method of choice for both programme types was live-trapping of invasive birds using live decoys and edible baits such as bread, fruit, pet food and tinned fish. Few programmes used shooting, with four out of five programmes employing firearms being conducted for population eradication purposes. Three out of the four programmes using poisoning were controlling (as opposed to eradicating) populations of invasive species. Only two eradication (but no control) programmes employed mist-netting to capture birds.

DISCUSSION

It was clear when we reviewed published studies and contacted programme managers that some programmes described as eradications should have been categorised as ongoing control programmes, according to our classification criteria outlined in Table 1. For those that did not carry out post-intervention monitoring, had not defined milestones during the intervention, had not identified invasion pathways or had not stipulated a period of quarantine post-intervention, we suggest that they should not be considered successful control programmes. We also request that programme managers consider carefully the contents of Table 1 as they plan and execute their intervention. Many studies were published before 2000 and they were unsuccessful in the case of eradication programmes because pathways of invasion were not closed and/or programme managers failed to remove all targeted birds (Tables 1 and 2). Remaining populations therefore recovered in numbers and, as a result, they expanded their ranges once again on islands. Such an example was provided by Millett, et al., (2005).

Table 4 Total number of mynas and bulbuls of six different invasive species caught on islands (see Table 3 for further details) as part of eradication and control programmes. Note that red-vented bulbul has been split into two subspecies – *cafer* and *bengalensis* – for historical reasons.

Island (year)	Invasive species (*A.*= *Acridotheres*; *P.*= *Pycnonotus*)					
	Common myna *A. tristis*	Jungle myna *A. fuscus*	*A.* hybrid	Red-whiskered bulbul *P. jocosus*	Red-vented bulbul *P. cafer cafer*	Red-vented bulbul *P. c. bengalensis*
Eradication						
Assumption (2014)				5,279		
Atiu (2016)	24,375					
Denis (2015)	1,186					
Fregate (2011)	758					
Fuerteventura (2008)	21					
Gran Canaria (2006)			3			
Mallorca (2006)	22					
Tarawa (2015)	3	1				
Tenerife (2000)	11					
Tenerife (2007)				7		
Total birds	26,376	1	3	5,286		
Control						
Ascension (Sept. 2009)	623					
Ascension (Nov. 2009)	114					
Fuerteventura (2010)					7	
North (2016)	1,600					
St Helena (2009)	342					
Tahiti (2012)	6,170					9,123
Tutuila (2016)	2,915	4,605				2,401
Total birds	11,764	4,605			7	11,524
Total birds for both programme types	38,140	4,606	3	5,286	7	11,524

Our empirical results document the species, the numbers of birds of each taxon and the methods employed during the targeting of birds in eradication and control programmes. It is clear that traps should be favoured to ‘capture’ invasive birds as we understand more about the biology of the target species and because trap design has markedly improved over recent years. From a practical perspective, the construction and establishment of traps on the ground are more preferable to applying continuously for permits from authorities on isolated islands to import and use firearms and poison. This said, national governmental agencies would be well advised to facilitate the use of complementary and effective management methods that can be combined with trapping to allow programme staff to progress invasive bird management on these islands and others in the future. As an example, the experience from Assumption suggests that combining mist-netting with shooting can result in removal of large numbers of red-whiskered bulbuls (now eradicated) and red fodies (*Foudia madagascariensis*).

As a result of considerations of both excluded (Table 2) and included studies (Tables 3 and 4) documenting eradication and control programmes, we briefly discuss below some of the fundamental considerations that should be undertaken in their future planning, execution and reporting. The outcome should be the adoption of processes that lead to best practice in managing invasive bird populations on oceanic islands.

Community engagement

Many programme managers historically argued that it was impossible to rely on local people to instigate actions on the ground, to remain committed to the programme and thus to constitute the main task force addressing the problems posed by the invasive species, as the programme will be destined to fail because of local apathy (SSC, pers. obs.). Nowadays, programme managers often assume that the programme’s aims will thrive mediated by the locals’ sense of community and shared aspirations for the programme. Success comes through the development of simple ‘tools’ that can be employed by the local community to manage invasive species for the benefit of the whole community. While people who want to become volunteers (whether trapping or otherwise) in any invasive species management programme have their own motivations for doing so, the success of any such intervention lies in the effective coordination of human power directed towards an achievable and beneficial community goal. This sustains commitment to the programme, especially from the community itself.

A successful programme will not only engage with the local community but also with wider audiences, requiring widespread availability of well-designed and well-delivered education campaigns, and comprehensive media coverage. The Canberra Indian Myna Action Group (CIMAG) provides an excellent example (albeit a mainland one) of a society-driven movement of volunteer

Table 5 Methods employed on eradication and control programmes targeting mynas and bulbuls of six different invasive species caught on islands (see Table 3 for further details).

Island (year)	Ocean	Method			
		Trapping	Shooting	Poisoning	Mist-netting
Eradication					
Assumption (2014)	Atlantic		✓		✓
Atiu (2016)	Pacific	✓	✓	✓	
Denis (2015)	Indian	✓	✓		
Fregate (2011)	Indian	✓			
Fuerteventura (2008)	Atlantic	✓			
Gran Canaria (2006)	Atlantic	✓			
Mallorca (2006)	Mediterranean	✓	✓		
Tarawa (2015)	Pacific		✓		
Tenerife (2000)	Atlantic	✓			
Tenerife (2007)	Atlantic	✓	✓		✓
Totals		8	6	1	2
Control					
Ascension (Sept. 2009)	Atlantic	✓			
Ascension (Nov. 2009)	Atlantic			✓	
Fuerteventura (2010)	Atlantic	✓			
North (2016)	Indian	✓			
St Helena (2009)	Atlantic	✓		✓	
Tahiti (2012)	Pacific	✓	✓	✓	
Tutuila (2016)	Pacific	✓			
Totals		6	1	3	

community trappers that has removed >69,000 common mynas and 8,900 common starlings (*Sturnus vulgaris*) through trapping over the last 11 years (CIMAG, pers. comm.). Their programme started in 2006 and it has achieved unprecedented successes in controlling birds on a continental scale, thereby demonstrating the effectiveness of well-coordinated volunteer efforts. Invasive birds have been managed effectively on Tahiti for the last seven years and on Tutuila for nearly the last three years.

Programme resourcing

We make a few general points about resourcing, based upon experiences of SSC gained from the control programme carried out on Tahiti in 2012 (Tables 3 and 4). This programme was driven by the need for urgent conservation action to promote the survival of the critically endangered Tahiti monarch (*Pomarea nigra*) (Blanvillain, et al., 2003; Ghestemme, 2011). It was a success because the programme engaged fully with the local community, and maintained high levels of motivation among local community members by sustaining frequent and dynamic communication between the local community and the programme's management team. It provided many insights that could be transferred to other such programmes. Contractors should provide an upfront realistic budget to meet the costs incurred in mobilising materials and having personnel in post at the start of actions on the ground. Mobilisation requires transport logistics, appropriate personnel to be available and fuel costs to be met at the start of the programme. Materials can include components for trap construction, mist-nets and their associated poles, firearms and ammunition, bait stations, bait and poisons, and storage facilities. Often equipment like traps has been used for centuries but knowledge about the appropriate deployment of them has been lost trans-generationally. Money spent on re-education and re-training to address the deployment of single traps and of coordinated networks of traps is particularly well received, especially in locations such as the Pacific islands (SSC, pers. obs.) where remote communities rely upon subsistence agriculture for food security and invasive bird species in part threaten their very existence.

There are costs associated with employing appropriate (i.e. informed) staff on such programmes (e.g. advertising, interviewing) and submitting applications for permits to relevant on- or off-island authorities for activities such as the use of mist-nets and traps, the handling of hazardous chemicals and the safe disposal of managed birds. Funding is also needed to maintain surveillance efforts to ensure that invasive birds have not returned (eradication programmes) or exist in low numbers as a result of sustained trapping efforts (control programmes).

Perceptions of invasive (and native) species

In many locations outside of their native ranges invasive species may be the first birds that locals observe and become familiar with (CIMAG, pers. comm.; SSC, pers. obs.). Their overwhelming presence can result in native species becoming 'invisible' in local communities both in terms of reduced numbers of birds on the ground and a loss of natural history knowledge through education and personal experiences. This erosion of so-called 'traditional ecological knowledge' (TEK) is a widespread phenomenon (Sinclair, et al., 2010) and is not just restricted to remote oceanic islands. Children often tend to consider invasive species as 'normal' because they observe them constantly throughout their formative years. In local community-based management projects, public awareness of native species for aesthetic, as well as ecosystem service, benefits is crucial in gaining public support, resulting in potent public engagement with invasive eradication and control programmes. Local people become highly motivated rapidly, especially if provided with effective management 'tools' to control invasive bird species. The challenge to the conservation manager is to promote native species' survival as a positive outcome of effective invasive species management in addition to other benefits to the local community. Whether this generates a conservation ethic in local peoples beyond that of their livelihoods remains aspirational but realistic, given experiences of SSC in the last seven years of control in Tahiti and three years in Tutuila.

Expertise networks

If invasive species are to be targeted successfully we must develop networks of expertise that are constituted not just by species experts (e.g. invasive species managers, professional ornithologists, avian pest controllers), but also by local experts who have developed detailed knowledge of the target species on the ground after training. Networks can thereby provide a detailed knowledge of the species' biological traits such as flocking patterns (Sinu, 2011), responses to novel foods (Martin & Fitzgerald, 2005), changes in food preference in relation to their location in their distributional range (Liebl & Martin, 2014), and trap shyness (Camacho, et al., 2017). For example, common mynas can be trapped for long periods of time without developing 'trap shyness' (SSC, pers. obs.), but only if trappers follow the recommended protocols.

Usually, local people have some biological knowledge of targeted invasive species, but on rare occasions some have detailed local knowledge about birds. All such knowledge can be obtained from full engagement with the local community who may have attempted eradication and control methods albeit in an uncoordinated manner that invariably results in unsuccessful outcomes. Knowledge can relate to where birds roost, favoured routes between roost and foraging sites, where they drink, their preferred foods and even how they behave in response to presentation of novel foods (e.g. Lermite, et al., 2017, SSC, pers. obs.). In some cases, ethno-ornithological knowledge (Tidemann & Gosler, 2010) could prove to be fundamental in the successful deployment of methods on the ground but to the best of our knowledge it has failed to inform eradication and control programmes to date.

Creating and sustaining networks of trappers on single islands and on chains of islands are fundamental in targeting high numbers of birds to be removed. Full engagement in terms of commitment and motivation by programme managers is key to retaining network integrity. Communication is the principal way to enlist assistance from trap builders and volunteer trappers, to inform the island population, to recruit local people to the programme, to educate the community about its benefits, and to update local people about the results of the programme to date. It is not just the general public that needs to be updated but, just as importantly, members of the trapper network itself. Sharing positive results from the ongoing programme motivates everyone and if a problem in the network is described in sufficient detail, a solution can be found rapidly because of shared experience and capacity in problem solving. Of course, a sustained line of communication also engages with stakeholders beyond the programme's location such as international agencies who might be partially funding the work.

The reality of most programmes is that training of staff takes the form of native biodiversity conservation but that of volunteers is focussed on local habitat protection, whether cash crop, farmland or otherwise. The practical training to build and deploy traps should be similar for both of the above groups, but is often viewed as being less exigent for volunteers. However, if local people are trained in partnership with programme staff through an established expertise network, often trap design and deployment can

Table 6 Attributes defining best practice in planning and executing effective eradication and control programmes of avian invasive species on oceanic islands.

Attribute	Eradication	Control
Pre- and during intervention		
Local government support	✓	✓
Stakeholders identified and engaged with	✓	✓
Internal and external communication channels identified and open	✓	✓
Training of local and contract staff	✓	✓
Milestones identified	✓	✓
Full financial resources	✓	✓
Full non-financial resources	✓	✓
Post-intervention		
Full financial resources (including contingencies)	✓	✓
Full non-financial resources (including contingencies)	✓	✓
Refresher training of local and contract staff	✓	✓
All communication channels remain open	✓	✓
Birds absent	✓	×
Monitoring for birds	✓	×
Pathways of invasion closed	✓	×
Ongoing management of pathways of invasion	✓	×
Defined period of quarantine	✓	×

be improved through inputs of local knowledge (Tidemann & Gosler, 2010). Part of such training should include emphasising the importance of record keeping. Recording data is key to a programme's success but can sometimes be problematic when carried out by local trappers without an appreciation for its importance. The transmission of data between trappers, programme managers and their staff can result in the loss of data when resources such as standardised datasheets, time, computer hardware and software etc. are lacking. Data collection should run smoothly with full commitment of participants on such programmes if training has been effective and expertise networks are maintained.

What constitutes best practice in control and eradication programmes targeted at invasive bird populations on oceanic islands?

To conclude, we refer the reader to Table 6 where we summarise the main attributes of effective control and eradication programmes. These attributes should be considered alongside others that we have discussed in this study. In conclusion, we have provided an account of the most common invasive avian species that have been targets for conservation action on oceanic islands where they threaten native species and the livelihoods of local human communities. Mynas and bulbuls still pose major threats to local economies and to native biodiversity, and we must find ways to plan and execute their eradication and control that engage with local communities while guaranteeing that programme outcomes are attained. Above, we have discussed effective planning through full engagement with and between the local community, programme managers and team members (whether volunteers or otherwise) to capacity build through education and training. This results in the construction and maintenance of expertise networks that are built on the ideas of local people, harnessing their local knowledge about the target species and on an appreciation of the benefits of the proposed actions to the local community. Executing plans involves coordinated action on the ground between programme managers, their staff and local volunteers that arises from sustained communication and motivation in meeting all of the programme's goals. Success involves far more than simply providing financial resources to cover various elements of a programme. If we were to propose one overarching recommendation it would be that programmes share information using standardised reporting protocols as everyone strives to adopt best practice.

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REFERENCES

Blackburn, T.M., Lockwood, J.L. and Cassey, P. (2009). *Avian Invasions: The Ecology and Evolution of Exotic Birds*. Oxford, UK: Oxford University Press.

Blanvillain, C., Salducci, J.M., Tutururai, G. and Maeura, M. (2003). 'Impact of introduced birds on the recovery of the Tahiti flycatcher (*Pomarea nigra*), a critically endangered forest bird of Tahiti'. *Biological Conservation* 109: 197–205.

Burns, F.E. (2011). 'Conservation Biology of the Endangered St Helena Plover *Charadrius sanctaehelenae*'. PhD thesis. Bath, UK: University of Bath.

Camacho, C., Canal, D. and Potti, J. (2017). 'Lifelong effects of trapping experience lead to age-biased sampling: Lessons from a wild bird population'. *Animal Behaviour* 130: 133–139.

Canning, G. (2011). 'Eradication of the invasive common myna, *Acridotheres tristis*, from Fregate Island, Seychelles'. *Phelsuma* 19: 43–53.

Coblentz, B.E. (1990). 'Exotic organisms: A dilemma for conservation biology'. *Conservation Biology* 4: 261–265.

Colautti, R.I. and MacIsaac, H.J. (2004). 'A neutral terminology to define 'invasive' species'. *Diversity and Distributions*10: 135–141.

Crowley, T.J. (2000). 'Causes of climate change over the past 1000 years'. *Science* 289: 270–277.

Donlan, C.J. and Wilcox, C. (2008). 'Diversity, invasive species and extinctions in insular ecosystems'. *Journal of Applied Ecology* 45: 1114–1123.

Dukes, J.S. and Mooney, H.A. (1999). 'Does global change increase the success of biological invaders?' *Trends in Ecology and Evolution* 14: 135–139.

Dulloo, M.E., Kell, S.P. and Jones, C.G. (2002). 'Conservation of endemic forest species and the threat of invasive species'. *International Forestry Review* 4: 277–335.

Earle, T. (1994). 'Positioning Exchange in the Evolution of Human Society'. In: T.G. Baugh and J.E. Ericson (eds.) *Prehistoric Exchange Systems in North America. Interdisciplinary Contributions to Archaeology*, pp. 419–437. Boston, MA, USA: Springer.

Feare, C.J. (2017). *Orange Omelettes & Dusky Wanderers: Studies and Travels in Seychelles Over Four Decades*. Mahé, Seychelles: Calusa Bay Publications.

Foley, S.F., Gronenborn, D., Andreae, M.O., Kadereit, J.W., Esper, J., Scholz, D., Pöschl, U., Jacob, D.E., Schöne, B.R., Schreg, R., Vött, A., Jordan, D., Lelieveld, J., Weller, C.G., Alt, K.W., Gaudzinski-Windheuser, S., Bruhn, K.-C., Tost, H., Sirocko, F. and Crutzen, P.J. (2013). 'The Palaeoanthropocene – the beginnings of anthropogenic environmental change'. *Anthropocene* 3: 83–88.

Ghestemme, T. (2011). 'Impact of introduced birds on Tahiti monarch'. *Pacific Invasives Initiative Newsletter* 12: 4.

Hughes, B.J., Martin, G.R. and Reynolds, S.J. (2008). 'Cats and seabirds: Effects of feral domestic cat *Felis silvestris catus* eradication on the population of sooty terns *Onychoprion fuscata* on Ascension Island, South Atlantic'. *Ibis* 150 (Suppl. 1): 121–129.

Hughes, B.J., Martin, G.R. and Reynolds, S.J. (2017). 'Estimating the extent of seabird egg depredation by introduced common mynas on Ascension Island in the South Atlantic.' *Biological Invasions* 19: 843–857.

Lermite, F., Perneaux, C. and Griffin, A.S. (2017). 'Personality and problem-solving in common mynas (*Acridotheres tristis*)'. *Behavioural Processes* 134: 87–94.

Liebl, A.L. and Martin, L.B. (2014). 'Living on the edge: Range edge birds consume novel foods sooner than established ones'. *Behavioral Ecology* 25: 1089–1096.

Lowe, S., Browne, M., Boudjelas, S. and De Poorter, M. (2000). *100 of the World's Worst Invasive Alien Species: A Selection from the Global Invasive Species Database.* Auckland, New Zealand: The Invasive Species Specialist Group (ISSG) of the World Conservation Union (IUCN).

Martin II, L.B. and Fitzgerald, L. (2005). 'A taste for novelty in invading house sparrows, *Passer domesticus*'. *Behavioral Ecology* 16: 702–707.

Millett, J., Climo, G. and Shah, N.J. (2005). 'Eradication of common myna *Acridotheres tristis* population in the granitic Seychelles: Successes, failures and lessons learned'. *Advances in Vertebrate Pest Management* 3: 169–183.

Parkes, J. (2012). *Review of Best Practice Management of Common Mynas (*Acridotheres tristis*) with Case Studies of Previous Attempts at Eradication and Control: A Working Document*. Lincoln, New Zealand: Landcare Research.

Reaser, J.K., Meyerson, L.A., Cronk, Q., de Poorter, M., Eldredge, L.G., Green, E., Kairo, M., Latasi, P., Mack, R.N., Mauremootoo, J., O'Dowd, D., Orapu, W., Sastroutomo, S., Saunders, A., Shine, C., Thrainsson, S. and Vaiutu, L. (2007). 'Ecological and socioeconomic impacts of invasive alien species in island ecosystems'. *Environmental Conservation* 34: 98–111.

Sinclair, R.J., Tuke, L. and Opiang, M. (2010). 'What the Locals Know: Comparing Traditional and Scientific Knowledge of Megapodes in Melanesia'. In: S. Tidemann and A. Gosler (eds.) *Ethno-ornithology: Birds, Indigenous Peoples, Culture and Society*, pp. 115–137. London, UK: Earthscan.

Sinu, P.A. (2011). 'Avian pest control in tea plantations of sub-Himalayan plains of Northeast India: Mixed-species foraging flock matters'. *Biological Control* 58: 362–366.

Tidemann, S. and Gosler, A. (2010). *Ethno-ornithology: Birds, Indigenous Peoples, Culture and Society*. London, UK: Earthscan.

Vitousek, P.M., D'Antonio, C.M., Loope, L.L., Rejmánek, M. and Westbrooks, R. (1997). 'Introduced species: A significant component of human-caused global change'. *New Zealand Journal of Ecology* 21: 1–16.

Westphal, M.I., Browne, M., MacKinnon, K. and Noble, I. (2008). 'The link between international trade and the global distribution of invasive alien species'. *Biological Invasions* 10: 391–398.

Williamson, M. and Fitter, A. (1996). 'The varying success of invaders'. *Ecology* 77: 1661–1666.

Zeder, M.A. (2008). 'Domestication and early agriculture in the Mediterranean Basin: Origins, diffusion, and impact'. *Proceedings of the National Academy of Sciences USA* 105: 11597–11604.

Invasion by the red-vented bulbul: an overview of recent studies in New Caledonia

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Abstract New Caledonia is a tropical archipelago of the South Pacific Ocean, and is one of the 36 world biodiversity hotspots. However, its unique biodiversity is increasingly threatened by habitat fragmentation and introductions of invasive alien species. Among these invaders, the red-vented bulbul (*Pycnonotus cafer*) is currently expanding towards the north of the main island. This passerine features in the IUCN-ISSG list of the 100 worst invasive species of the world because of impacts caused by its diet. Thirty-five years after its introduction, we present an overview of data from recent studies conducted in New Caledonia that describe the local status of the red-vented bulbul, its range expansion, and potential impacts on both the local biodiversity and agriculture. Biannual monitoring of the distribution coupled with surveillance at the edges of native forests highlighted a tight association of the bulbul with man-modified habitats. Using a distance sampling method, we estimated that bulbul densities within the distribution core varied from a peak of 200 individuals/km^2 in the main city of Nouméa, where the species has been introduced, to 30 individuals/km^2 in rural habitats located 50 km away from Nouméa. We conducted a diet analysis on 40 bulbul corpses and found that 82% and 55% of individuals had consumed plant and animal items, respectively. We identified plant and insect species that may be of concern in the contexts of seed dispersal and predation by the red-vented bulbul. Finally, a food colour selection experiment and an open field test showed that the red-vented bulbul had a significant preference for red and sweet fruits. We estimated the economic loss caused by bulbuls to a tomato grower and discuss the result with respect to the development of an adapted management strategy, to prevent further impacts of the red-vented bulbul on the biodiversity and agriculture in the tropical island hotspot of New Caledonia.

Keywords: density, diet, distribution, impacts, invasive bird, *Pycnonotus cafer*

INTRODUCTION

New Caledonia is a tropical archipelago located to the east of Australia, in the SouthPacific Ocean. The archipelago has been classified as one of the world's 36 biodiversity hotspots because of its high levels of endemism in such a small territory (Williams, et al., 2011). Among notable features of local biodiversity in New Caledonia, Myers, et al. (2000) highlighted five endemic families and 112 endemic genera of plants, and one endemic family and three endemic genera of birds. However, a significant proportion of this biological richness is increasingly threatened by human activities and global changes, as is the case for most of the world's biodiversity hotspots (Bellard, et al., 2014). Among factors that foster these changes, habitat fragmentation and climate change are widely recognized (Garcia, et al., 2014; Haddad, et al., 2015), although the best response from scientists and managers to species' introductions is still a matter of debate (Russell & Blackburn, 2017; Davis & Chew, 2017).

The effects of invasive species have been widely documented (Early, et al., 2016). Impacts are accentuated in island ecosystems (Russell, et al., 2017), often because of the naivety of insular species (Gerard, et al., 2016) and environmental, ecological and evolutionary factors associated with geographic isolation (Cabral, et al., 2017). Humans play a key role in the transportation of plant and animal species worldwide (Ricciardi, et al., 2017). Trade in animals (Cardador, et al., 2017; Su, et al., 2016) and the release or escape of cage birds are frequently identified as the main mechanisms for alien bird introductions and the dispersal of wild birds outside of their native ranges (Dyer, et al., 2017).

Tropical bird species, particularly those from South-east Asia, occupy an important place in global bird trade (Nijman, 2010), with bulbuls, starlings, mynas and robins figuring amongst the most traded species from this region (Harris, et al., 2015). As a result, two out of three species considered in the IUCN-ISSG list of 100 worst invasive species are native to Southern Asia: the red-vented bulbul (*Pycnonotus cafer*) and the common myna (*Acridotheres tristis*) (Lowe, et al., 2000). These two species historically were widely transported from India to Pacific Islands (Watling, 1978) and both are now established in New Caledonia (Brochier, et al., 2010). Our global review on the impact and management of alien red-vented bulbuls identified 37 islands in the alien distribution of this species (Thibault, et al., 2018a). This study also highlighted the lack of quantitative data and evidence-based assessments of the impacts associated with this invasive species. The red-vented bulbul was introduced into New Caledonia in 1983 (Gill, et al., 1995) and its local distribution range is currently expanding from Nouméa toward the north and south of the main island. For 25 years following its introduction into Nouméa, no studies were conducted to investigate the ecology, distribution or impacts of the species at a local scale. This lack of information has precluded any detailed assessment of the threats posed by the establishment of the red-vented bulbul in New Caledonia. Consequently, it has not thus far been possible to implement an evidence-based management strategy.

The goal of this paper is to present an overview of data from recent studies conducted in New Caledonia to describe the local status of the red-vented bulbul, its

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range expansion, and potential impacts on both the local biodiversity and agriculture. We firstly report the local distribution range of the species, the rate and nature of its range expansion, and its habitat selection and densities in different habitats. We then use diet analysis to explore potential negative effects of the red-vented bulbul on natural and agricultural systems. We present original data on an ongoing invasion process in a tropical island biodiversity hotspot and highlight priority areas for local red-vented bulbul research and risk management.

METHODS

Red-vented bulbul range expansion

Red-vented bulbul dispersal was monitored over time using static 10-min point counts combined with 2-min playback of recorded calls to increase detection probability (Ralph, et al., 1995). Points were sampled within the four hours following sunrise, between November and December in 2008, 2012, 2014 and 2016. Each point was geo-referenced, and the observers accounted for seen and heard individuals. In 2008, 136 points were sampled that covered Nouméa and suburbs as well as borders of the two main roads going to the north and south. Random points were also located in major urban areas along these roads to search for potential pioneering individuals. The method was replicated in 2012, 2014 and 2016, covering 203, 96 and 99 points respectively. Data were compiled and plotted in Qgis software version 2.18.1 (Quantum GIS Development Team, 2016).

In April 2016, we selected six additional sites across native and man-modified habitats to explore the future establishment of the red-vented bulbul in forests. We chose three sites across urban and dry forest habitats, and three across urban and wet forest habitats. These sites were located close to the core of the distribution range, where red-vented bulbul densities were highest. We placed 10 points spaced at least 250 m apart at each site (five points per habitat) and counted red-vented bulbul individuals seen and heard. The method used was the same as for distribution monitoring. Data were compiled in Qgis software version 2.18.1 (Quantum GIS Development Team, 2016) and plotted in R software version 3.4.0 (R Core Team, 2017).

Red-vented bulbul densities

Red-vented bulbul density was measured using a distance sampling method (Thomas, et al., 2010) in four sites located within the core of the red-vented bulbul distribution range. This method relies on three key assumptions: i) individuals at zero metres distance are detected with certainty, ii) individuals are detected once and at their initial location, and iii) distance measurements are exact.

Sites were selected in man-modified habitats, along a distance gradient from Nouméa to Tomo, a village located about 50 km farther north. Three transects of 1 km were established at each site and sampled between October and December 2015. A pair of observers walked along each transect for 30 minutes and counted the number of individuals seen on both sides. The distance of observed individuals from each transect was recorded with a laser telemeter. Transects were sampled three times between 0500 and 0900 hours and data from the three sessions were used independently and pooled to prevent a potential bias due to time of day.

Data were analysed with the "Distance" package (Miller, 2016) using R software version 3.4.0 (R Core Team, 2017). This method considers potential missed observations in the estimated bird densities thanks to the calculation of a detection probability curve. We first estimated the bird density at each site using data from the three sessions separately. Then, we estimated densities at each site using data of the three sessions together, considering the nine transects at each site as independent. Finally, we chose to present the estimates from the pooled dataset as it provided a smoothed estimation of densities regarding the influence of time of day on bird detection.

Red-vented bulbul diet analysis

Gut content analysis was conducted on 40 dead red-vented bulbuls provided by local hunters. There is no morphological dimorphism between male and female red-vented bulbul, so we were only able to determine the sex of sexually mature individuals, using anatomical analysis. Gastrointestinal tracts were excised and the contents removed and washed with tap water through a 0.2 mm sieve. The retained contents were placed in a Petri dish filled with 70% alcohol and examined under a dissecting microscope at 10× magnification (Olympus SZ61). Items were photographed (Toupcam UCMOS camera and Toupview software) for subsequent identification (Lopes, et al., 2005).

Fruit colour selection

According to the literature, damage to cultivated plants is the most frequently reported impact of the red-vented bulbul in its alien range (Thibault, et al., 2018a). This is also the impact category most often reported locally both by professionals (Caplong & Barjon, 2010) and non-professionals. We tackled this issue through two distinct experiments, a colour preference test and an open-field test.

We conducted an experiment on fruit colour selection to test whether the red-vented bulbul was attracted by some fruit colours more than others. We trapped eight adult individuals, maintained them in an aviary for at least a month, and in individual cages for three days. We created false-coloured fruits of four distinct colours, following the method presented in Duan & Quan, (2013). Artificial fruits were made of banana, chicken grain and water, and three quarters of the fruits were coloured with red, green and yellow food colouring. Ten fruits of each colour were placed in four different petri dishes in cages with bulbuls held individually and observed for 25 minutes from a hidden position. Each bird was tested once during five consecutive days, following either two hours or six hours of fasting. For each repetition, the colour of the first fruit eaten as well as the total number of fruit eaten per colour were recorded. ANOVA tests were conducted in R software version 3.4.0 (R Core Team, 2017) with hypothesis H0 being that each fruit colour had the same probability of being eaten first.

Damage to crops

In 2016, we conducted an open field test to explore the range of damage caused by red-vented bulbuls to tomato crops. We planted eight tomato plants inside each of 20 square plots spaced by one metre, and randomly covered half of the plots with bird netting during the flowering stage. During the fruiting period in August and September, each plot was monitored twice a week. Ripe and damaged fruit were harvested and separated in three categories; i) marketable; ii) pecked fruits; and iii) other damage. For each category, the colour, size, and sugar levels (in Brix degrees; Bates, 1942) of fruit were recorded. Tomatoes that were pecked by the birds were easily recognizable by beak marks, and the mark's size together with direct observations were used to determine the fruits that were damaged by red-vented bulbuls. The relative economic loss in marketable tomatoes due to bulbul damage was then calculated as the total weight of pecked tomatoes divided by the total weight of tomatoes harvested in 'unprotected plots'. This percentage was then extrapolated to the national production recorded during the month of our experiment. Data were analysed with the "nlme" package (Pinheiro, et al., 2017) using in R software version 3.4.0 (R Core Team, 2017).

RESULTS

Red-vented bulbul range expansion

The 2008–2016 red-vented bulbul biannual distribution map (Fig. 1) shows a continuous increase in the distribution range occupied by the red-vented bulbul in New Caledonia. Coloured polygons contain all points where red-vented bulbul individuals were observed in 2008, 2012, 2014 and 2016. Conversely, green dots represent all points where red-vented bulbul were not detected either during point-counts or during playback calls. The green triangles and diamonds represent, respectively, absence points located in natural dry forest patches within the city of Nouméa, and in humid forest, which represents the northern border of the capital and its suburbs. This absence data suggests that the species is not yet spreading into natural forest. Indeed, over the 60 point counts conducted at frontiers between urban and forest habitats, we detected red-vented bulbul individuals at 16 points in urban habitats and one point in dry forest habitat. We also received testimonies from local people about red-vented bulbul sightings. These testimonies were rarely confirmed by further observations but sometimes led to new detections. Figure 1 shows a continuous distribution of the red-vented bulbuls with range expansion particularly along main roads. It also presents absence data from another study (Thibault, et al., 2018b) which are consistent with this hypothesis. The two road axes from Nouméa to La Foa (100 km north) and Yaté (95 km south) appeared to be the main dispersal pathways. In 2012, 25 years after its introduction in the city of Nouméa, the red-vented bulbul had reached Tontouta, 42 kilometres north. From 2012 to 2016 the species travelled 35 kilometres north (Fig. 2). Nowadays the red-vented bulbul occupies at least 1,350 km^2 (8% of the New Caledonia territory), mostly restricted to the west coast of the southern province.

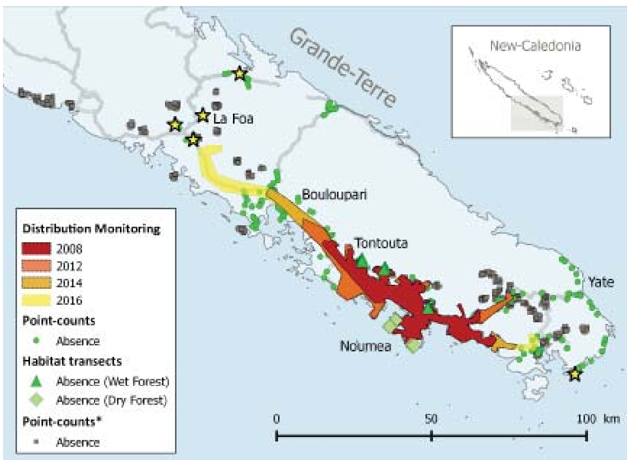

Fig. 1 Map of the expanding distribution of the red-vented bulbul between 2008 and 2016 according to the biannual monitoring. Stars represent observations from local people. Green dots represent point absence data (point counts) from the distribution monitoring. Green triangles and diamonds represent absence data (point counts along transects) in natural forests surrounding the distribution core. Grey dots show absence data (point counts) from another study (Thibault, et al., 2018b).

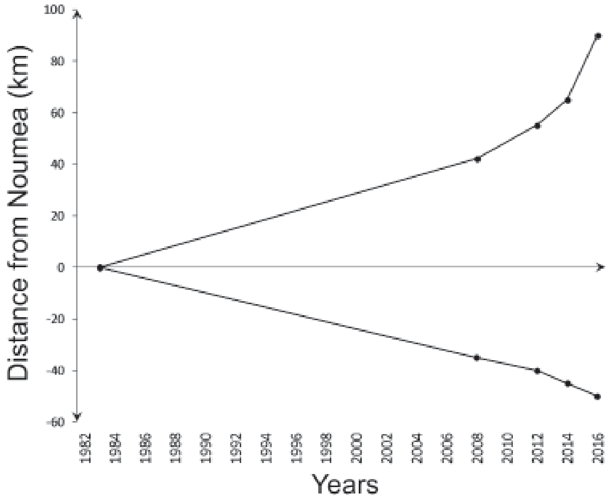

Fig. 2 Rate of red-vented bulbul dispersal toward the North and South of Nouméa.

Red-vented bulbul densities

Most birds were both heard and seen during our sampling sessions in inhabited areas. We fitted our data to a half-normal distribution (Thomas, et al., 2010) to calculate the detection function (Fig. 3**)**. Density estimates from the three sessions and from the pooled data set are presented in Fig. 4. Red-vented bulbul estimated density was six times higher in the city of Nouméa (*d*: 204 ± 23 individuals/km^2) than in the village of Tomo which is located 50 kilometres north (*d*: 31 ± 11 individuals/km^2, Table 1). Estimates from the two suburban areas, Robinson and Paita, were almost identical (*d*: 160 ± 32 individuals/km^2 and *d*: 131 ± 18 individuals/km^2, respectively). The density estimates are corrected by a detection function curve which represents the probability of an observer detecting a red-vented bulbul depending on its distance from the transect. In the four urban habitats we sampled, the average probability of detecting a red-vented bulbul was 50% when the bird was approximately 25 metres from the observer.

Red-vented bulbul diet analysis

We extracted and analysed the gut contents of 40 red-vented bulbuls. Results of the diet study are presented in

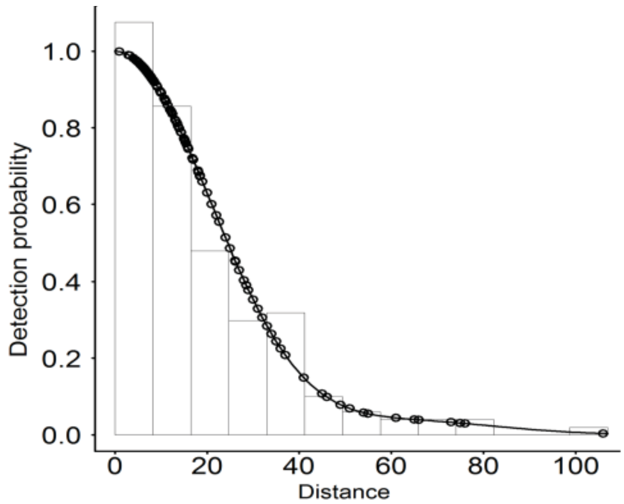

Fig. 3 Probability of detecting a red-vented bulbul individual as a function of distance from the transect in inhabited areas.

Table 1 Sampling statistics and density estimates at four urban sites within the current distribution range of the red-vented bulbuls according to distance from the introduction point. (n) total number of individuals over the three sessions.

Site	Distance (km)	Habitat	Area (m2)	n	Density estimate (ind/km2)	Standard error
Nouméa	0	urban	787.3	117	204	± 23
Paita	10	suburban	816.3	66	160	± 32
Robinson	25	suburban	492.8	65	131	± 18
Tomo	50	rural	993.3	15	31	± 11

Table 2 Occurrences (n) and frequency (%) of food items identified in the gut content of 40 bulbul individuals.

	n	F (n=40)
Fruit parts		
Whole fruit	16	40
Seeds	22	55
Fruit skin	7	17.5
Fruit flesh	17	42.5
Plant families	33	82.5
Myrtaceae	20	50
Passifloraceae	1	2.5
Sapindaceae	2	5
Solanaceae	4	10
Insects	22	55
Coleoptera	8	20
Diptera	1	2.5
Hemiptera	13	32.5
Hymenoptera	3	7.5
Odonata	1	2.5

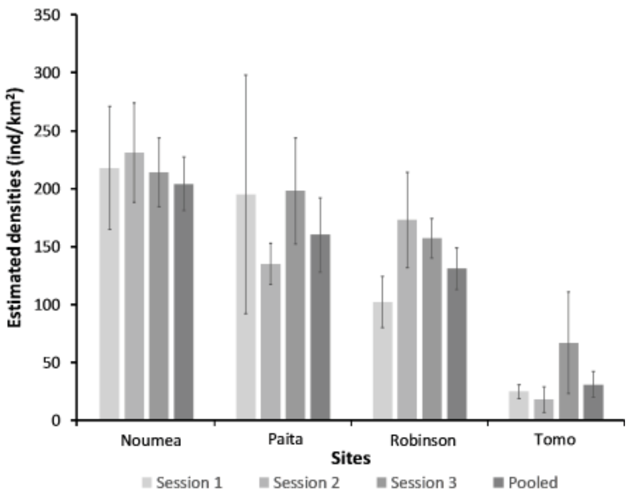

Fig. 4 Densities of red-vented bulbuls at each site calculated from the three sampling sessions, and from the pooled dataset.

Table 2. Mean weight of mature individuals was 38.3 ± 4.9 g for females and 44.1 ± 5.6 g for males. We found plant remains in the gut content of 33 individuals (82.5%) and animal items in 22 (55%). Among plant items, seeds (55%) and fruit flesh (42.5%) were the most frequent. The most frequent plant family was Myrtaceae (20 individuals), and the most consumed insect orders were Hemiptera (13 individuals) and Coleoptera (8 individuals). Identification of the remains highlighted the consumption of one endemic plant species (*Myrtastrum rufopunctatum*), two cultivated species (*Syzygium cumini* and *Lichi chinensis*) and two invasive alien species (*Passiflora foetida* and *Solanum torvum).* Exoskeleton parts from cicada individuals were frequent in this sample (F=32.5%). No vertebrate remains were found during this analysis.

Fruit colour selection

Colour selection tests were replicated 102 times. The first pecked fruit was red in 77% of samples, followed by green (10% of samples). The average number of consumed fruits per colour is presented in Fig 5. Red fruits were the most often consumed (5 ± 0.3), and yellow ones were consumed five times less often (0.9 ± 0.16). Colour explained the consumption of fruits significantly (ANOVA: F: 8.3; p<0.001). In our analysis, fasting period did not contribute to explain the choice of coloured fruits (ANOVA: F: 2.7; p=0.1).

Damage to crops

On our 20 plots, we produced a total of 2,310 tomatoes (345.5 kg). Unfortunately, three plots with nets were damaged by feral dogs just before the beginning of the fruiting season, and were thus considered to be unprotected. Red-vented bulbuls were the only birds that fed on tomato fruit during the experiment. Results are presented in Fig 6. On average, production per plot was homogenous in net-protected pots (18.5 ± 2.1 kg) compared to 'unprotected' ones (16.6 ± 2.3 kg). Losses due to bird damage were recorded almost exclusively in 'unprotected' plots and corresponded to 2.95 ± 0.24 kg per plot (17.5%), as only three tomatoes were pecked at the edge of protected plots (0.5% in weight). These losses were similar to those caused by other pests: 2.63 ± 0.3 kg in unprotected plots, and 3.9 ± 0.3 kg in protected plots. Pecked fruits were mainly red (ANOVA F: 7.6; p=0.009), between 50 and 70 mm in size and with high sugar levels (5°Bx, ANOVA: 5.95; p=0.016). Considering that 34 tons of tomato were sold at 3.18 USD/kg in September 2016 in New Caledonia, the 17.5% loss we recorded because of bird damage would have corresponded to an economic loss of approximately $18,355 USD for September 2016 alone.

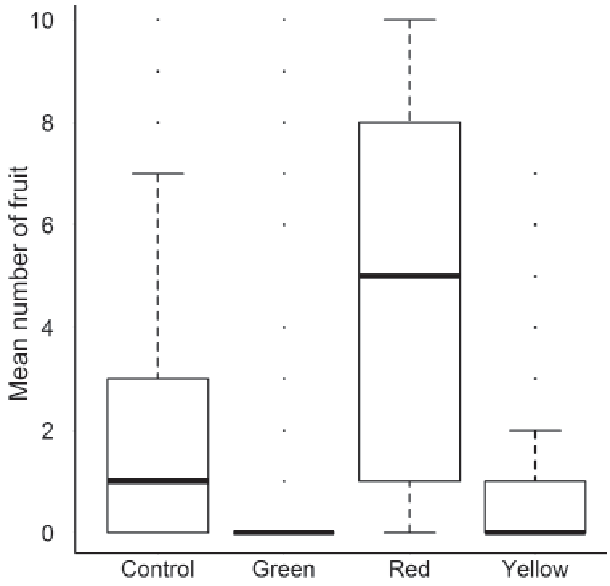

Fig. 5 Result of 102 colour preference tests with red-vented bulbuls. The y-axis represents the average number of fruits consumed by tested individuals during one session.

DISCUSSION

Dispersal along urban corridors

The red-vented bulbul has continuously increased its distribution range in New Caledonia since its introduction 25 years ago. The distribution map suggests that roads and urban habitats are the main dispersal pathways for the species. The dispersal rates we estimated were different depending on the direction. One reason for this may be differences in habitat to the north and south of Nouméa. The south of the main island of New Caledonia is dominated by ultramafic soils and the dominant vegetation type is the "maquis minier", a shrubland characterised by xerophytic plants (Jaffré, et al., 2003; Jaffré, et al., 2004) which may be less attractive in terms of food source for red-vented bulbuls. Considering the dispersal speed, we know that the red-vented bulbul's range expanded 40 kilometres toward the north of its introduction point in 25 years. Its range expansion then increased more quickly, extending a further 35 kilometres in just four years. This is consistent with findings of Aagaard & Lockwood (2014) on growth lag in alien bird populations and suggests that this range expansion could continue to accelerate. Our observation of a lagged expansion in the red-vented bulbul could thus be explained both by a demographic time-lag, inter-specific relationships, or by the carrying capacity of the different habitats.

Study of red-vented bulbul occurrence at the frontiers between urban and forest habitats confirmed the association of the species with man-modified habitats. Our results suggest that the red-vented bulbul is not spreading from invaded urban areas into either dry forest patches or into native rainforests. This is consistent with previous observations of Watling (1979) in Fiji. However, in Tahiti (French Polynesia) the red-vented bulbul is able to colonize native tropical forests with major impacts on native avifauna (Blanvillain, et al., 2003). Further monitoring of the distribution is thus crucial to anticipate potential shifts in the habitat occupancy and resulting threats on forest bird communities. A specific effort could be dedicated by managers to prevent future establishment of pioneer individuals out of the current range, toward the north, the Loyalty Islands or specific areas of high conservation/agricultural value. Quick detection coupled with control actions at the edges of the red-vented bulbul range will reduce the colonisation speed and prevent future negative effects.

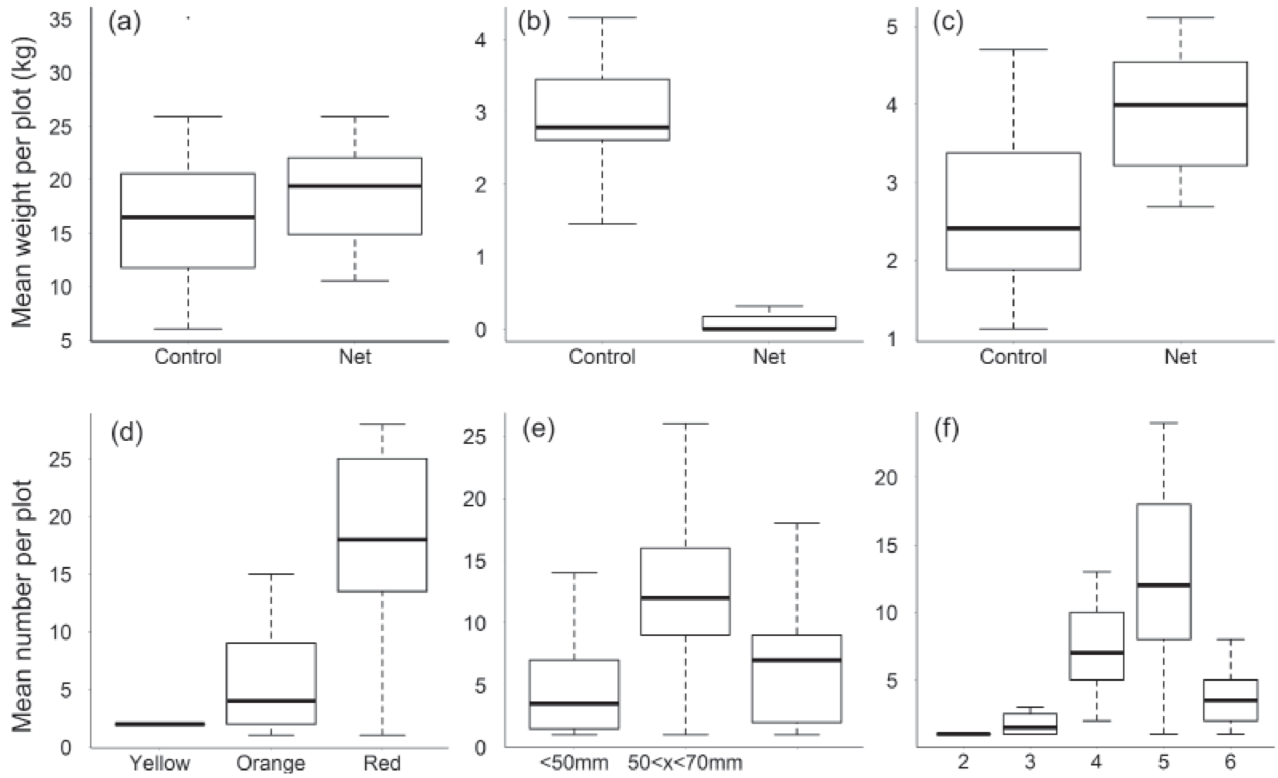

Fig. 6 Result of the open field test conducted with tomato plants. (a) Represents the global production weights for "net-protected" and "unprotected" treatments, (b) and (c) represent the mean weight of fruit damaged per plot by birds and other pests. (d), (e) and (f) represent the average number of damaged fruit depending on fruit colour, size, and sugar content, respectively.

Density gradient

Dispersal of bird species is partly related to population densities (Matthysen, 2005), so the anticipation of future dispersal events may be facilitated by the knowledge of bird density in specific locations. Our density estimates showed a density gradient in the red-vented bulbul, depending on the degree of urbanisation and the distance from the introduction point. This has been frequently observed in alien bird populations (Chace & Walsh, 2006). The density level we estimated in the rural village of Tomo was similar to those reported by Radhakrishanan & Asokan (2015) in two villages of the Cauvery delta region in Southern India, to which the red-vented bulbul is native. However, our estimates for the centre of Nouméa and suburbs are similar to those found for common bird species in European/American urban centres (Clergeau, et al., 1998). High bird densities in urban habitats are often associated with low bird-community species richness (Matthysen, 2005). Regarding its density, the red-vented bulbul is already a predominant species in Nouméa. Monitoring the change in red-vented bulbul densities over time will contribute to a better understanding of the species' dynamics. It will also allow the estimation of the density-impact relationship in further management programmes (Yokomizo, et al., 2009), as management of invasive alien species populations often relies on abundance/density reductions (Genovesi, 2005; Simberloff, et al., 2005). For example, control operations could be feasible at low densities, whereas mitigation of specific impacts could be more cost-efficient at high-density levels.

Predation and frugivory

Results of the diet analysis were consistent with previous observations elsewhere in both the alien and native range of the species (Watling, 1978; Bhatt & Kumar, 2001; Brooks, 2013 Bates, et al., 2014). The diet comprised mostly fruits and a significant part of animal remains. We observed several red-vented bulbul individuals feeding on house geckos (*Hemidactylus frenatus*) and skinks in the field, but we did not find any reptile or gastropod remains in the gut contents we analysed. Such food items have been reported in the red-vented bulbul native range (Bhatt & Kumar, 2001). Much of the gut contents we analysed (n=13, F=32.5%) contained remains from cicadas. Considering the periodic lifecycle of these insects (May, 1974), this observation suggests that red-vented bulbuls can adapt their diet to this temporary resource. Levels of endemism are high in New-Caledonia, with approximately 92% of reptiles and nearly 100% of cicadas (Smith, et al., 2007; Grandcolas, et al., 2008; Delorme, et al., 2016) being endemic. Predation by alien species such as the red-vented bulbul could thus represent an additional threat for these species of high conservation value.

Seeds and whole fruits were found in 50% of individuals. This observation emphasizes the red-vented bulbul's capacity to participate in seed dispersal, particularly in association with invasive alien plant species like *Miconia calvescens* or *Lantana camara* (Meyer, 1996; Spotswood, et al., 2012; Spotswood, et al., 2013). In our diet study, we identified several candidates for red-vented bulbul-mediated dispersal. Most of them were invasive or cultivated species, but we also identified one endemic (*Myrtastrum rufopunctatum*) that is used for mining-site restoration (Lemay, et al., 2009). Consumption of native species by the red-vented bulbul could result in a service, by improving the dispersal capacity of some species (Kawakami, et al., 2009). However, it can also lead to competitive interactions with native avifauna (Sherman & Fall, 2010; Thibault, et al., 2018b) which can turn into a conservation issue (Blanvillain, et al., 2003). New Caledonia is considered a biodiversity hotspot (Myers, et al., 2000) thanks to its plant diversity, with 3060 species of flowering plants recorded (78% endemic; Munzinger, et al., 2016). Exploring variations in the red-vented bulbul diet over different habitat, seasons and maturity stages will contribute to better prediction of the dispersal of both alien and native plants as well as potential negative interactions with endemic species. At a wider scale, such quantitative and qualitative data will contribute to the assessment of impacts caused by red-vented bulbuls (Thibault, et al., 2018a).

Colour selection and damages on crops

Diet and preference for specific resources plays a key role in impacts caused by vertebrate pest species (Herrero, et al., 2006; Gebhardt, et al., 2011). Sometimes, these preferences can be strong enough to aid bait selection for both hunters and environment managers. In our experiment, the red-vented bulbul preferred red, consistent with colour preference in the red-whiskered bulbul (*Pycnonotus jocosus*) (Duan & Quan, 2013). In a French Polynesian study, authors concluded that preference may sometimes be stronger than abundance in fruit selection by birds, including the red-vented bulbul (Spotswood, et al., 2013).

Such preference for specific fruits implies that red-vented bulbuls are likely to disperse or damage the fruit of some species more than others, and that predictions can be made about species that are likely to be most vulnerable. Observations made during our open field experiment were consistent with this hypothesis, with red tomatoes being damaged more than orange or yellow ones. In unprotected plots, damage caused by birds was equivalent to that of all the parasites and corresponded to 17.5% loss in weight of marketable fruit. This corresponds to the average losses presented in Oerke (2006) in their global estimation of economic losses due to animal pests over 11 production types including tomato, between 2001 and 2003. In this study, recorded losses attributed to animal species and other pathogens on unprotected crops were of 18% and 15%, respectively. Oerke suggested that pest control operations allowed a 39% reduction in losses due to animal pests. Here we showed that protecting tomato plants with nets efficiently protected 99% of fruits, reducing by 97% the loss in weight of marketable fruit. This early assessment of colour selection and damage on production suggests that red and sweet fruit/flower crops could be more sensitive to red-vented bulbul damages. Such information is already used in the development of trapping systems dedicated to this species. Indeed, fruit and fresh vegetables represented 5115 and 6292 tons, respectively, of production in New Caledonia in 2012, corresponding to 25% and 30% of the total plant production that year (ISEE, 2012). The red-vented bulbul is currently restricted to suburban areas in a limited range, but up to 35% loss has already been recorded on fruit production there (Caplong & Barjon, 2010). Future establishment of the species in cultivated areas of the main island could thus represent an additional risk to crop productivity.

CONCLUSION

The global distribution and population trends of red-vented bulbul have been poorly reported, relative to many other tropical invasive birds. The potential overlap in the impacts associated with tropical passerine species from south Asia, suggested by Kumschick et al. (2015), has not been explored either. Authors have claimed that introduced populations of red-vented bulbuls were harmless (Watling, 1979), while in other locations their role in noxious seed dispersal (Meyer, 1996), competition with native birds

(Blanvillain, et al., 2003, Thibault, et al., 2018b) and damage to crops (Walker, 2008) was suggested. New Caledonia must deal with the current dispersal of this species on its territory with only a few quantitative data available from the literature (Thibault, et al., 2018a). However, the establishment, on-going dispersal, and impacts of the red-vented bulbul deserve attention from conservation biologists, environment managers and local people. Perceptions of this invasive species differ across groups of people (Fischer, et al., 2014), but a coordinated joint effort is required to improve our knowledge of invasion mechanisms for the red-vented bulbul in the New Caledonia archipelago. New Caledonia recently produced a list of priority invasive species for management actions, and the studies we presented here contributed to the consideration of the red-vented bulbul among the six species on this list.

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REFERENCES

Aagaard, K. and Lockwood, J. (2014). 'Exotic birds show lags in population growth'. *Diversity and distributions* 20(5): 547–554.

Bates, F.J. (1942). *Polarimetry, Saccharimetry and the Sugars*. Washington, DC: National Bureau of Standards of the US Government.

Bates, J.H., Spotswood, E.N. and Russell, J.C. (2014). 'Foraging behaviour and habitat partitioning in sympatric invasive birds in French Polynesia'. *Notornis* 61: 35–42.

Bellard, C., Leclerc, C., Leroy, B., Bakkenes, M., Veloz, S., Thuiller, W. and Courchamp, F. (2014). 'Vulnerability of biodiversity hotspots to global change'. *Global Ecology and Biogeography* 23(12): 1376–1386.

Bhatt, D. and Kumar, A. (2001). 'Foraging ecology of red-vented bulbul *Pycnonotus cafer* in Haridwar, India'. *Forktail* 17: 109–110.

Blanvillain, C., Salducci, J.M., Tutururai, G. and Maeura, M. (2003). 'Impact of introduced birds on the recovery of the Tahiti flycatcher (*Pomareanigra*), a critically endangered forest bird of Tahiti'. *Biological Conservation* 109(2): 197–205.

Brochier, B., Vangeluwe, D. and Van den Berg, T. (2010). 'Alien invasive birds'. *Revue scientifique et technique* 29(2): 217–226.

Brooks, D.M. (2013). 'Ecology, behavior, and reproduction of an introduced population of red-vented bulbuls (*Pycnonotus cafer*) in Houston, Texas'. *Wilson Journal of Ornithology* 125: 800–808.

Cabral, J.S., Wiegand, K. and Kreft, H. (2017). 'Interactions between ecological, evolutionary, and environmental processes unveil complex dynamics of island biodiversity'. *bioRχiv*. doi :10.1101/099978.

Caplong, P. and Barjon, F. (2010). 'Le Bulbul à ventre rouge, une star méconnue en Nouvelle Caledonie'. *La Caledonie Agricole* 123: 22–25.

Cardador, L., Lattuada, M., Strubbe, D., Tella, J.L., Reino, L., Figueira, R. and Carrete, M. (2017). 'Regional bans on wild-bird trade modify invasion risks at a global scale'. *Conservation Letters*. doi:10.1111/conl.12361.

Chace, J.F. and Walsh, J.J. (2006). 'Urban effects on native avifauna: a review'. *Landscape and urban planning* 74(1): 46–69.

Clergeau, P., Savard, J.P.L., Mennechez, G. and Falardeau, G. (1998). 'Bird abundance and diversity along an urban-rural gradient: a comparative study between two cities on different continents'. *Condor* 100(3): 413–425.

Davis, M.A. and Chew, M.K. (2017). "The denialists are coming!' Well, not exactly: A response to Russell and Blackburn'. *Trends in Ecology and Evolution* 32(4): 229–230.

Delorme, Q., Mille, C. and Jourdan, H., (2016). 'A review of the genus *Kanakia* Distant, 1892 (Insecta: Hemiptera, Cicadoidea, Cicadidae) from New Caledonia'. *Zootaxa* 4092: 301–338. doi:10.11646/zootaxa.4092.3.1.

Duan, Q. and Quan, R.C. (2013). 'The effect of color on fruit selection in six tropical Asian birds'. *The Condor* 115(3): 623–629.

Dyer, E.E., Redding, D.W. and Blackburn, T.M. (2017). 'The global avian invasions atlas, a database of alien bird distributions worldwide'. *Scientific Data* 4: doi:10.1038/sdata.2017.41.

Early, R., Bradley, B.A., Dukes, J.S., Lawler, J.J., Olden, J.D., Blumenthal, D. M., Gonzales, P., Grosholz, E.D., Ibañez, I., Miller, L.P., Sorte, C.J.B., and Tatem, A.J. (2016). 'Global threats from invasive alien species in the twenty-first century and national response capacities'. *Nature Communications* 7: doi:10.1038/ncomms12485.

Fischer, A., Selge, S., van der Wal, R. and Larson, B.M. (2014). 'The public and professionals reason similarly about the management of non-native invasive species: a quantitative investigation of the relationship between beliefs and attitudes'. *PLOS one* 9(8): e105495.

Garcia, R.A., Cabeza, M., Rahbek, C. and Araújo, M.B. (2014). 'Multiple dimensions of climate change and their implications for biodiversity'. *Science* 344(6183): 1247579.

Gerard, A., Jourdan, H., Millon, A. and Vidal, E. (2016). 'Knocking on heaven's door: Are novel invaders necessarily facing naïve native species on islands?'. *PLOS one* 11(3): e0151545.

Gebhardt, K., Anderson, A.M., Kirkpatrick, K.N. and Shwiff, S.A. (2011). 'A review and synthesis of bird and rodent damage estimates to select California crops'. *Crop Protection* 30(9): 1109–1116.

Genovesi, P. (2005). 'Eradications of invasive alien species in Europe: a review'. *Biological Invasions* 7: 127–133.

Gill, B.J., Hunt, G.R. and Sirgouant, S. (1995). 'Red-vented bulbuls (*Pycnonotuscafer*) in New Caledonia'. *Notornis* 42: 214–215.

Grandcolas, P., Murienne, J., Robillard, T., Desutter-Grandcolas, L., Jourdan, H., Guilbert, E. and Deharveng, L. (2008). 'New Caledonia: a very old Darwinian island?'. *Philosophical Transactions of the Royal Society of London B: Biological Sciences* 363(1508): 3309–3317.

Haddad, N.M., Brudvig, L.A., Clobert, J., Davies, K.F., Gonzalez, A., Holt, R.D., Lovejoy, T.E., Sexton, J.O., Austin, M.P., Collins, C.D., Cook, W.M., Damschen, E.I., Ewers, R.M., Foster, B.L., Jenkins, C.N., King, A.J., Laurance, W.F., Levey, D.J., Margules, C.R., Melbourne, B.A., Nicholls, A.O., Orrock, J.L., Song, D. and Townshend, J.R. (2015). 'Habitat fragmentation and its lasting impact on Earth's ecosystems'. *Science Advances* 1(2): e1500052.

Harris, J.B.C., Green, J.M., Prawiradilaga, D.M., Giam, X., Hikmatullah, D., Putra, C.A. and Wilcove, D.S. (2015). 'Using market data and expert opinion to identify overexploited species in the wild bird trade'. *Biological Conservation* 187: 51–60.

Herrero, J., García-Serrano, A., Couto, S., Ortuño, V.M. and García-González, R. (2006). 'Diet of wild boar *Sus scrofa* L. and crop damage in an intensive agroecosystem'. *European Journal of Wildlife Research* 52(4): 245–250.

ISEE (2012). '*Bilan economique et social Nouvelle Calédonie'*. Nouméa, New Caledonia: ISEE - Bilan economique et social, pp. 93–40.

Jaffré, T., Dagostini, G. and Rigault, F. (2003). '*Identification, typologie et cartographie des groupements végétaux de basse altitude du Grand Sud Calédonien et de la vallée de la Tontouta'*. Nouméa, New Caledonia: ORSTOM.

Jaffré, T., Dagostini, G., Rigault, F. and Coic, N. (2004). *'Inventaire Floristique des Unités de Végétation de la Zone d'Implantation des Infrastructures Minières et Industrielles de Goro Nickel'*. Nouméa, New Caledonia: Institut de Recherche pour le Developpement.

Kawakami, K., Mizusawa, L. and Higuchi, H. (2009). 'Re-established mutualism in a seed-dispersal system consisting of native and introduced birds and plants on the Bonin Islands, Japan'. *Ecological Research* 24(4): 741–748.

Kumschick S., Blackburn T.M. and Richardson D.M. (2015). 'Managing alien bird species: Time to move beyond '100 of the worst' lists?' *Bird Conservation International* 26(2): 1–10.

Lemay, V., Gâteblé, G., and McCoy, S. (2009). 'Vegetative propagation of two endemic species of *Cloezia* Brongn. & Gris for conservation and mining revegetation activities in New Caledonia'. *New Forests* 37(1) : 1–8.

Lopes, L.E., Fernandes, A.M. and Marini, M.Â. (2005). 'Diet of some Atlantic forest birds'. *Ararajuba* 13(1): 95-103.

Lowe, S., Browne, M., Boudjelas, S. and De Poorter, M. (2000) '*100 of the World's Worst Invasive Alien Species A Selection from the Global Invasive Species Database'*. Auckland, New Zealand: The Invasive Species Specialist Group (ISSG).

Matthysen, E. (2005). 'Density-dependent dispersal in birds and mammals'. *Ecography* 28(3): 403–416.

May, R.M. (1974). 'Biological populations with nonoverlapping generations: stable points, stable cycles, and chaos'. *Science* 186(4164): 645–647.

Meyer, J-Y. (1996). 'Status of *Miconia calvescens* (Melastomataceae), a dominant invasive tree in the Society Islands (French Polynesia)'. *Pacific Science* 50: 66–79.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G.A. and Kent, J. (2000). 'Biodiversity hotspots for conservation priorities'. *Nature* 403(6772): 853–858.

Miller, D.L. (2016). *Distance: Distance Sampling Detection Function and Abundance Estimation*. R package version 0.9.6. <https://CRAN.R-project.org/package=Distance>.

Munzinger, J., Morat, P., Jaffré, T., Gâteblé, G., Pillon, Y., Tronchet, F., Veillon, J-M. and Chalopin, M. (2016). *FLORICAL: Checklist of the vascular indigenous flora of New Caledonia. vers. 22.IV.2016*. <http://www.botanique.nc/herbier/florical>.

Nijman, V. (2010). 'An overview of international wildlife trade from Southeast Asia'. *Biodiversity and conservation* 19(4): 1101–1114.

Oerke, E.C. (2006). 'Crop losses to pests'. *The Journal of Agricultural Science* 144(01): 31–43.

Pinheiro, J., Bates, D., DebRoy, S., Sarker, D. and R Core Team (2017). *nlme: Linear and Nonlinear Mixed Effects Models*. R package version 3.1–131. 2017.

Quantum GIS Development Team (2016). *Quantum GIS Geographic Information System*. Open Source Geospatial Foundation Project. <http://qgis.osgeo.org>.

R Core Team (2017). *R: A language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing. <https://www.R-project.org/>.

Radhakrishanan, P. and Asokan, S. (2015). 'Population density of red-vented bulbul *Pycnonotus cafer* in a portion of Cauvery Delta Region, Southern India'. *Journal of Life Sciences Research* 2(1): 1–4.

Ralph, C.J., Sauer, J.R. and S. Droege. (1995). 'Managing and monitoring bird populations using point counts: standards and applications'. In: C.J. Ralph, J.R. Sauer and S. Droege (eds.) *Monitoring Bird Populations by Point Counts*, pp. 161–168. USDA Forest Service General Technical, USA: Report PSW-GTR-149.

Ricciardi, A., Blackburn, T.M., Carlton, J.T., Dick, J.T., Hulme, P.E., Iacarella, J.C., and Pyšek, P. (2017). 'Invasion science: A horizon scan of emerging challenges and opportunities'. *Trends in Ecology and Evolution* 32(6): 464–474.

Russell, J.C. and Blackburn, T.M. (2017). 'The rise of invasive species denialism'. *Trends in Ecology and Evolution* 32(1): 3–6.

Russell, J.C., Meyer, J.Y., Holmes, N.D. and Pagad, S. (2017). 'Invasive alien species on islands: impacts, distribution, interactions and management'. *Environmental Conservation*: doi:10.1017/S0376892917000297.

Sherman, J. and Fall, P. (2010). 'Observations on feeding frequencies among native and exotic birds and fruit bats at *Erythrina variegata* and *Dysoxylum* trees on American Samoa'. In: S. Haberle, J. Stevenson, and M. Prebble (eds.) *Altered Ecologies: fire, climate and human influence on terrestrial landscapes*, pp.101–116. Canberra, Australia: Australian National University Press.

Simberloff, D., Parker, I.M., and Windle, P.N. (2005). 'Introduced species policy, management, and future research needs'. *Frontiers in Ecology and the Environment* 3(1): 12–20.

Smith, S.A., Sadlier, R.A., Bauer, A.M., Austin, C.C. and Jackman, T. (2007). 'Molecular phylogeny of the scincid lizards of New Caledonia and adjacent areas: evidence for a single origin of the endemic skinks of Tasmantis'. *Molecular Phylogenetics and Evolution* 43(3): 1151–1166.

Spotswood, E.N., Meyer, J-Y. and Bartolome, J.W. (2012). 'An invasive tree alters the structure of seed dispersal networks between birds and plants in French Polynesia'. *Journal of Biogeography* 39(11): 2007–2020.

Spotswood, E.N., Meyer, J-Y. and Bartolome, J.W. (2013). 'Preference for an invasive fruit trumps fruit abundance in selection by an introduced bird in the Society Islands, French Polynesia'. *Biological invasions* 15(10): 2147–2156.

Su, S., Cassey, P. and Blackburn, T.M. (2016). 'The wildlife pet trade as a driver of introduction and establishment in alien birds in Taiwan'. *Biological invasions* 18(1): 215–229.

Thibault, M., Vidal, E., Potter, M.A. and Brescia, F. (2018a). 'The red-vented bulbul (*Pycnonotus cafer*): serious pest or understudied invasive?'. *Biological invasions* 20(121): 121–136.

Thibault, M., Vidal, E., Potter, M.A., Sanchez, T. and Brescia F. (2018b). 'The invasive red-vented bulbul (*Pycnonotus cafer*) outcompetes native birds in a tropical biodiversity hotspot'. *PLOS one* 13(2): e0192249.

Thomas, L., Buckland, S.T., Rexstad, E.A., Laake, J.L., Strindberg, S., Hedley, S.L., Bishop, J.R.B., Marques, T.A. and Burnham, K.P. (2010). 'Distance software: design and analysis of distance sampling surveys for estimating population size'. *Journal of Applied Ecology* 47(1): 5–14.

Walker, R. (2008). 'The red-vented bulbul, superbird?'. '*Elepaio* 68: 71–78.

Watling, D. (1978). 'Observation on the naturalised distribution of the red-vented bulbul in the Pacific, with special reference to the Fiji Islands'. *Notornis,* 25: 109–117.

Watling, D. (1979). 'The bulbul gets a clean bill'. *New Scientist* 81: 963–965.

Williams, K., Ford, A., Rosauer, D.F., De Silva, N., Mittermeier, R., Bruce, C. and Larsen, F.W. (2011). 'Forests of East Australia: The 35th biodiversity hotspot'. In: F.E. Zachos and J.C. Habel (eds.) *Biodiversity Hotspots,* pp. 295–310. Berlin Heidelberg: Springer**.**

Yokomizo, H., Possingham, H.P., Thomas, M.B. and Buckley, Y.M. (2009). 'Managing the impact of invasive species: the value of knowing the density–impact curve'. *Ecological Applications* 19(2): 376–386.

Population assessment of a novel island invasive: tegu (*Salvator merianae*) of Fernando de Noronha

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ABSTRACT Fernando de Noronha is an oceanic archipelago in the Atlantic Ocean, 345 km offshore from the Brazilian coast. It comprises 21 islands and islets, of which the main island (FN) is 17 km^2 with a rapidly growing tourism industry in the last decades. Despite being a protected area and bearing Ramsar and UNESCO World Heritage site status, it is threatened by multiple terrestrial invasive species since its colonisation in the early 16th century. Invasive species and the increasing tourism contributes to a list of at least 15 endangered or critically endangered species according to IUCN criteria. The black and white tegu (*Salvator merianae*) is the largest lizard in South America, occurring in most of the Brazilian territory and reaching up to 8 kg and 1.6 m from head to tail. As an omnivorous and opportunistic lizard, it feeds on a variety of available items, including smaller vertebrates and eggs. The introduction of the tegu to FN as well as its immediate impact on local fauna were not recorded; however, its ongoing impact is expected to be high. We captured and marked 103 tegu in FN during the months of February and November of 2015 and 2016. We also counted animals by line-transect census in a sparsely inhabited and an uninhabited area of FN. Body size affected the capture probabilities, while season and sex had little or no effect. Densities estimated by capture-recapture in the sparsely inhabited area varied from 2.29 to 8.28 animals/ha according to sampling season. Line transect census in the same area revealed a density of 3.98 (±1.1) animals/ha and in the uninhabited area 13.83 (±3.9) animals/ha. Home range was 10.54 ha, ranging from 7.36 to 15.33 hectares. Tegu activity decreased in the months of July and August of 2015. Results from this study can assist conservation managers and decision makers to implement a science-based tegu management programme in the future.

Keywords: conservation, invasive species, lizard, oceanic island, reptile, *Salvator*, Teiidae, *Tupinambis*

INTRODUCTION

Islands are simplified ecosystems where each species plays an important role in its functioning (Simberloff, 1974). In these environments, the loss of a species and its functional role are not easily replaced, as would be the case in more species-rich ecosystems such as on continents. Despite corresponding to only about 5% of land area globally, islands contain more than 15% of terrestrial biodiversity (Tershy, et al., 2015). A lack of certain behaviour or life-history traits makes native insular species more vulnerable to the impacts of invasive species (Vitousek, 1988; Tershy, et al., 2015).

Introduction of invasive species is one of the major causes of contemporary biodiversity loss (Vitousek, et al., 1997; Chapin, et al., 2000). On islands, it is probably the major cause (Veitch & Clout, 2002; Reaser, et al., 2007). Direct and indirect competition, predation and introduction of diseases are some of the negative influences that invasive species can bring to native populations (Wyatt, et al., 2008; McCreless, et al., 2016; Russell, et al., 2017). Invasive predators are implicated in at least 58% of the worldwide contemporary extinctions for birds, mammals and reptiles (Doherty, et al., 2016). The insular ecosystem frailty combined with invasive species results in islands bearing 37% of all critically endangered species and 61% of all recorded extinct species, according to the IUCN Red List (Tershy, et al., 2015). Furthermore, the impact of invasive species is not constrained to local biodiversity, but also affects the economy, agriculture, health and human culture (Russell, et al., 2017)

Some invasive species, such as rodents, are globally widespread and their impacts on islands have been well described (Reaser, et al., 2007; Russell, et al., 2017). However, some invasive predators are only found regionally or locally and their impacts and management are not fully understood (see Eales, et al., 2010; Powell, et al., 2011). Those less well-known species must not be overlooked, as their impact might be equal to, if not larger than, common widespread invaders (Phillips, et al., 2007; Simberloff, 2009; Dorcas, et al., 2012; Neves, et al., 2017; Russell, et al., 2017).

Fernando de Noronha

Fernando de Noronha archipelago consists of 21 islands and islets, 340 km offshore from the northeast Brazilian coast. The total land area of the archipelago is 18 km^2 where the main island, also called Fernando de Noronha (FN) is about 16.7 km^2. The archipelago is a UNESCO world heritage site (since 2001) and has recently been named as a Ramsar site. Fernando de Noronha archipelago is an important breeding site for several species of birds, sea turtles and reptiles, some endemic and threatened with extinction (Sazima & Haemig, 2012; Reis & Hayward, 2013). At the moment, at least 22 invasive species of plants and animals are known in the archipelago (Sampaio & Schmidt, 2014).

The local economy is fundamentally based on tourism, with minimal production of goods and other services. The number of inhabitants on FN has increased substantially within the last decade due to a lack of control from Pernambuco State and the opportunities created by the growing tourism (Gasparini, et al., 2007). The total number of human inhabitants is debateable, with available information varying from two to five thousand people, with an additional up to three thousand tourists per year in the peak seasons (Andrade, et al., 2009; Marinho, 2016; IBGE, 2017; Pernambuco, 2017).

Urbanised areas are restricted to the main island, in the environmental protected area (APA), a protected area with sustainable use of natural resources – IUCN category

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VI – of approximately 8 km^2. The remainder of the main island, including the other islands and islets from the archipelago, is uninhabited and constitutes the National Park (PARNAMAR), where only indirect use is permitted – IUCN category II.

Tegu lizard

The black and white tegu lizard (*Salvator merianae* syn. *Tupinambis merianae*) (Fig. 1), hereby referred to as tegu, is the largest lizard in South America, up to 160 cm in total length and weighting up to 8 kg in its native range (Lopes & Abe, 1999; Andrade, et al., 2004). In their natural distribution in South America, tegu are commonly seen living and feeding close to inhabited areas, as well as forested areas (Oren, 1984; Sazima & Haddad, 1992; Bovendorp, et al., 2008; Winck, et al., 2011; Klug, et al., 2015; Muscat, et al., 2016). This omnivorous, opportunistic species feeds on fruits, vegetables, insects, small vertebrates, garbage and even carcasses when available (Sazima & Haddad, 1992; Kiefer & Sazima, 2002; Manes, et al., 2007; Bovendorp, et al., 2008; da Silva, et al., 2013; Muscat, et al., 2016). In South America, they can be found from south of the Amazon River to Argentina (Presch, 1973; Lanfri, et al., 2013; Passos, et al., 2013). In most areas where the tegu occurs, they are hunted for their skin and meat (Oren, 1984; Alves, et al., 2012), which has warranted the inclusion of the species on the CITES II appendix (UNEP-WCMC, 2014). In South America, adult females can lay up to 54 eggs per year (Donadio & Gallardo, 1984) and in captivity this species can possibly live up to 20 years (Brito, et al., 2001). The tegu is also considered an invasive species in Florida, where it is suspected to have a large impact on the already impacted local fauna (Pernas, et al., 2012; Mazzotti, et al., 2015).

Available data indicate that the tegu was deliberately introduced to the main island of Fernando de Noronha at the beginning of the 20th century (Santos, 1950), despite other publications suggesting a different period of introduction (e.g. Oren, 1984; Silva-Jr., et al., 2005). Whether to serve as hunting game or to help the control of rodents and toads, reasons for the introduction of tegu are speculative (Oren, 1984; Gasparini, et al., 2007; Ramalho, et al., 2009). Descriptions of FN fauna prior to the 20th century don't mention the tegu, despite mentioning the other endemic reptiles on the archipelago (Branner, 1888; Ridley, 1890). In the last century, very little was done to study the tegu population and impacts on the island ecosystem. Control or eradication methods were also never attempted, despite the management of the tegu being considered important to promote the conservation of endangered species living on the island (Brasil, 2004).

Fig. 1 Juvenile of *Salvator merianae* at Sancho Beach, Fernando de Noronha (photo: Vinicius Gasparotto).

We provide up to date information on the tegu population size and structure on Fernando de Noronha to contribute to an informed control programme to be undertaken by island conservation managers in the future.

METHODS

Study areas

To access the tegu population in the archipelago we selected two representative areas from the main island and visited the main vegetated islets that are used as nesting sites by resident birds. Land use in FN was simplified into three types, according to human usage: i) Densely inhabited areas, including hotels, houses and commercial buildings, paved streets and dense traffic, also with a higher density of uncontrolled dogs and cats; ii) Sparsely inhabited areas, including: rural areas similar to those found on the continent, and small villages with unpaved roads and sparse houses surrounded by crops and livestock animals. These two inhabited areas constitute most of the APA land; iii) Uninhabited areas, including areas of natural vegetation and secondary regeneration, with a few abandoned buildings and sporadic tourist usage. This area constitutes most of the PARNAMAR land (Fig. 2).

Within the inhabited areas, we chose the Boldró village that is a good representation of a sparsely inhabited area, with tourist visits, a small amount of commerce, paved and unpaved roads and houses of local workers. It is common to find domestic animals (dogs, cats, chickens), and crops and fruit trees in backyards. In the PARNAMAR we chose the southwestern Capim-açu region that represents the most intact? area of native vegetation on the main island (Mello & Adalardo de Oliveira, 2016). In Boldró village we performed a mark-recapture study and a line transect census study. In Capim-açu we performed a line transect census only.

Mark-recapture

To apply this method we chose the Boldró village located in a sparsely inhabited area of FN. This area is representative of the most common vegetation types on the main island and is subject to various levels of human interference while leaving space for native vegetation. Sampling seasons occurred during the years of 2015 and

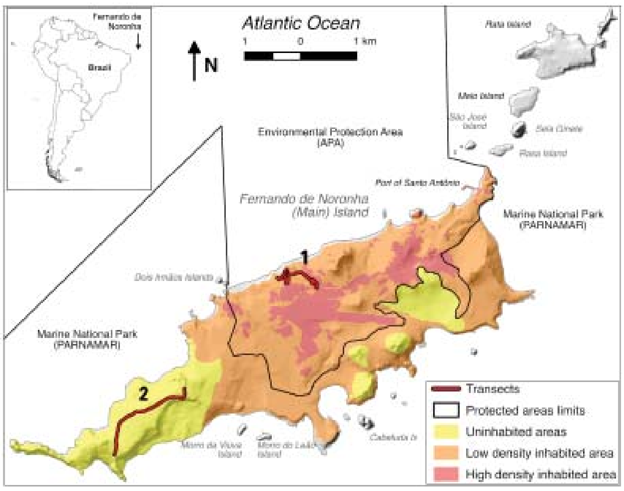

Fig. 2 Map of the protected areas of Fernando de Noronha Archipelago. Note: 1 is Boldró village transect, and 2 is Capim-açu transect (Land use layer by Vívian Uhlig – RAN/ICMBio).

2016, with 14 to 19 days of consecutive sampling in the beginning (Jan–Feb) and end (Oct–Nov) of the dry season. We opted to sample in the summer-spring as this species has been known to hibernate during the autumn-winter seasons on the continent (Andrade, et al., 2004; de Souza, et al., 2004).

We used ten funnel traps made out of PVC pipes (150 mm × 1 m) with one end closed. Those traps were placed in shaded spots next to vegetation borders, next to habitations, restaurants and areas that a tegu could use for hiding or foraging (Fig. 3). Each trap was placed in a stable position over trunks or stones in order to maintain at least a 20 degree angle to the closed end. The inclined position and lack of friction provided by the PVC material prevents animals from leaving the trap, where they remain until release. Raw chicken was used as baits and replaced every two days. Tegu locates the bait through smell (Yanosky, et al., 1993) and enters through the higher open entrance of the trap to get the bait that rests in the closed lower end of the pipe.

Traps were checked at the end of each day, when the individuals become inactive. Every animal was then restrained and marked with a transponder implanted subcutaneously. Snout vent length (SVL) was measured to the nearest 0.5 cm, with the use of a tape measure. The weight was taken using a Pesola® scale with a 10 g precision. Animals recaptured in the same season (e.g. less than 30 days interval) were considered to have the same weight and length, thus these data were collected only on the first capture of the season.

To estimate density (D) through mark-recapture data, we used the maximum-likelihood spatially explicit capture-recapture (ML SECR) package from R (Team, 2000; Borchers & Efford, 2008). We assumed a Poisson distribution of range centres (i.e. random) with a half-normal curve detection function parameterised by g_0 (probability of detection when trap and range centre coincide) and σ (spatial scale of the detection function). Removals from the population (i.e. poaching or death) are assigned known capture histories of 0 with probability equals 1 following death. The conditional likelihood was used to derive density, incorporating individual covariates of SVL and sex. Models were compared using an AIC framework, but due to sparse data, subsets of models on σ and then g_0 were considered independently. The area of capture exposure, which usually would be related to an individual's home-range, was approximated by a 95% circular probability density area of capture as:

$$HR_{95} = \pi(2.45\sigma)^2$$

Fig. 3 PVC Funnel trap to catch tegu mounted near a tree at the edge of a clearing.

Line-transect census

Two tracks were chosen to undertake the census counting (Fig. 2). One in the Boldró village, 1,820 m in length, to make possible comparisons between density methods in the same area, another in the Capim-açu track, 2,000 m in length, to make possible a comparison between a sparsely inhabited area and uninhabited area. A trained volunteer walked each track counting tegu in the high activity hours (10 a.m. to 2 p.m.). For six days, the Capim-açu track was walked in one direction and after a 30 min break at the farthest point, it was walked back. Atypical days with rain, temperatures below 25°C or excessive wind were avoided to prevent weather interference on abundance data. Counting along Boldró track was repeated nine times and Capim-açu 35 times during this study. When a tegu was sighted, the observer took the perpendicular distance of the animal from the centre of the track using a scale tape, to the nearest 0.5 m and up to 20 m distance. Any tegu sightings over 20 m of distance were discarded, but the thick vegetation in this region prevents seeing animals in the vegetated area on the transect borders.

We calculated the density of animals along the transect using distance sampling analysis, but zero spiking in the data (excessive observations close to the line) violated basic premises, likely due to a much higher level of detection, and potentially tegu abundance, along the clear open tracks in the dense forest. We subsequently used the line-transect census methodology (Burnham, et al., 1980; McDiarmid, et al., 2012) on a subset of the data, for observations directly on the open track only. Total number of individuals observed along the line-transect were used to represent the abundance on the track area, assuming every individual within the transect was observed. The area was calculated by using the average width of the track (measured every 100 m) and then multiplied by its length. Open areas were not measured and were assumed to have the same average width as the forested areas. Only animals observed within the established width of the track (e.g. clear area) were considered for such analysis.

We used a two-tailed t-test with unequal variances to compare daily density data between Boldró and Capim-açu. Only the high activity months (Feb–Jun, Sep–Nov) were used for this comparison, since we did not have data from the dry season in the Boldró area. The same method was used to compare densities observed on Capim-açu in the high activity months and low-activity months (Jul–Aug). To coarsely calculate the total abundance of tegu in FN, we stratified the map according to three main land uses: i): densely inhabited areas (226 ha); ii) sparsely inhabited areas (960 ha), and iii) uninhabited areas (417 ha) (Fig. 2). Average density and ranges from Capim-açu line-transect counts were used to estimate the abundance of tegu in the uninhabited areas of FN. The same method was used in Boldró to estimate the abundance of tegu in the sparsely inhabited areas of FN. Densely inhabited areas and areas with no vegetation (e.g. beaches, sand dunes and rocky areas – 97 ha) were excluded from the abundance calculations for they were not represented in the study area and were considered poor tegu habitat.

Islet surveys

We visited seven of the larger vegetated offshore islets of the archipelago (Rata, Rasa, do Meio, Conceição, Morro Dois Irmãos, Morro da Viuvinha and Morro do Chapéu) at least once during the study period (Fig. 2). We spent from one to twelve hours actively searching on each islet, searching for sightings or indirect signs of tegu presence (tracks or burrows). We also inquired with local inhabitants and other researchers for records of tegu presence on the other islands, since tegu can swim and also could have been brought to other islands intentionally in the past.

RESULTS

In the mark-recapture study we had a total of 190 captures over 69 trapping days in the Boldró village. From the 190 captures, we captured 103 unique individuals with 87 recaptures. Of the ten traps installed, two had to be moved in the last sampling season to avoid interference by people. These traps remained a total of 55 days in the first location and 14 days in the second location, less than 50 m away from where they were previously placed. Since tegu weight and size (SVL) were highly correlated (R^2=0.84), we have chosen only SVL as a covariate on σ and g_0. SVL also provides a better measure than total length, for it excludes the tail that can be lost or be regenerated to a variable size. Given the relatively low number of recaptures, we had to specify reasonable starting values for the likelihood maximisation with starting values of $g_0 = 0.1$ and sigma = 50 from a preliminary inspection of the data.

Ranging behaviour and probability of capture

We first fitted and ranked models combining the influence of sex and size on the ranging behaviour (σ) of the animals, while keeping a fixed capture probability (g_0). The simplest model, with fixed probability of capture and fixed ranging behaviour had 91% support, showing that size has no effect on the ranging behaviour of animals, while sex has little effect (Table 1).

Based on the best adjusted model for ranging behaviour, we kept σ constant across sessions and tested the influence of sampling period, sex and size on the probability of capture of the individuals. As seen in Table 2, the model including SVL had 44% support showing that body size as a continuous variable is the most important of the tested covariates to affect the probability of capture. Session also showed some importance in explaining the variation as seen in models 2 and 3.

Home-range

To produce real estimates for capture probabilities (g_0) and ranging behaviour (σ), we took the model including the most important covariates (session and size), for probability of capture and fixed ranging behaviour. The average size (SVL) used in the estimates was 30.2 cm (Table 3).

Based on real parameters obtained from the chosen model, we calculated 95% home ranges (HR_{95}) for average size and both sexes as 10.54 ha, ranging from 7.26 to 15.33 ha.

Density, abundance and activity

Finally, we estimated densities and sampled areas for each sampling season over the chosen model (Table 4).

In the line transect study, the Boldró transect (0.419 ha) was surveyed six times in the high-activity months (Nov 2015 and Feb 2016), with a total linear effort of 10.92 km. Only ten animals were sighted in this transect within the established transect width of 2.3 m during the period of study. The calculated density for Boldró is 3.98 (±1.1) animals/ha. The Capim-açu transect (0.492 ha) was surveyed 35 times from February 2015 to February 2016, with a total linear effort of 70 km. In this transect, 260 animals were sighted within the established average width of 2.46 m during the study. The calculated density for Capim-açu is 13.83 (±3.9) animals/ha.

Densities calculated using the line transect method were different between Boldró village and Capim-açu transects (t=6.45, P≤0.00001). There were no surveys in the Boldró transect during the low-activity months, thus, only densities from high-activity months in both transects were used to compare the densities averages from different areas. In Capim-açu, densities also differed between high-activity months and low-activity months (t=3.29, P≤0.01). The number of sightings on each occasion for Capim-açu transect is shown in Fig. 4 where a decline in number of sightings can be seen in the months of July and August.

To estimate the abundance of tegu in FN we used the calculated uninhabited area of FN as being 417 ha and the total sparsely inhabited area of FN as being 960 ha (see Fig. 2). Considering Capim-açu transect densities, calculated abundances range from 4,141 to 7,393 tegu in the uninhabited area. Using densities from Boldró transect for the sparsely inhabited areas, we estimated abundance from 2,765 to 4,877 tegu in that area. Total number of animals estimated for both calculated areas is from 6,906 to 12,270 tegu. High-density inhabited areas and non-vegetated areas of the island (463 ha) were excluded from this calculation for they were not represented in the samples; however, tegu are expected to be using those areas in a lower rate, thus abundance results should be taken as an underestimation of the whole population.

Population parameters

Males constituted the majority of the sampled population in all but the first sampling period. Males were also larger and heavier than females in all sampled periods. Male weight ranged from 400 g to 2,450 g and female weight ranged from 600 g to 1,940 g. Snout–vent (i.e. body) length for males ranged from 24 to 40 cm and for females from 26 to 36 cm. Averages and range by season and sex are given in Table 5.

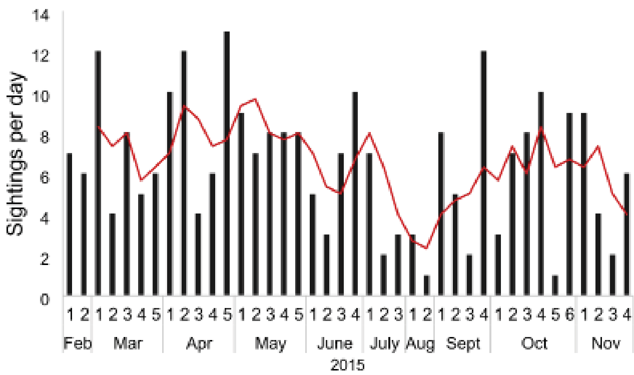

Fig. 4 Number of sightings of tegu in the Capim-açu transect during the 2015 sampling period. The line represents a moving average of three samples.

Table 1 Model results of tegu detection function for covariates of the scale parameter (σ) and the probability of capture equal to the home range centre (g_0).

σ Models	Detection function	Npar	Log likelihood	AICc	Rank	Weight %
sigma~1	Half normal	2	-609.159	1,222.411	1	91%
sigma~sex	Half normal	3	-610.382	1,226.950	2	9%
sigma~SVL	Half normal	3	-766.307	1,538.800	3	0%

Table 2 Best adjustment on models tested for constant probability of capture (g_0) with covariates as sampling size (SVL), sex, season (session) and ranging behaviour (sigma).

g_0 Models	Detection function	Npar	Log likelihood	AICc	Rank	Weight %
g_0~svl	Half normal	3	-605.896	1,217.978	1	44.26
g_0~session + svl	Half normal	6	-602.833	1,218.333	2	37.06
g_0~session	Half normal	5	-605.242	1,220.957	3	9.98
g_0~1	Half normal	2	-609.159	1,222.411	4	4.82
g_0~sex + svl	Half normal	4	-608.020	1,224.353	5	1.83
g_0~session + sex + svl	Half normal	7	-605.046	1,224.988	6	1.33
g_0~sex	Half normal	3	-610.016	1,226.219	7	0.72
g_0~session + sex	Half normal	6	-608.184	1,229.034	8	0

Table 3 Estimates of real parameters for σ and g_0 in each sampling season, using average size of 30.2 cm SVL. Given standard errors and 95% confidence intervals (lower class and upper class).

Real parameters SVL=30.2	Estimate	SE	lcl	ucl
g_0 Feb/2015	0.012	0.006	0.004	0.033
g_0 Nov/2015	0.035	0.010	0.019	0.062
g_0 Feb/2016	0.032	0.011	0.016	0.062
g_0 Nov/2016	0.039	0.010	0.023	0.064
σ	74.780	7.148	62.030	90.150

Islets

From the seven visited islets of the archipelago, only Rata Island had indirect records of the presence of tegu. There was an effort of 58.5 person-hours of active searching, plus 72 trap-hours divided among three visits to Rata Island, but no direct sights or captures were made. Tracks, faeces and burrows were found, indicating the presence of tegu, possibly at a lower density than the main island.

DISCUSSION

Policy makers, managers and the general public need to be informed of the consequences of invasive species in order to manage their impacts. Understanding the population biology of an invasive species is a first step to acquire essential information for management decisions that may alleviate impacts. Despite Fernando de Noronha being inhabited since the 16th century, very little has been done to understand or prevent the impact of invasive species on endangered and endemic species that struggle to coexist in the archipelago (Sampaio & Schmidt, 2014; Mello & Adalardo de Oliveira, 2016; Dias, et al., 2017).

Ranging behaviour and probability of capture

Spatial detection models show that size and sex had little influence on tegu ranging behaviour on FN. Klug, et al. (2015) found that size differences were not likely to be contributing to movement differences for tegu. In a subtropical coastal region on southern Brazil, Winck, et al. (2011) found tegu to be more active when temperatures start rising by the end of spring and early summer. They also related peaks of activity while males were dispersing and after the emergence period, to be due to the beginning of foraging and sexual activity. The present study does not capture full seasonal variation because of time-constrained sampling, but a drop in activity was observed in July and August, as observed in other tegu studies (Winck & Cechin, 2008; Tattersall, et al., 2016). This small window of low activity of tegu on FN may not promote a significant variation in relation to the impacts it causes to other species. On the main island, only masked booby (*Sula dactylatra*) still nest on the ground in a small peninsula next to the end of Capim-açu track. Their eggs are laid in the first months of the year as observed by e Silva & Neves (2008) on secondary islands of the archipelago. The Noronha skink (*Trachylepis atlantica*) is also a common prey item in the tegu diet. Despite being relatively abundant, nothing is known about its reproduction. It is thought to reproduce throughout the year as for *Trachylepis sechellensis* on the Seychelles, another tropical archipelago (Bringsøe, 2008). Sea turtle nests are also preyed upon by tegu, as recorded by TAMAR project for *Chelonia mydas* on FN (Bellini & Sanches, 1996; e Silva & Neves, 2008), including predation of hatchlings (Ayrton K. Péres-Jr, pers. comm.). Turtles on FN nest from January to June, when tegu are active.

For the probability of capture, size was an important determinant, but population studies using traps often fail to collect a broad representative sample of the population as seen in Carter, et al. (2012). A hole of 3 cm diameter was made in the closed end of the pipe to avoid flooding of the trap and unwanted capture of native lizards. This safety measure may bias the sample as it allows small animals to escape. These animals would possibly not be able to be marked by transponder implant and thus were of less importance for this study in any case. Behavioural traits such as niche separation due to intraspecific competition could also explain a size interference on capture probabilities (Herrel, et al., 2006; Siqueira & Rocha, 2008). The observed small influence of season on capture probabilities is primarily in the first session and possibly due to adjustments of methodology following that

Table 4 Densities and estimated sampling areas in Boldró village for each season sampled derived from the best adjustment models.

Period	Density/ha	Std. Error	Min (95%)	Max (95%)	ESA
Feb/2015	4.19	0.93	2.72	6.44	7.40
Nov/2015	4.45	0.94	2.95	6.71	7.42
Feb/2016	3.59	0.84	2.29	5.64	7.52
Nov/2016	5.07	1.05	3.39	7.59	8.28

Table 5 Tegu sex, size and weight averages with ranges in each sampling season.

Period	Sex	n (%)	$\bar{x}$ SVL (cm)	SVL range	$\bar{x}$ Weight (g)	Weight range
Feb 2015	M	15 (42)	33.58	29–37	1,491.00	875–2,175
	F	21 (58)	31.22	28–36	1,051.67	640–1,940
Nov 2015	M	21 (60)	32.26	28–39	1,347.14	740–2,240
	F	14 (40)	29.35	26–33	965.00	600–1,550
Feb 2016	M	18 (55)	33.28	28–37	1,368.89	660–1,930
	F	15 (45)	30.40	27–34	914.67	600–1,590
Nov 2016	M	47 (68)	32.53	24–40	1,294.26	400–2,450
	F	22 (32)	30.67	26–35	1,030.45	600–1,560

first sampling season. A variation in capture probabilities is not expected once sampling seasons were chosen within the high-activity periods for tegu.

Home-range

In the tegu natural distribution, older males have larger territories, while juvenile males and females have smaller territories with higher overlap. A peak of activity in males was observed at the end of the low-activity period (Winck, et al., 2011). A decrease in home range after the mating season was also observed in the El Palmar National Park, in Argentina (Fitzgerald, et al., 1991). Results from this study suggest little influence of sex on tegu capture in traps. Since SECR only estimates spatial exposure area to traps, sex could be affecting the ranging behaviour of tegu in FN but this method is not precise enough to detect such variation. This result may also have been affected by biases in the capture probability of certain tegu size classes (i.e. juveniles).

Lirio, et al. (2004) tracked six radio-implanted tegu in FN and estimated home-ranges varied from 0.73 to 7.8 ha (3.3 ha on average). The authors also comment that a gravid female was used in the study, representing the smallest home-range, and that the activity centre was usually close to the shelter. Winck, et al. (2011), found home-ranges from 0.05 to 26.4 ha for a continental population in southern Brazil. Home-ranges as measured in the present study are within the previous findings for the natural distribution of tegu and are a little higher than those described by Lirio, et al. (2004).

Since σ did not differ between sexes, estimated home-ranges were considered the same for males and females. Home-ranges can provide necessary information to set management on invasive species, such as the density of control devices (Hays & Conant, 2007; Howald, et al., 2007; Anderson, et al., 2016). For continental tegu, behavioural traits such as season, age and reproductive status can be implicated in home-range variation (Winck, et al., 2011). In FN, factors such as the lack of competitors, predators and resource availability could be also influencing tegu home ranges (Ballinger, 1977; Shine, 1987; Novosolov, et al., 2016). With an average home range (HR_{95}) of 10.54 ha, tegu on FN are quite mobile. This behaviour allows them to look for resources in a vast area and feed even when resources are not abundant (e.g. dry season). We also noticed an overlap of territories throughout the year, as juveniles forage together and coexist with adult males and females in the same area. Only youngsters seem to avoid larger tegu, having more secretive habits. In general a large home range also increases the probability of a species being exposed to a control method (Howald, et al., 2007). That means managers might need fewer traps (e.g. one every few ha) in order to control tegu on FN.

Density, abundance and activity

In Boldró village, density estimates from capture-recapture ranged from 2.29 to 7.59 animals/ha while estimates from the line transect census ranged from 2.88 to 5.08 animals/ha. Those densities are much higher than the 0.83 animals/ha observed for a tegu population living in Anchieta Island or the 0.63 animals/ha as seen in the Espírito Santo Atlantic rainforest, both in south-eastern Brazil (Bovendorp, et al., 2008; Chiarello, et al., 2010). These higher estimates could be due to a tropical climate in FN that favours reptiles with a low variation in temperature over the year. Abundance of resources and the lack of natural predators can also contribute to the higher density observed in FN as seen for other island invasive predators (Pekelharing, et al., 1998; Hays & Conant, 2007; Ferreira, et al., 2012).

Since density estimates from both methods used in this study (line transect census and mark-recapture) were similar, we opted to use values from the line transect census because it also provided density for the Capim-açu transect. Density from those transects was applied to the region represented by each transect to obtain abundance for both represented areas. There is a possible error associated with the extrapolation of the transect densities over the whole area, especially to areas with dense vegetation, as observers may find a higher number of tegu using the open areas, causing an overestimation of density. However, a similar density estimated by two different methods supports the idea of transect counts being a reliable method, despite the associated error. An estimate of abundance can help management decisions in quantifying the effort and costs required to control or eradicate (Holmes, et al., 2015; Keitt, et al., 2015). Density estimated in Capim-açu was higher than that estimated in Boldró and a broad list of factors could explain such differences, the most important are discussed here.

Animals are not distributed uniformly in the environment and they tend to occupy environments that seem more favourable, while less favourable habitats are occupied in lower densities (Diaz & Carrascal, 1991; Fraga, et al., 2013). In FN, presence of predators and competitors, such as cats, could negatively affect tegu populations by preying on juveniles and hunting other potential prey of tegu such as rats and other reptiles. In Boldró village and other high-density inhabited areas of the island, the influence of cats is higher, since the cat population is denser when closer to inhabited areas (Dias, et al., 2017). Dogs also inhibit presence of tegu by chasing and killing tegu when they cross territories, making inhabited areas again less suitable for tegu (C.A. pers. obs.).

Tegu are appreciated for their meat in the northeast of Brazil, where the species can be a delicacy and an important source of protein in poor communities (Mendonça, et al., 2011; Nóbrega Alves, et al., 2012). Poaching of tegu in FN is driven by different reasons, with tegu being commonly hunted by poultry farmers when they break into henhouses to eat eggs and chicks. Hunting in FN is done with fishing line and hooks, baited with fish or chicken, in the areas close to residences (C.A. pers. obs.). Tegu abdominal fat is also widely known as a medicine and is used by locals to treat sore throat, earache and other ailments (Nóbrega Alves, et al., 2012). Those properties are scientifically based since the anti-inflammatory properties of tegu fat has been proven (Ferreira, et al., 2010).

Tegu are generalists and feed on any available resources, including vegetation, fruits, insects, vertebrates and eggs (Vanzolini, et al., 1980; Kiefer & Sazima, 2002; Mourthé, 2010). Those adaptations do not restrict resources for the tegu population in FN, where it possibly lives with plenty of food throughout the year. A reduction in the tegu population is more likely to be present in human altered environments such as densely inhabited areas, with negative effects of domestic animals and poaching, despite a possible higher availability of food (crops, fruit trees and rubbish). Another factor that could be affecting the results is of behavioural origin. The negative impact of human presence seems to make the tegu population shift towards uninhabited areas that offer better habitat with less interference and still plenty of resources. Despite density underestimation being a possibility when failing to observe all animals on the transect (e.g. behaviour to avoid human contact in inhabited areas), the more intensive mark-recapture study showed similar estimates of density thereby disproving a possible methodological interference.

Population parameters

Size in this study was inferred by SVL and was also closely correlated to weight. Although, size can be affected by external factors when trying to infer individuals' ages (Halliday & Verrell, 1988; Adolph & Porter, 1996), weight can also reflect body condition and be influenced by the loss of the tail, a common finding in the FN population. Size can be related to sexual maturity (Fitzgerald, et al., 1993), while movement and home-ranges can be affected by sex and reproductive status (Winck, et al., 2011). Size is also related to reproductive capacity of females (Fitzgerald, et al., 1993). Tegu on FN seem to be smaller than those found in continental South America, thus, the female reproduction index in FN should be lower than in the continent (Fitzgerald et al., 1991; Winck et al., 2011). The smaller size on FN can also be related to a much higher density caused by lower competition and predation rates than the ones found in the continent (Novosolov, et al., 2016).

Males seem to be a higher fraction of the population on FN, which might influence reproduction and population growth (Le Galliard, et al., 2005). Sex ratio can be affected by average temperature (e.g. natality rates) or by any factor that increases mortality rates in only one of the sexes. Populations of tegu in Paraguay were consistently male-biased (Mieres & Fitzgerald, 2006), possibly leading to a higher fecundity rate of females or having a negative effect on lizard populations as observed by Le Galliard, et al. (2005).

Islets

Tegu are good swimmers and there are various sightings from local residents of tegu swimming or diving near to the main island. A video made by Elias Pereira and Nelly Burella shows a juvenile tegu voluntarily swimming across Baia dos Golfinhos, on the main island. Other than swimming, tegu could have been taken to other islands in the past for the same reason they were taken to the main island (either to control rats or serve as a food supply). Manoel P. dos Santos, who lived on Rata Island until 1986, says tegu were abundant there during that time. It seems that after his family left Rata, the population of tegu has decreased. However, the island seems to be big enough to maintain a small population of tegu. Some animals might also occasionally swim to other islets, but even a single animal could hardly live for long on the scarce resources available on those smaller islands, forcing them back to the main island.

Future steps

The reasons why the tegu was introduced to Fernando de Noronha, when it happened and the impacts this predator has caused to the archipelago were not documented and remain unknown. However, the understanding of impacts caused by invasive predators in islands worldwide provides sufficient evidence that management is required in order to protect local biodiversity. Eradication is usually the best option when the tools are available, but when they are absent, control measures may be better than the do-nothing approach (Fletcher, et al., 2015; Russell, et al., 2017).

On Fernando de Noronha, managing the impacts of tegu over native fauna is already on the list of priorities, as documented in the management plan of the APA (Brasil, 2004). However, providing up to date information on tegu population structure and biology in FN is expected to contribute to the implementation of a science-based invasive species programme in the future. Based on results from this work and field experience of the authors in FN, our contribution to this programme is offered here as a suggestion to local managers and decision-makers.

Measures of tegu control in FN should be placed in strategic locations where impacts on native fauna are considered higher, such as ground nesting sites for birds, nesting beaches for turtles and most preserved vegetated areas for other reptiles, crabs and even invertebrate fauna. Live or kill traps could be used, depending on the destination identified for the animals. Traps like the ones used in this study proved to be very efficient for adult tegu and seem very cost-effective. Considering the relatively high probability of capture observed, live traps needs to be checked at least once a day. Traps also need to be placed in the shade as lizards are easily prone to overheating in the tropics. Traps can be baited with eggs, bacon, chicken, fish or any other scent-driven attractant, since smell is the main sense for area exploration of tegu.

Considering the very high density of animals, an equally high number of traps should be required (one per ha or more). Control areas can be fenced by a tegu-proof fence to prevent quick reestablishment of the population by recruitment of juveniles. Traps should be placed preferentially in transition areas between vegetated and clear areas, where tegu transit to control body temperature during times of higher activity. Management effort should be stronger after the low-activity period, up to the end of the reproduction season (expected to be from September to March in FN). However, since there are animals active throughout the year, effort should also be made according to the reproduction of potential prey species such as the ground-nesting birds, sea turtle nests and crab spawning period. Control effort is expected to be up to four times higher in the uninhabited areas than in the inhabited areas of FN, given tegu density variation between those areas.

There are no specific tools available to control tegu and poison should not currently be considered as an option, since it would also threaten other endemic reptiles in FN. Hunting also requires special firearm permits and doesn't seem to be an option when in a tourist location like FN. For the moment, only fencing and trapping seem to be feasible solutions to manage tegu impacts on the archipelago's biodiversity.

CONCLUSION

Some invasive species are not commonly widespread and attract little attention of researchers. However, once established, those species can pose a real threat to native biodiversity (Simberloff, 2009; Neves, et al., 2017). Tegu have been established on FN for a century (Santos, 1950), but their population structure and impacts on native fauna remained understudied. This assessment provides focal information for a future control programme of tegu on Fernando de Noronha archipelago. We also aim to contribute to a larger ongoing process in Brazil, where invasive species move towards being a primary problem to be addressed for biodiversity conservation. Finally, we call on researchers worldwide to focus on other neglected invasive insular species as they represent a challenge and a frontier for island conservation.

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REFERENCES

Adolph, S.C. and Porter, W.P. (1996). 'Growth, seasonality and lizard life histories: Age and size at maturity'. *Oikos* 77: 267–278.

Alves, R.R.N., Vieira, K.S., Santana, G.G., Vieira, W.L.S., Almeida, W.O., Souto, W.M.S., Montenegro, P.F.G.P. and Pezzuti, J.C.B. (2012). 'A review on human attitudes towards reptiles in Brazil'. *Environmental Monitoring and Assessment* 184: 6877–6901.

Anderson, D.P., McMurtrie, P., Edge, K.A., Baxter, P.W.J. and Byrom, A.E. (2016). 'Inferential and forward projection modeling to evaluate options for controlling invasive mammals on islands'. *Ecological Applications* 26: 2546–2557.

Andrade, D.V, Sanders, C., Milsom, W.K. and Abe, A.S. (2004). 'Overwintering in Tegu Lizards'. In: B.M. Barnes and H.V. Carey (eds.) *Life in the Cold: Evolution, Mechanisms, Adaptation and Application. Twelfth International Hibernation Symposium*, pp. 339–348. Fairbanks: University of Alaska.

Andrade, L., Gomes, V. and Dias, M. (2009). 'Desafios para o futuro sustentável da Ilha de Fernando de Noronha: A visão ecossistêmica da ocupação urbana'. *Oculum Ensaios* 9: 86–105.

Ballinger, R.E. (1977). 'Reproductive strategies: Food availability as a source of proximal variation in a lizard'. *Ecology* 58: 628–635.

Bellini, C. and Sanches, T.M. (1996). 'Reproduction and feeding of marine turtles in the Fernando de Noronha Archipelago, Brazil'. *Marine Turtle Newsletter* 74: 12–13.

Borchers, D.L. and Efford, M.G. (2008). 'Spatially explicit maximum likelihood methods for capture-recapture studies'. *Biometrics* 64: 377–385.

Bovendorp, R.S., Alvarez, A.D. and Galetti, M. (2008). 'Density of the tegu lizard (*Tupinambis merianae*) and its role as nest predator at Anchieta Island, Brazil'. *Neotropical Biology and Conservation* 3: 9–12.

Branner, J.C. (1888). 'Notes on the fauna of the islands of Fernando de Noronha'. *The American Naturalist* 22: 861–871.

Brasil. (2004). 'Plano de manejo da APA de Fernando de Noronha - Rocas - São Pedro e São Paulo'. *Mma/Ibama/Arcadis/Tetraplan S.a* 1206.

Bringsøe, H. (2008). '*Trachylepis sechellensis* (Seychelles skink): Reproduction'. *Herpetological Bulletin* 103: 42–43.

Brito, S.P., Abe, A.S. and Andrade, D.V. (2001). '*Tupinambis merianae* (tegu lizard) longevity'. *Herpetological Review* 32: 260–261.

Burnham, K.P., Anderson, D.R. and Laake, J.L. (1980). 'Estimation of density from line transect sampling of biological populations'. *Wildlife Monographs* 72: 938–942.

Carter, A. J., Heinsohn, R., Goldizen, A. W., and Biro, P. A. (2012). 'Boldness, trappability and sampling bias in wild lizards'. *Animal Behaviour* 83(4): 1051–1058.

Chapin, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., Mack, M.C. and Díaz, S. (2000). 'Consequences of changing biodiversity'. *Nature* 405: 234–242.

Chiarello, A.G., Srbek-Araujo, A.C., Del-Duque Jr., H.J., Coelho, E.D.R. and Rocha, C.F.D. (2010). 'Abundance of tegu lizards (*Tupinambis merianae*) in a remnant of the Brazilian Atlantic forest'. *Amphibia Reptilia* 31: 563–570.

da Silva, E.A.P., Santos, T.D., Leite, G.N. and Ribeiro, L.B. (2013). '*Tropidurus hispidus* (Squamata: Tropiduridae) and *Leptodactylus* cf. *fuscus* (Anura: Leptodactylidae) as prey of the teiid lizards *Salvator merianae* and *Ameiva ameiva*'. *Herpetology Notes* 6: 51–53.

de Souza, S.C.R., de Carvalho, J.E., Abe, A.S., Bicudo, J.E.P.W. and Bianconcini, M.S.C. (2004). 'Seasonal metabolic depression, substrate utilisation and changes in scaling patterns during the first year cycle of tegu lizards (*Tupinambis merianae*)'. *Journal of Experimental Biology* 207: 307–318.

Dias, R.A., Abrahão, C.R., Micheletti, T., Mangini, P.R., de Oliveira Gasparotto, V.P., de Jesus Pena, H.F., Ferreira, F., Russell, J.C. and Silva, J.C.R. (2017). 'Prospects for domestic and feral cat management on an inhabited tropical island'. *Biological Invasions* 19: 2339–2553.

Diaz, J.A. and Carrascal, L.M. (1991). 'Regional distribution of a Mediterranean lizard – influence of habitat cues and prey abundance'. *Journal of Biogeography* 18: 291–297.

Doherty, T.S., Glen, A.S., Nimmo, D.G., Ritchie, E.G. and Dickman, C.R. (2016). 'Invasive predators and global biodiversity loss'. *Proceedings of the National Academy of Sciences* 113: 11261–11265.

Donadio, O.E. and Gallardo, J.M. (1984). 'Biología y conservación de las especies del género Tupinambis (Squamata, Sauria, Teiidae) en la República Argentina'. *Revista del Museo Argentino de Ciencias Naturales "Bernardino Rivadavia" Zoologia* 13: 117–127.

Dorcas, M.E., Willson, J.D., Reed, R.N., Snow, R.W., Rochford, M.R., Miller, M.A., Meshaka, W.E., Andreadis, P.T., Mazzotti, F.J., Romagosa, C.M. and Hart, K.M. (2012). 'Severe mammal declines coincide with proliferation of invasive Burmese pythons in Everglades National Park'. *Proceedings of the National Academy of Sciences* 109: 2418–2422.

e Silva, R. S. and Neves, F. O. C. (2008). *Aves de Fernando de Noronha: Birds of Fernando de Noronha*. Vinhedo: Avis Brasilis.

Eales, J., Thorpe, R.S. and Malhotra, A. (2010). 'Colonization history and genetic diversity: adaptive potential in early stage invasions'. *Molecular Ecology* 19: 2858–2869.

Ferreira, F.S., Brito, S. V., Saraiva, R.A., Araruna, M.K.A., Menezes, I.R.A., Costa, J.G.M., Coutinho, H.D.M., Almeida, W.O. and Alves, R.R.N. (2010). 'Topical anti-inflammatory activity of body fat from the lizard *Tupinambis merianae*'. *Journal of Ethnopharmacology* 130: 514–520.

Ferreira, R.B., Beard, K.H., Peterson, S.L., Poessel, S.A. and Callahan, C.M. (2012). 'Establishment of introduced reptiles increases with the presence and richness of native congeners'. *Amphibia-Reptilia* 33: 387–392.

Fitzgerald, L.A., Chani, J.M. and Donadio, O.E. (1991). '*Tupinambis* Lizards in Argentina: Implementing Management of a Traditionally Exploited Resource'. In: J. Robinson and K. Redford (eds.) *Neotropical Wildlife: Use and Conservation*, pp. 303–316. Chicago: University of Chicago.

Fitzgerald, L.A., Cruz, F.B. and Perotti, G. (1993). 'Reproductive cycle and the size at maturity of *Tupinambis rufescens* (Sauria : Teiidae) in the dry chaco of Argentina'. *Journal of Herpetology* 27: 70–78.

Fletcher, C. S., Westcott, D. A., Murphy, H. T., Grice, A. C., and Clarkson, J. R. (2015). 'Managing breaches of containment and eradication of invasive plant populations'. *Journal of Applied Ecology* 52(1): 59–68.

Fraga, R. De, Magnusson, W.E., Abrahão, C.R., Sanaiotti, T. and Lima, A.P. (2013). 'Habitat Selection by *Bothrops atrox* (Serpentes: Viperidae) in Central Amazonia, Brazil'. *Copeia* 2013: 684–690.

Gasparini, J.L., Peloso, P.L. and Sazima, I. (2007). 'New opportunities and hazards brought by humans to the island habitat of the skink *Euprepis atlanticus*'. *Herpetological Bulletin* 100: 30–33.

Halliday, T.R. and Verrell, P.A. (1988). 'Body size and age in amphibians and reptiles'. *Journal of Herpetology* 22: 253–265.

Hays, W.S.T. and Conant, S. (2007). 'Biology and impacts of Pacific island invasive species. 1. A worldwide review of effects of the small Indian mongoose, *Herpestes javanicus* (Carnivora: Herpestidae)'. *Pacific Science* 61: 3–16.

Herrel, A., Joachim, R., Vanhooydonck, B. and Irschick, D.J. (2006). 'Ecological consequences of ontogenetic changes in head shape and bite performance in the Jamaican lizard *Anolis lineatopus*'. *Biological Journal of the Linnean Society* 89: 443–454.

Holmes, N.D., Campbell, K.J., Keitt, B.S., Griffiths, R., Beek, J., Donlan, C.J. and Broome, K.G. (2015). 'Reporting costs for invasive vertebrate eradications'. *Biological Invasions* 17: 2913–2925.

Howald, G., Donlan, C.J., Galván, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

IBGE (2017). Demografia de Fernando de Noronha [WWW Document] *Instituto Brasileiro de Geografia e Estatística*. <http://cod.ibge.gov.br/68D>.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

Kiefer, M.C. and Sazima, I. (2002). 'Diet of juvenile tegu lizard *Tupinambis merianae* (Teiidae) in southeastern Brazil'. *Amphibia-Reptilia* 23: 105–108.
Klug, P.E., Reed, R.N., Mazzotti, F.J., McEachern, M.A., Vinci, J.J., Craven, K.K. and Yackel Adams, A.A. (2015). 'The influence of disturbed habitat on the spatial ecology of Argentine black and white tegu (*Tupinambis merianae*), a recent invader in the Everglades ecosystem (Florida, USA)'. *Biological Invasions* 17: 1785–1797.
Lanfri, S., Di Cola, V., Naretto, S., Chiaraviglio, M. and Cardozo, G. (2013). 'Understanding the ecological niche to elucidate spatial strategies of the southernmost *Tupinambis* lizards'. *Amphibia-Reptilia* 34: 551–565.
Le Galliard, J.F., Fitze, P.S., Ferrière, R. and Clobert, J. (2005). 'Sex ratio bias, male aggression, and population collapse in lizards'. *Proceedings of the National Academy of Sciences* 102: 18231–18236.
Lirio, M.R., Araujo, A.F.B., Surita, L. and Péres-Jr., A.K. (2004). 'Monitoramento do Lagarto *Tupinambis merianae* em Fernando de Noronha Através de Rádiotelemetria'. In: *Congresso Brasileiro de Zoologia – Reptilia –anais*. Brasília, p. 413.
Lopes, H.R. and Abe, A.S. (1999). 'Biologia Reprodutiva e Comportamento do Teiú, *Tupinambis merianae* (Reptilia, Teiidae), em Cativero'. In: *Manejo Y Conservación de Fauna Silvestre En América Latina*, pp. 259–272. La Paz, Bolivia: Editorial Instituto de Ecología.
Manes, M.E., Parry, H.V. and Manlla, A., (2007). 'Growth and feeding conversion responses to different protein and energy supplies in *Tupinambis merianae* (Squamata: Teiidae)'. *Applied Herpetology* 4: 279–285.
Marinho, A.C. (2016). Qual a população de Fernando de Noronha? *Portal G1*. <http://g1.globo.com/pernambuco/blog/viver-noronha/post/qual-populacao-de-fernando-de-noronha.html>.
Mazzotti, F.J., McEachern, M., Rochford, M., Reed, R.N., Eckles, J.K., Vinci, J., Edwards, J. and Wasilewski, J. (2015). '*Tupinambis merianae* as nest predators of crocodilians and turtles in Florida, USA'. *Biological Invasions* 17: 47–50.
McCreless, E.E., Huff, D.D., Croll, D.A., Tershy, B.R., Holmes, N.D., Spatz, D.R., Butchart, S.H.M. and Wilcox, C. (2016). 'Past and predicted future impact of invasive alien mammals on island threatened vertebrate populations'. *Nature Communications* 7: 1–11.
McDiarmid, R.W., Foster, M.S., Guyer, C., Whithfield Gibbons, J., Chernoff, N. and Dodd Jr, C.K. (2012). *Reptile Biodiversity: Standard Methods for Inventory and Monitoring*. Los Angeles: Copeia.
Mello, T.J. and Adalardo de Oliveira, A. (2016). 'Making a bad situation worse: An invasive species altering the balance of interactions between local species'. *PLOS ONE* 11: 1–17.
Mendonça, L.E.T., Souto, C.M., Andrelino, L.L., Souto, W.M.S., Vieira, W.L.S. and Alves, R.R.N. (2011). 'Conflitos entre pessoas e animais silvestres no semiárido paraibano e suas implicações para conservação'. *Sitientibus série Ciências Biológicas* 11: 185–199.
Mieres, M.M. and Fitzgerald, L.A. (2006). 'Monitoring and managing the harvest of tegu lizards in Paraguay'. *Journal of Wildlife Management* 70: 1723–1734.
Mourthé, Í.M.C. (2010). '*Tupinambis merianae* (common tegu) feeding behavior'. *Herpetological Review* 41: 232–233.
Muscat, E., Olmos, F. and Rotenberg, E. (2016). '*Salvator merianae* (black and white tegu) scavenging around the nests of *Fregata magnificens* (frigatebird)'. *Herpetological Bulletin* 135: 36–37.
Neves, V.C., Nava, C., Monteiro, E. V, Monteiro, P.R. and Bried, J. (2017). 'Depredation of Monteiro's storm-petrel (*Hydrobates monteiroi*) chicks by Madeiran wall lizards (*Lacerta dugesii*)'. *Waterbirds* 40: 82–86.
Nóbrega Alves, R., Pereira Filho, G., Silva Vieira, K., Silva Souto, W., Mendonça, L.E., Montenegro, P., Almeida, W. de and Silva Vieira, W. (2012). 'A zoological catalogue of hunted reptiles in the semiarid region of Brazil'. *Journal of Ethnobiology and Ethnomedicine* 8: 27.
Novosolov, M., Rodda, G.H., Feldman, A., Kadison, A.E., Dor, R. and Meiri, S. (2016). 'Power in numbers. Drivers of high population density in insular lizards'. *Global Ecology and Biogeography* 25: 87–95.
Oren, D.C. (1984). 'Resultados de uma nova expedição zoológica a Fernando de Noronha'. *Boletim do Museu Paranaense Emilio Goeldi Zoologia* 1: 19–44.
Passos, D.C., Lima-Araujo, F., Melo, A.C.B. and Borges-Nojosa, D.M. (2013). 'New state record and distribution extension of the golden tegu *Tupinambis teguixin* (Linnaeus, 1758) (Squamata: Teiidae) to the Caatinga biome, northeastern Brazil'. *Check List* 9: 1524–1526.
Pekelharing, C.J., Parkes, J.P. and Barker, R.J. (1998). 'Possum (*Trichosurus vulpecula*) densities and the impacts on fuchsia (*Fuchsia excorticata*) in South Westland, New Zealand'. *New Zealand Journal of Ecology* 22: 197–203.
Pernambuco. (2017). *Arquipélago de Fernando de Noronha, Informações Turísticas*. *Site oficial do governo de Pernambuco*. <http://www.prontotecnologia.com.br/noronha2/turInfo.php>.
Pernas, T., Giardina, D.J., McKinley, A., Parns, A. and Mazzotti, F.J. (2012). 'First observations of nesting by the Argentine black and white tegu, *Tupinambis merianae*, in South Florida'. *Southeastern Naturalist* 11: 765–770.
Phillips, B.L., Brown, G.P., Grennlees, M., Webb, J.P. and Shine, R. (2007). 'Rapid expansion of the cane toad (*Bufo marinus*) invasion front in tropical Australia'. *Austral Ecology* 32: 169–176.
Powell, R., Henderson, R.W., H., Farmer, M.C., Breuil, M., Echternacht, A.C., van Buurt, G., Romagosa, C.M. and Perry G. (2011). 'Introduced Amphibians and Reptiles in the Greater Caribbean: Patterns and Conservation Implications'. In: A. Hailey, B.S. Wilson and J. Horrocks (eds.) *Conservation of Caribbean Island Herpetofaunas Volume 1: Conservation Biology and the Wider Caribbean*, pp. 63–143. Leiden and Boston: Brill.
Presch, W. (1973). 'A Review of the tegus, lizard genus *Tupinambis* (Sauria : Teiidae) from South America'. *Copeia* 1973: 740–746.
Ramalho, A.C.O., da Silva, R.J., Schwartz, H.O. and Péres-Jr, A.K. (2009). 'Helminths from an introduced species (*Tupinambis merianae*), and two endemic species (*Trachylepis atlantica* and *Amphisbaena ridleyi*) from Fernando de Noronha Archipelago, Brazil.'. *The Journal of Parasitology* 95: 1026–1028.
Reaser, J.K., Meyerson, L.A., Cronk, Q., De Poorter, M., Eldrege, L.G., Green, E., Kairo, M., Latasi, P., Mack, R.N., Mauremootoo, J., O'Dowd, D., Orapa, W., Sastroutomo, S., Saunders, A., Shine, C., Thrainsson, S. and Vaiutu, L. (2007). 'Ecological and socioeconomic impacts of invasive alien species in island ecosystems'. *Environmental Conservation* 34: 98–111.
Reis, A. and Hayward, P. (2013). 'Pronounced particularity: A comparison of governance structures on Lord Howe Island and Fernando de Noronha'. *Island Studies Journal* 8: 285–298.
Ridley, H.N. (1890). 'Notes on the zoology of Fernando de Noronha'. *Linnean Journal of Zoology* 20: 473–570.
Russell, J.C., Meyer, J.-Y., Holmes, N. and Pagad, S. (2017). 'Invasive alien species on islands: Impacts, distribution, interactions and management'. *Environmental Conservation* 44: 1–13.
Sampaio, A.B. and Schmidt, I.B. (2014). 'Espécies exóticas invasoras em unidades de conservação federais do Brasil'. *Biodiversidade Brasileira* 2: 32–49.
Santos, A.T. dos (1950). *Fernando de Noronha, Ilha de Cenários Múltiplos*, 1st ed. Rio de Janeiro, RIO.
Sazima, I. and Haddad, C.F.B. (1992). 'Répteis da Serra do Japi: Notas sobre História Natural'. In: L.P.C. Morellato (ed.) *História Natural da Serra do Japi: Ecologia E Preservação de Uma Área Florestal do Sudeste do Brasil*, pp. 212–236. Campinas: Unicamp e FAPESP.
Sazima, I. and Haemig, P.D.P. (2012). 'Aves, mamíferos e répteis de Fernando de Noronha'. *Ecologia.Info 17*. < http://ecologia.info/fernando-de-noronha.htm>.
Shine, R. (1987). 'Ecological comparisons of island and mainland populations of Australian tigersnakes (*Notechis*: Elapidae)'. *Herpetologica* 43: 233–240.
Silva-Jr, J. M., Pérez-Jr, A. K., and Sazima, I. (2005). '*Euprepis atlanticus* (noronha skink). Predation'. *Herpetological review* 36: 62–63.
Simberloff, D. (2009). 'Rats are not the only introduced rodents producing ecosystem impacts on islands'. *Biological Invasions* 11: 1735–1742.
Simberloff, D.S. (1974). 'Equilibrium theory of island biogeography and ecology'. *Annual Review of Ecology and Systematics* 5: 161–182.
Siqueira, C.D.C. and Rocha, C.F.D. (2008). 'Predation by lizards as a mortality source for juvenile lizards in Brazil'. *South American Journal of Herpetology* 3: 82–87.
Tattersall, G.J., Leite, C.A.C., Sanders, C.E., Cadena, V., Andrade, D.V, Abe, A.S. and Milsom, W.K. (2016). 'Seasonal reproductive endothermy in tegu lizards'. *Science Advances* 2: 1–8.
Team, R.C. (2000). 'R language definition'. *Vienna, Austria: R foundation for statistical computing.*
Tershy, B.R., Shen, K.W., Newton, K.M., Holmes, N.D. and Croll, D.A. (2015). 'The importance of islands for the protection of biological and linguistic diversity'. *BioScience* 65: 592–597.
UNEP-WCMC. (2014). Checklist of CITES species [WWW Document]. *CITES Secretariat, United Nations Environmental Programme*. <http://checklist.cites.org>.
Vanzolini, P., Ramos-costa, A.M.M. and Vitt, L.J. (1980). '*Répteis das caatingas*', 1st ed., FNDCT-FINEP. Rio de Janeiro, Academia Brasileira de Ciências.
Veitch, C.R. and Clout, M.N. (eds.) 2002. *Turning the tide: the eradication of invasive species*. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.
Vitousek, P.M. (1988). 'Diversity and Biological Invasions of Oceanic Islands'. In: E.O. Wilson, F.M. Peter (eds.) *Biodiversity*, pp. 181–189. Washington, DC: National Academy Press.
Vitousek, P.M., D'Antonio, C.M., Loope, L.L., Rejmánek, M. and Westbrooks, R. (1997). 'Introduced species: A significant component of human-caused global change'. *New Zealand Journal of Ecology* 21: 1–16.
Winck, G.R. and Cechin, S.Z. (2008). 'Hibernation and emergence pattern of *Tupinambis merianae* (Squamata: Teiidae) in the Taim Ecological Station, southern Brazil'. *Journal of Natural History* 42: 239–247.
Winck, G.R., Blanco, C.C. and Cechin, S.Z. (2011). 'Population ecology of *Tupinambis merianae* (Squamata, Teiidae): home-range, activity and space use'. *Animal Biology* 61: 493–510.
Wyatt, K.B., Campos, P.F., Gilbert, M.T.P., Kolokotronis, S.O., Hynes, W.H., DeSalle, R., Daszak, P., MacPhee, R.D.E. and Greenwood, A.D. (2008). 'Historical mammal extinction on Christmas Island (Indian Ocean) correlates with introduced infectious disease'. *PLOS ONE* 3: e3602.
Yanosky, A.A., Iriart, D.E. and Mercolli, C. (1993). 'Predatory behavior in *Tupinambis teguixin* (Sauria: Teiidae). Island Tongue-flicking responses to chemical food stimuli'. *Journal of Chemical Ecology* 19: 291–299.

Life-history comparisons between the native range and an invasive island population of a colubrid snake

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Abstract Invasive snakes can lead to the rapid extinction of endemic vertebrates on insular ecosystems, usually because snakes are an efficient and novel predator. There have been no successful (i.e. complete) eradications to date of invasive snakes on islands. In this study we assess a novel invasion on Gran Canaria in the Canary Islands. The invader, the California king snake (*Lampropeltis californiae*), arrived from California via several generations in the pet trade. King snakes are captive bred for various phenotypes, and first were detected in the wild on Gran Canaria in the 1990s. Because very little natural history data exist from within their native range, we focused on developing datasets from native habitats to compare with similar data for the introduced snakes in the Canary Islands. We found that most aspects of the snake's life history have not changed since invasion, except that there appears to be a lower level of juvenile recruitment along with an increase in the length and body mass of adult snakes on Gran Canaria. We identified environmental parameters for when capture/trapping could be completed to reduce effort and maximize success. Additionally, we show different trap success on the various life stages of the snakes. Risk assessments could be required prior to permitting pet trade or allowing captive bred snakes into regions where they are not native.

Keywords: California, Canary Islands, detection, *Lampropeltis californiae*, morphology, pet trade

INTRODUCTION

Invasive species on islands drive high levels of extinction globally (Jones, et al., 2016). No examples of eradications of invasive snakes are known from islands (DIISE, 2015). Unlike mammals, where successful methods of eradication exist and great conservation success has been achieved (Jones, et al., 2016), snakes continue to invade cryptically, often with dramatic impacts (Willson, 2017). The accidental introduction of the brown tree snake (*Boiga irregularis*) to Guam has led to the loss of almost all the bird and much of the lizard diversity of the island (Rodda & Savidge, 2007). When this invasion was recognised, major changes in the biodiversity of the island had already taken place (Savidge, 1987; Rodda & Fritts, 1992). The brown tree snake, like several other snake invaders, is poorly known biologically in its native range, and thus any biological changes to the invader during the invasion cannot be easily detected (Rodda & Savidge, 2007).

One of the main pathways for introductions of reptiles is the pet trade, which is linked to many invasive species issues globally (Krysko, et al., 2016; McFadden, et al., 2017). Little is known about the effects of having captive raised snakes released into the wild. In addition, there is little information regarding the biology (morphology, reproduction, behaviour, etc.) of non-native snakes when they are introduced to islands. California king snakes (*Lampropeltis californiae*; CKS) were originally caught and bred for the pet trade, and many are from San Diego County, California. The CKS has been a major element of the international pet trade since the 1980s (Hubbs, 2009). They have been artificially selected for certain coloration and pattern phenotypes in captivity, including albino, striped, and banded. They were originally imported to the Canary Islands as well as many other places to be bred in captivity and sold as pets. They were released accidentally or escaped into the wild and have subsequently been on the Canary Islands as an invasive species since the late 1990s, adversely affecting the native wildlife and currently occurring in two discrete populations (Cabrera- Pérez, et al., 2012; Monzón-Arguello, et al., 2015). There have been perceived morphological changes in the snakes, and their expansion could be exponential as they irrupt without competition or predation (Cabrera- Pérez, et al., 2012). When trying to compare the invasive snakes with those in their natural habitat, we found that there is little known of the life history of CKS from their native range, especially southern California, and most references cite only the regional field guides, without much primary literature to support this information. Recently for the first time, movement data, which is very useful for understanding the invasion process, has been published for this species (Anguiano & Diffendorfer, 2015).

The Canary Islands are isolated oceanic islands off the coast of West Africa. They have low biodiversity, but high endemism, with some species that have important adaptations (Rando, et al., 2008; Fernandez-Palacios, et al., 2011). These include endemic lizards, of which the lacertids (*Gallotia* spp.) are herbivorous and are important seed dispersers (Valido, et al., 2003). The islands contain no native species of snake. On the Canary Islands, the invasive CKS have become a major predator for all of the native lizard species and are therefore threatening this island's biodiversity (Cabrera- Pérez, et al., 2012; Monzón-Arguello, et al., 2015). As with other invasive species, CKS on the Canaries have gone after the most abundant prey first, so they have been preying on the native lizards primarily and then secondarily on invasive small mammals. Birds do not make up a large part of their diet yet (Cabrera- Pérez, et al., 2012), but there are endangered birds present that might become snake prey over time as other prey become exhausted (Carrascal, et al., 2017). In addition, there are limited control efforts over the spread of the snakes on the Canary Islands and potentially all of Macaronesia (Azores, Madiera, and Cape Verde Islands). This could potentially threaten the biodiversity of the entire area if they are not eradicated. The snakes appear to have no predators in the Canary Islands.

How snakes invade and the dynamics of the early invasion process, in particular the changes to their phenology, phenotype, and reproduction during the irruption phase, have not been previously studied. Most

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snake invasions are more mature before study. The Canary Islands offer a unique opportunity to study these issues as it is a novel environment for snakes, and the snake invading is a species from the mainland of North America where numerous museum specimens and other field data are available. Because CKS are relatively well known, developing detailed life history parameters should be more straightforward than for other poorly known tropical species of snakes, such as brown tree snakes or Burmese pythons (*Python bivittatus*). The CKS is widespread from southern Oregon, south to the tip of the Baja Peninsula in Mexico, and east to mid-Nevada, southern Utah and the majority of Arizona; throughout its range it occurs naturally with many other snake species. The goal of this paper is to use museum and field datasets to resolve critical life history traits for this species, which can help to interpret CKS invasion dynamics within the Canary Islands and may be useful for optimising eradication/control techniques and efforts (i.e. trapping timing and placement).

MATERIALS AND METHODS

To document potential biological changes in the snake's natural history during the invasion process, we sampled CKS in their native range across 22.8°N to 40°N and made comparisons with the invasive snakes. Most samples were from southern California. Data were collected from 1,538 museum specimens (California Academy of Sciences, Natural History Museum of Los Angeles, San Diego Natural History Museum, University of California, Santa Barbara Cheadle Center for Biodiversity & Ecological Restoration) and augmented with records from wild caught CKS delivered to the San Diego Zoo (electronic supplementary materials). Additionally, we used southern California field data from 778 CKS captured between 1995 and 2012 in pit-fall and snake trap arrays by USGS (methods from Fisher, et al., 2008; electronic supplementary materials). These data from southern California included all snake species caught in these traps (n=4,708) and were used to assess the capture rate ranking of the CKS species compared to the other 24 native snake species for which we had contemporary capture data from these traps. We also obtained two different field datasets from the native range for CKS. One was a citizen science dataset from HerpMapper (HerpMapper, 2017) which had 1,299 records for the snakes from which we used capture/detection dates. The second was an unpublished dataset from Brian Hinds (BH) which represented 717 detections with associated observation dates. We compared these four native-range datasets to the Canary Island dataset, which encompassed 668 snakes (hand and trap caught from 2012 to 2014) on Gran Canaria Island (28°N), all from the western of the two populations on the island.

The museum specimens of CKS were measured for snout-vent length (SVL) and tail length using measuring tapes. Adults were defined as >600 mm in SVL (Hubbs, 2009). Sex was determined either through dissection or tail length and width. Some snakes were found dead on road (DOR) and the sex could not be determined. Many of the older museum specimens were missing reproductive systems; therefore, only a subset of data was available from these. Specimens missing their organs were used for length comparisons, but not for sex or reproductive status. Dorsal patterning and evidence of tail breaks were recorded and tail breaks were documented photographically.

The pit-fall and snake trap samples were collected from the wild in the native range in southern California primarily from south of Los Angeles to the Mexican border. Individuals were sexed, weighed, measured, and released. Data for colour pattern and tail status were lacking for most specimens. We also analysed the total capture for all snake species from these traps to look at the relative capture success of CKS compared to all other snakes for which we had data in the native snake community in California. To further look at activity phenology within their native range, we used data from HerpMapper (2017) and BH to assess observations by month as a recent sample to compare against our older native range data sources. Many of these records are from active searches under artificial cover (AC), and others are from night driving. Both of these are techniques that might have high seasonal biases in detections. This is because snakes under AC could be non-active, but using the cover to environmentally thermoregulate; whereas snakes detected on roads at night would be animals that are actively moving. These behaviours would change seasonally based on climatic conditions.

Samples from Gran Canaria Island were collected by hand or by trap then euthanized and frozen for later dissection. Sex, weight, SVL, tail breaks or scarring, were recorded.

Comparisons were made among these five study population samples for the relevant metrics and controlled for differences in sampling types. For example, the museum series is similar to the invasive population in that animals were collected by hand, trap, or opportunistically, but no comparison of weight could be completed, as the preserved weight of the museum snakes is not comparable to live weight. In contrast, live weight and length of the pit-fall and snake trap series could be compared to the invasive series, but reproductive states could not be compared, as these data were not available for the trapped and released snakes from their native range. These trap records are from snakes that are actively foraging, as they have to be moving in the landscape to encounter a trap. The last two field data sets (HerpMapper and BH) could only be used for detection/capture date comparisons with the other data sets, as they involved primarily active searches, especially under artificial cover, and not necessarily surface-active snakes. They also lacked length/weight measurements for individual snakes. We used means of the top decile to highlight comparisons between populations.

RESULTS

Snake community structure in California

Within a community of 25 native snake species captured via pit-fall and snake trap arrays in southern California, CKS was found to be the second most abundant species following the California whipsnake (*Masticophis lateralis*) and represented approximately 17% of the 4,708 captures across these species (Fig. 1). Snakes in this dataset were captured when snakes entered traps; no active searching for snakes took place. Thus, these records would be biased towards species more frequently moving over the landscape. These data indicate that within its native range the CKS is one of the most abundant snakes captured with this technique.

Trap success by size class

Using the USGS pit-fall and snake trap dataset, we were able to look at the effect of trap type on capture success by snake length, as a proxy for age (Fig. 2). We found that pit-fall trap buckets (18.9 L) buried in the ground were most successful, capturing snakes less than 500 mm in length. Wire-mesh snake traps had the greatest success with snakes exceeding 500 mm in length. Additionally, there was no trend in body size of CKS incidentally observed while conducting sampling using these traps.

Snake detections by month

We plotted the monthly detections/captures across five different datasets to assess variability across months.

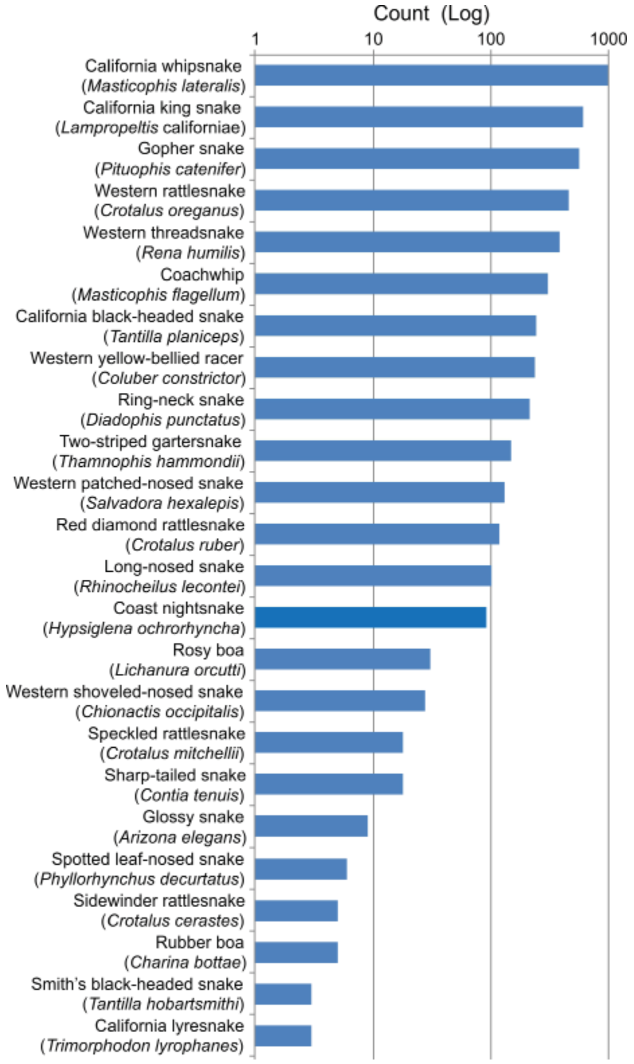

Fig. 1 All snake captures in southern California from the USGS pit-fall and snake trap study (n=4,708). *Lampropeltis californiae* is the second most common snake species captured.

Overall, monthly detections across datasets were highest between March and June, with the various peaks being due to variance in detection technique used. The citizen science (HerpMapper) and BH datasets, where they were actively searching for snakes, had peaks between March and April.

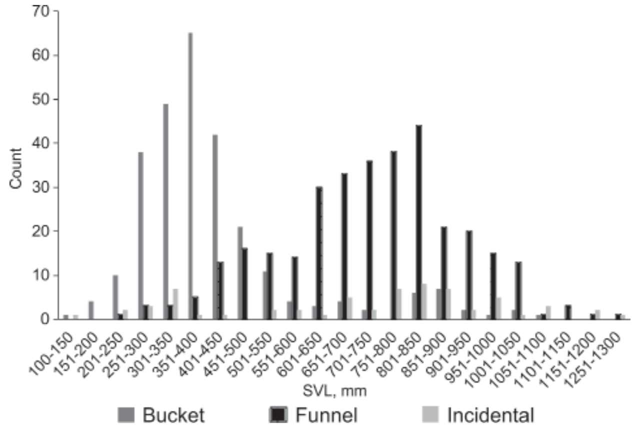

Fig. 2 Body size of *Lampropeltis californiae* by trap type for southern California. This figure has 50 mm breaks in size groups and highlights the different capture success of the two different trap types (pit-fall versus snake trap).

The museum and Canary Island datasets both had their peaks in the month of May, and these were identified using a variety of detection types, including active searching and traps. Finally, the pit-fall and snake trap dataset, with its passive traps for detections, had its peak in June. This last dataset was the only method based solely on active snakes. From August to January there was <10% per month of total snake detections across all datasets and from November to January there was <5% per month of total snake detections (Fig. 3).

Sex ratios, body size, and tail injury comparisons between California and the Canary Islands

We were able to make more detailed comparisons across three datasets, two from the native range (museum and pit-fall/snake trap) and the Canary Islands (Table 1). We found that there was a greater proportion of adults captured in the Canary Islands compared to the native range pit-fall/snake trap captures or museum specimens. There was no difference between the two native populations in the percent of juveniles, with about 49% of the samples representing juveniles; in contrast, only 22% of the invasive snakes were juveniles (Table 1). Thus, there were 2.3 times more juveniles detected in the native range than in the Canary Islands regardless of dataset used (museum or pit-fall/snake trap). For the pit-fall/snake trap and Canary Island captures, we compared the frequency by 50 mm size classes to see where this juvenile/adult bias was

Table 1 Morphological comparisons between native and invasive populations of *Lampropeltis californiae*. Differences between values of the invasive versus native populations were calculated as percentages to illustrate variance from 100%. Values in parentheses in table are sample sizes for top deciles.

	Gran Canaria Island	Southern California field	Museum specimens	Difference Gran Canaria Is. vs California
Total	668	780[a]	1,538[b]	
Total # adults (>600mm)	519	335	769	
Percent non-adults	21.9	49.6	48.4	0.44
Mean SVL (top decile) (mm)	1,071.7 (n=52)	1,069.1 (n=33)	1,032.2 (n=77)	1.00
Largest SVL (mm)	1,474	1,290	1197	1.14
Mean weight (top decile) (g)	412.8 (n=52)	334.9 (n=28)	-	1.23
Weight largest (g)	770.3	570	-	1.35
Tail break frequency	16.64	-	6.72	2.48

[a]670 with measurements that could be used
[b] except non-wild caught ~70 individuals

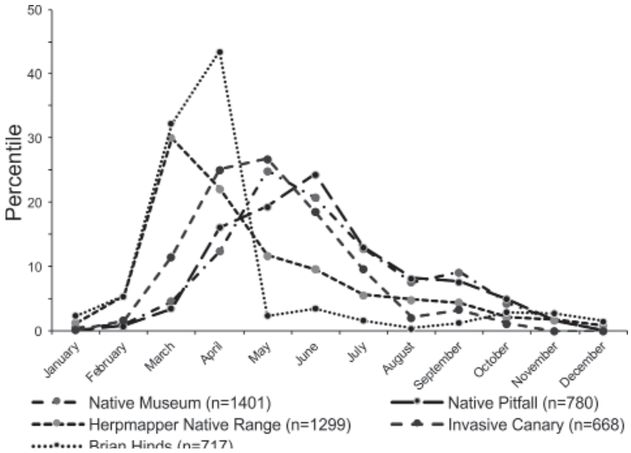

Fig. 3 Monthly percent of total detections of *Lampropeltis californiae* across the four datasets from the native range and the dataset from the invasive snakes on Gran Canaria.

the greatest. We found the native range had only one size class (351–400 mm) occurring in greater than 10% of the sample, whereas four consecutive size classes (701–900 mm) occur in greater than 10% each of the sample from the Canary Island. Thus, our data from the native range had a bimodal distribution between juvenile and adult captures compared to the Canary Island data (Fig. 4).

The invasive group of CKS did not have greater mean of the top decile compared to snakes in their native range (Table 1). The longest snake in the Canary Islands was 1,474 mm, 14% longer than the longest snake in the California sample (1,290 mm) and 21% longer than the next longest snake in the Canary Islands (1,217 mm). The invasive snakes had 23% greater average mass within the top decile compared to the USGS pit-fall/snake trap captures (Table 1). The heaviest snake in the Canary Islands was 770 g, 35% greater than the heaviest snake within the California sample (570 g).

One of the natural history traits we looked at was the frequency of tail breaks or scarring, as a proxy for predation risk. In the museum dataset, 6.7% of the snakes had broken tails, whereas 16.6% of the CKS on the Canary Islands had broken tails (2.48 times higher frequency of tail breaks compared with snakes in the native range) (Table 1). There was no noticeable association between tail break and colour pattern for either of these datasets.

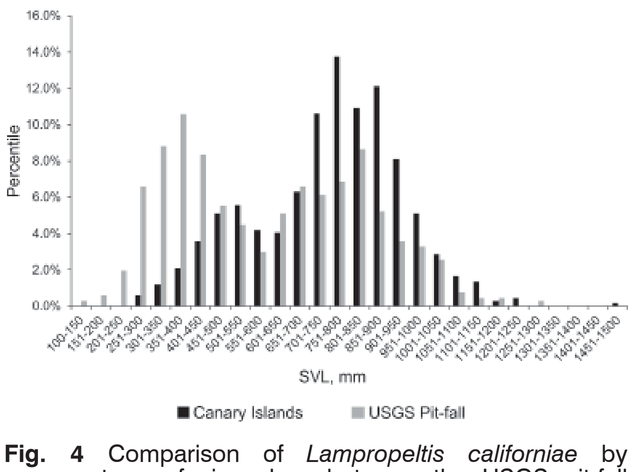

Fig. 4 Comparison of *Lampropeltis californiae* by percentage of size class between the USGS pit-fall snake trap dataset and the Canary Islands dataset. This graph highlights the lack of smaller size classes in the Canary Islands and the greater frequency of the larger size classes.

DISCUSSION

Since 2009, field work on control and eradication of the invasive CKS in the Canary Islands has resulted in the removal of over 4,500 snakes from the invaded habitats (<www.lifelampropeltis.com>). There was one population on Gran Canaria when the snakes were discovered, but now there are at least three populations on the island, indicating they are still spreading even with the control activities. We were able to compare various life history traits for native range CKS to the invasive range in the Canary Islands. Overall, we compiled records for 4,404 CKS for various aspects of their biology from the native range across four different data sources. These data were compared to 668 records for snakes from the Canary Islands. Below, we make comparisons on their biology and then suggestions on how they might be controlled or managed as an invasive species.

Snake community structure in California

We found that CKS was the second most captured species across the 25 species detected by the USGS pit-fall and snake trap sampling in California (Fig. 1). This sampling is based on the species actively entering the traps, and since the traps are passive, they only detect snakes when the snakes are active. Klauber (1931), using primarily road-riding for eight years (1923–1930), found that CKS were the third most detected snake species in his sample. They comprised 14% of the total record of 6,231 snakes across 24 species he detected for San Diego County, following the gopher snake (*Pituophis catenifer*) and the two-striped garter snake (*Thamnophis hammondii*). As our data were collected 70 years after his, this difference could represent actual changes in the abundance of the snakes due to habitat shifts over time, but it most likely represents the different sampling techniques. Both studies found CKS to be in the top three most captured snakes in the region across habitat types, indicating that even in a diverse snake community, CKS is one of the dominant species. This suggests that as an invasive species, it possibly could be successful even in regions with native snake communities, such as mainland Europe. Within the Canary Islands, it appears to have the ability to broadly utilise the habitats present on these islands.

Trap success by size class and lack of juvenile snakes in the Canary Islands

It was a quite striking find that juveniles are not detected in high numbers in the Canary Islands yet the snake is clearly expanding its range every year. This is very difficult to explain. The juvenile detection could be affected by several factors, including trapping technique, foraging distances and activity, growth rate, etc., but with the data we have to date we cannot determine the source of this issue. We know that sampling techniques to detect snakes vary in their effectiveness. We found a distinctive pattern of smaller snakes (<500 mm) being detected primarily by bucket traps (Fig. 2). This indicated there was a size bias in the sampling, with the buckets being necessary to capture the smaller snakes (<500 mm) and the mesh wire snake traps having greater success with the larger snakes (>500 mm) (Fig. 2). In the Canary Islands bucket traps are not being used (Cabrera- Pérez, et al., 2012; Monzón-Arguello, et al., 2015), and this could possibly explain the lack of juveniles being collected in the invasive range (Table 1). However, the museum specimens from animals captured in the wild in California include juvenile snakes, suggesting their absence could be due to something implicit in the Canary Islands. It could be there is some increased predation within the Canary Islands targeting juveniles, but if that was the case, the population might not be expanding as rapidly as it appears to be spreading.

It is likely there is a greater abundance of naïve prey in the Canary Islands reducing the need for juvenile snakes to move long distances to forage, thus limiting their exposure for detection or as prey. Abundant food resources might also increase their growth rate so that detecting individuals while they are still juveniles would be more difficult. When prey presence in captured snakes was evaluated for 270 individuals in the Canary Islands, 36% of these snakes had at least one prey item in their digestive tract (Monzón-Arguello, et al., 2015). In contrast, within their native range, a recently published study found only about 8% of the snakes assessed contained prey items in their digestive tract (Wiseman, et al., 2019). This suggests that the invasive snakes are finding prey at four times the rate of snakes within their native range, which could be a proxy for increased prey abundance in the Canary Islands.

Another possible explanation for lower detection rates of juveniles might be their activity levels compared to adults. Juveniles might only be active when foraging and under cover items between foraging bouts, while adults are active while foraging and also when searching for mates for reproduction, thus even though foraging exposure might be reduced for adults in the Canary Islands, they are still exposed for capture during mating season. Overall this could result in the lower detection of juveniles in the invasive range versus the native range, because the high food availability which could lead to rapid growth rate in the Canary Islands might limit detection probability (Pike, et al., 2008).

Snake detections by month

The effectiveness of detection tools varied with the time of year. Active searches under artificial cover (HerpMapper and BH) were more effective early in the year (March and April) before snakes were fully active as they used cover to thermoregulate (Fig. 3). We found that pit-fall and snake traps which are dependent on active snakes to enter the traps were more effective in May and June. Overall, focused field effort with various sampling techniques from March to July would maximise the detection success for CKS versus other months of the year. November, December, and January had the lowest detection rates across all five datasets, indicating that lowering field efforts during that period of time would be justified.

Body size and tail injury comparisons between California and the Canary Islands

We found no difference in mean SVL of the top decile between snakes in the invasive range versus the native range (Table 1). This result indicates that there has not been a population shift to longer body size within the invasive range, although the maximum length of the largest snake in the Canary Islands was 14% longer than any California snake, and 21% longer than the next largest snake in the Canary Islands. This snake was an outlier, as it was greater than three standard deviations longer than the next longest snake in the Canary Islands. As this snake was the second heaviest snake we don't think this resulted from measurement or recording error. This lack of population shift in body size contrasts with what has been observed in other invasive species, some of which have been shown to grow larger within their invasive range (Rodda & Savidge, 2007), but this outlier snake indicates that this pattern could change as the age since invasion gets longer. We did find that the invasive snakes were 23% heavier for the top decile, and the heaviest invasive snake was 35% larger than the heaviest snake from the California trap study (Table 1). Increased weight in invasive snakes is most likely tied to their increased predation success on naïve prey.

We observed a higher percentage of tail breaks and scarring of the snakes in the Canary Islands. This could be due to incomplete predation from cats (*Felis catus*) or other predators, from defensive wounds of their prey (e.g. *Gallotia stehlini*), or possibly some other unknown process (Medina & Nogales, 2009; Santos, et al., 2011). Increased frequency of tail breaks does not necessarily affect body condition, for some species (Pleguezuelos, et al., 2013). Within the snakes' native range, predators may be more efficient resulting more often in complete predation, especially by raptors, leaving fewer individuals with incomplete predation scars.

Trophic cascades

A major concern with novel invasive species is that their removal of highly specialised endemic species with unique roles in the island ecosystems may result in unexpected downstream changes in biodiversity and in the landscape. The Canary Islands have a small but unique and ancient biodiversity that could be highly susceptible to perturbations from invasive species (Fernandez-Palacios, et al., 2011). One example is the endemic *Gallotia* lizard which is an essential part of the trophic cascade/feedback loop that enables the dispersal of trees on the Canary Islands (Valido, et al., 2003). The lizards eat the fruit off the trees and shrubs, effectively spreading the seeds of the endemic flora. The invasive CKS are consuming these lizards at a high rate, with complete removal of juveniles in areas where snakes have invaded, and over time will impede the proliferation of these native trees and shrubs, altering the biodiversity and native habitat (Cabrera- Pérez, et al., 2012; Monzón-Arguello, et al., 2015). Published examples of trophic cascades tied to snake invasions include the relationship between spiders and birds in Guam now caused by the snake irruption, and the dynamics of python and mid-sized mammals in Florida (Rogers, et al., 2012; Willson, 2017).

The CKS has a varied diet in its native range, including venomous snakes and juvenile birds (Morrison & Bolger, 2002). Because there are currently no birds recorded in the diet of the invasive snakes (Monzón-Arguello, et al., 2015), initiating intensive sampling of birds in areas with and without snakes to get an assessment of bird density and recruitment may be valuable. From the literature it seems clear that these snakes could target birds, many of which are endemic and some are currently endangered, as prey as they exhaust the lizards and rodents present (Morrison & Bolger, 2002; Carrascal, et al., 2017). This may also be valuable because the published diet data are five years old, and there might already be a change in their diet if there is a depletion of the main reptile and rodent prey.

If it looks like the snakes are going to achieve an island-wide distribution, then one approach is to pre-emptively safeguard various biologically intact areas around the island at different elevations. This approach could preserve biodiversity and create reservoirs of native animals in the event that the snake control/eradication fails.

Pet trade and captive breeding/selection and then released into wild

The invasive CKS has a unique history as it came from several generations of selection in captivity for various colour morphs and albinism, in addition to rapid growth and reproduction. Their release to the wild in the Canary Islands is concerning as this selection might provide some reproductive advantage versus the release of wild animals not subjected to selection in captivity. This trade of potential invasive species is concerning as more and more reptile species become bred for sale globally in the pet trade (Robinson, et al., 2015).

Considerations for snake management in the Canary Islands

Looking at CKS published movement data suggests that placing snake traps with sterile female snakes, or proxies, less than every 150 m apart may be effective for snake management. This distance may be appropriate because the literature indicates that 98% of the males and 100% of the females radio-tracked do not move farther than this (Anguiano & Diffendorfer, 2015). Having a grid of traps in closer proximity across the snake-occupied parts of the island would be optimal for a snake removal programme.

There are large ecological and monetary costs to invasive animals, and costs of control and/or eradication often exceed the available funding. We suggest (1) stronger controls on snakes in the pet trade, (2) rapid response to prevent spread when detection first occurs, and (3) use of citizen science as a tool to detect early invasions.

CONCLUSION

Our results show that data from the native range of the snake can inform management and control for CKS within their invasive range. Also, we found that they flourish within a diverse native snake community; they have a high natural abundance, both historically (Klauber, 1931) and currently (Fig. 1).

We suggest that the continued use of a variety of traps in addition to active surveys be used to maximise detection of snakes of all sizes, especially within the months of March through July. We also suggest that managers consider protection of natural areas with critical biodiversity on Gran Canaria from invasion by CKS. In addition, managers may wish to consider increased controls to prevent spread to other areas in the Canary Islands.

There is no literature on where the CKS lays its eggs in its natural habitat or in the Canary Islands. A comparison of this and other reproductive characteristics may be important as well as a better understanding of how to detect juveniles within the invasive range. Greater support for risk assessments of species, within the pet trade in particular, could help to identify species of greatest concern which would help reduce these types of invasions elsewhere.

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REFERENCES

Anguiano, M. and Diffendorfer, J. (2015). 'Effects of fragmentation on the spatial ecology of the California kingsnake (*Lampropeltis californiae*)'. *Journal of Herpetology* 49: 420–427.

Cabrera-Pérez, M.Á., Gallo-Barneto, R., Esteve, I., Patiño-Martínez, C. and López-Jurado, L.F. (2012). 'The management and control of the California kingsnake in Gran Canaria (Canary Islands): Project LIFE+ Lampropeltis'. *Aliens: The Invasive Species Bulletin Newsletter of the IUCN/SSC Invasive Species Specialist Group* 32: 20–28.

Carrascal, L.M., Moreno, A.C., Delgado, A., Suarez, V. and Trujillo, D. (2017). 'Habitat suitability – density relationship in an endangered woodland species: the case of the blue chaffinch (*Fringilla polatzeki*)'. *PeerJ* 5: e3771.

DIISE. (2015). 'The Database of Island Invasive Species Eradications, developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand'. <http://diise.islandconservation.org>. Accessed 3 September 2017.

Fernandez-Palacios, J.M., de Nascimento, L., Otto, R., Delgado, J.D., Garcia-del-Rey, E., Arevalo, J.R. and Whittaker, R.J. (2011). 'A reconstruction of Palaeo-Macaronesia, with particular reference to the long-term biogeography of the Atlantic island laurel forests'. *Journal of Biogeography* 38: 226–246.

Fisher, R.N., Stokes, D., Rochester, C., Brehme, C., Hathaway, S. and Case, T.J. (2008). *Herpetological Monitoring Using a Pitfall Trapping Design in Southern California*. Reston, Virginia: US Geological Survey.

HerpMapper. (2017). 'HerpMapper – A Global Herp Atlas and Data Hub. Iowa, U.S.A'. <www.herpmapper.org>. Accessed 1 September 2017).

Hubbs, B. (2009). *Common Kingsnakes: A Natural History of* Lampropeltis getula'. Tempe, AZ: Tricolor Books.

Jones, H.P., Holmes, N.D., Butchart, S.H., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P.E., Cuthbert, R.J.1., Ebbert, S., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R. and Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences* 113: 4033–4038.

Klauber, L.M. (1931). 'A statistical survey of the snakes of the southern border of California'. *Bulletin of the Zoological Society of San Diego* 8: 1–93.

Krysko, K.L., Somma, L.A., Smith, D.C., Gillette, C.R., Cueva, D., Wasilewski, J.A., Enge, K.M., Johnson, S.A., Campbell, T.S., Edwards, J.R., Rochford, M.R., Tompkins, R., Fobb, J.L., Mullin, S., Lechowicz, C.J., Hazelton, D. and Warren, A. (2016). 'New verified nonindigenous amphibians and reptiles in Florida through 2015, with a summary of over 152 years of introductions'. *International Reptile Conservation Foundation Reptiles and Amphibians* 23: 110–143.

McFadden, M.S., Topham, P. and Harlow, P.S. (2017). 'A ticking time bomb: Is the illegal pet trade a pathway for the establishment of corn snake (*Elaphe guttata*) populations in Australia?' *Australian Zoologist* 38: 499–504.

Medina, F.M. and Nogales, M. (2009). 'A review on the impacts of feral cats (*Felis silvestris catus*) in the Canary Islands: Implications for the conservation of its endangered fauna'. *Biodiversity and Conservation* 18: 829–846.

Monzón -Arguello, C., Patino-Martinez, C., Christiansen, F., Gallo-Barneto, R., Cabrera- Pérez, M.A., Pena-Estevez, M.A., Lopez-Jurado, L.F. and Lee, P.L.M. (2015). 'Snakes on an island: Independent introductions have different potentials for invasion'. *Conservation Genetics* 16: 1225–1241.

Morrison, S.A. and Bolger, D.T. (2002). 'Lack of an urban edge effect on reproduction in a fragmentation-sensitive sparrow'. *Ecological Applications* 12: 398–411.

Pike, D.A., Pizzatto, L., Pike, B.A. and Shine, R. (2008). 'Estimating survival rates of uncatchable animals: The myth of high juvenile mortality in reptiles'. *Ecology* 89: 607–611.

Pleguezuelos, J.M., Feriche, M. and Santos, X. (2013). 'Tail-breakage effects on snake-body condition'. *Zoologischer Anzeiger* 252: 243–245.

Rando, J.C., Alcover, J.A., Navarro, J.F., Garcia-Talavera, F., Hutterer, R. and Michaux, J. (2008). 'Chronology and causes of the extinction of the lava mouse, *Malpaisomys insularis* (Rodentia: Muridae) from the Canary Islands'. *Quaternary Research* 70: 141–148.

Robinson, J.E., Griffiths, R.A., St. John, F.A.V. and Robert, D.L. (2015). 'Dynamics of the global trade in live reptiles: Shifting trends in production and consequences for sustainability'. *Biological Conservation* 184: 42–50.

Rodda, G.H. and Fritts, T.H. (1992). 'The impact of the introduction of the brown tree snake, *Boiga irregularis*, on Guam's lizards'. *Journal of Herpetology* 2: 166–174.

Rodda, G.H. and Savidge, J.A. (2007). 'Biology and impacts of Pacific island invasive species. 2. *Boiga irregularis*, the brown tree snake (Reptilia: Colubridae)'. *Pacific Science* 61: 307–324.

Rogers, H., Hille Ris Lambers, J., Miller, R. and Tewksbury, J.J. (2012). ''Natural experiment' demonstrates top-down control of spiders by birds on a landscape level'. *PLOS ONE* 8(4): e43446.

Santos, X., Feriche, M., León, R., Filippakopoulou, A., Vidal-García, M., Llorente, G.A. and Pleguezuelos, J.M. (2011). 'Tail breakage frequency as an indicator of predation risk for the aquatic snake *Natrix maura'*. *Amphibia-Reptilia* 32: 375–383.

Savidge, J.A. (1987). 'Extinction of an island forest avifauna by an introduced snake'. *Ecology* 68: 660–668.

Valido, A. Nogales, M. and Medina, F. (2003). 'Fleshy fruits in the diet of Canarian lizards *Gallotia galloti* (Lacertidae) in a xeric habitat of the island of Tenerife'. *Journal of Herpetology* 37(4): 741–747.

Willson, J.D. (2017). 'Indirect effects of invasive Burmese pythons on ecosystems in southern Florida'. *Journal of Applied Ecology* 54: 1251–1258.

Wiseman, K.D., Greene, H.W., Koo, M.S. and Long, D.J. (2019). 'Feeding ecology of a generalist predator, the California kingsnake (*Lampropeltis californiae*): why rare prey matter.' Herpetological Conservation and Biology (in press).

Control of the ladder snake (*Rhinechis scalaris*) on Formentera using experimental live-traps

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Abstract The ladder snake (*Rhinechis scalaris*) is a recent alien invasive species found on Formentera (83 km^2), in the Balearic Archipelago (4,492 km^2). It has been introduced in the last decade as cargo stowaway hidden within ornamental olive trees from the Iberian Peninsula, causing negative impacts on native fauna. This paper describes the methodology used to reduce the ladder snake population as a first attempt since it was detected in 2006. For this purpose, an experimental live-trap was designed by the wildlife management team of the Consorci per a la Recuperació de la Fauna de les Illes Balears (COFIB) during the 2016 campaign. As a result, 314 *R. scalaris* were trapped in an area of 472 ha, achieving an efficiency of up to 0.167 captures per trap and night, and 0.040 captures per unit effort on average. This outcome encourages the use of the live-trap as a cost-effective method for reducing the snake population in Formentera. Nonetheless, this method should be considered a starting point toward *R. scalaris* control.

Keywords: alien species, Balearic Islands, ophidian, population

INTRODUCTION

The accidental transportation of invasive alien species to new locations is a major cause of biodiversity loss worldwide. This is of special concern in island ecosystems, where native species are especially vulnerable to biological invasions (Quammen, 1996). In this regard, the presence of reptiles in the Balearic Islands is a paradigmatic case, with a greater number of alien species (19) than native ones (2), namely, the Lilford's wall lizard (*Podarcis lilfordi*) and the Ibiza wall lizard (*Podarcis pityusensis*) (Silva-Rocha, et al., 2015). The ladder snake (*Rhinechis scalaris*) is a Mediterranean species which is present in most of the Iberian Peninsula (just missing on the Cantabric ledge), in the south-east of France and the north-east of Italy (Pleguezuelos & Honrubia, 2002). This ophidian is also considered an introduced species on other Spanish islands, namely, Ons, Aurosa (in Pontevedra), Mallorca, Menorca, Ibiza and Formentera (in the Balearic Islands) (Pinya & Carretero, 2011), but fortunately not on any of the surrounding islets (Carretero & Silva-Rocha, 2015). However, in Menorca *R. scalaris* has a wide distribution and is catalogued as a protected species in the Catàleg Balear d'Espècies Amenaçades (Decret, 2005) due to its presence on the island dating from the pre-Roman period (Vigne & Alcover, 1985). Conversely, on Mallorca, Ibiza and Formentera *R. scalaris* is a recent introduction (Álvarez, et al., 2010; Mateo & Ayllón, 2012), so its presence is still isolated to particular locations and its range is expanding. In fact, in Ibiza and Formentera this ophidian is catalogued as an invasive alien species (Real Decreto, 2013).

Until 2006 Formentera was considered snake-free. The first *R. scalaris* was detected on 25 May 2008, followed by another sighting on 17 July 2008; both located near La Mola. Then, a third specimen, not identified, was recorded the 20 May 2009 (Álvarez, et al., 2010; Mateo & Ayllón, 2012). It is presumed that the first ophidian was introduced to the Pityusic islands through the trade of ornamental olive trees originating from the Iberian Peninsula (Álvarez, et al., 2010; Carretero & Silva-Rocha, 2015; Montes, et al., 2015), and genetic studies suggest that the whole *R. scalaris* population comes from one introduction event (Silva-Rocha, et al., 2015). Nonetheless, it would be expected that all the snakes spotted in Formentera during the first years could come from Ibiza, since direct connections between Formentera and the mainland are rather limited (Álvarez, et al., 2010; Mateo & Ayllón, 2012).

The naturalisation of this ophidian could result in important consequences for the Pityusic ecosystem and also for the demographic stability of the endemic Ibiza wall lizard (Rodríguez-Pérez, 2009; Álvarez, et al., 2010). Previous cases of introduction of snakes to island ecosystems have been terrible in terms of ecological balance as experienced by the ancient settlement of ophidians on the neighbouring islands of Mallorca and Menorca (SPE, 2007), the deliberate release of the Californian kingsnake (*Lampropeltis getula californiae*) on Gran Canaria (Cabrera-Pérez, et al., 2012) and the accidental introduction of the brown treesnake (*Boiga irregularis*) to the island of Guam (Savidge, 1987; Rodda, et al., 1997; Fritts & Rodda, 1998; Wiles, et al., 2003).

In the last decade, sightings from local people have increased and as Carretero & Silva-Rocha (2015) stated, "the area of Formentera where ladder snakes were spotted in the past, should be checked thoroughly and regularly". So, the need to monitor the presence of *R. scalaris* on Formentera is a real concern.

The present paper reports the first experience of trying to catch and reduce the presumed population of *R. scalaris* in the vicinity of La Mola in Formentera during the 2016 campaign. For this purpose, an experimental live-trap was designed by the wildlife management team of the Consorci per a la Recuperació de la Fauna de les Illes Balears (COFIB), along with the Government of the Balearic Islands. Budget constraints restricted the scope of this first campaign to confirming and mapping the presence of the ladder snake in the vicinity of La Mola. So, the aim was to determine the effectiveness of the trap, as defined by captures per unit effort (CPUE), in order to establish a starting point for future campaigns.

MATERIALS AND METHODS

Study area

This study was conducted on Formentera, the smallest (83 km^2) and southernmost island of the Balearic archipelago

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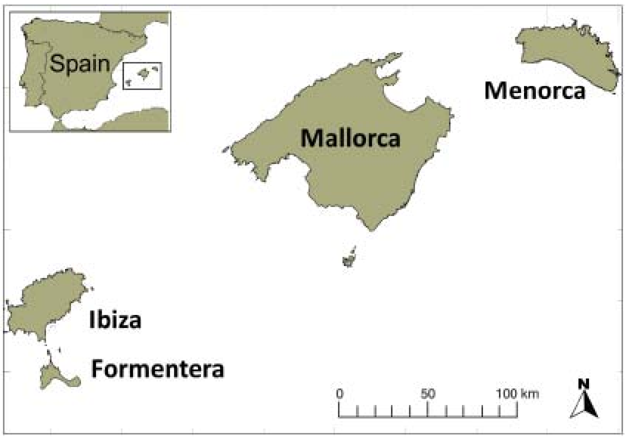

Fig. 1 Map of the Balearic Islands, showing the name of the main islands.

(Fig. 1). A channel of 3.6 km separates Formentera from the other Pityusic Island, Ibiza, and it is 100 km away from the Iberian Peninsula. Vegetation consists of sand dunes with pine forest, oak groves and brushwood. It is considered a flat island, with the highest point being La Mola, at 192 m above sea level, in the south-east of the island. This is where the *R. scalaris* population seems to be concentrated, thus, the study focuses on this area

The live-trap

To conduct this study, live-traps were designed by the COFIB for the purpose of capturing colubrid snakes. The trap used on Formentera was the same as those used in the project "Análisis de la efectividad de métodos de control de especies exóticas invasoras de la familia *colubridae* en islas" (COFIB, 2016) that took place simultaneously in a parallel campaign on Ibiza.

The trap measures 50.0 × 35.5 × 17.0 cm and is made of marine plank (1 cm thick) in order to endure inclement weather conditions (Fig. 2). The box consists of two compartments separated by a galvanised steel mesh of 5 × 5 mm, with two large doors on top to allow snake removal and bait maintenance. These doors are secured with a bolt in order to prevent escapes. The front side is also made of a galvanized steel mesh, allowing air flow through the mesh

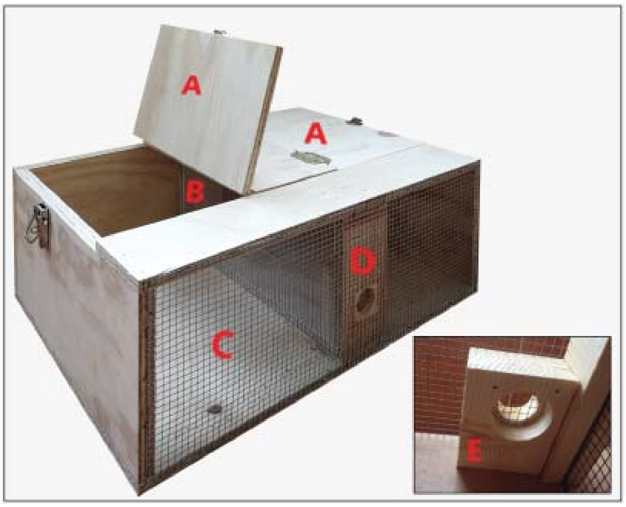

Fig. 2 Model of the live-trap designed by the COFIB. A: the two doors of the top, B: galvanized steel mesh that separates the two compartments, C: galvanized steel mesh of the front side, D: one-way flap door, E: detail of the trapping door viewed from the inside.

and the opportunity to glimpse the animals. Therefore, the trap is not completely opaque. There is just one entrance with a one-way flap door positioned on the mesh front of the trap, with a diameter of 3.5 cm. The flap only opens inwards, falling closed behind the snake to prevent escape and allowing multiple captures. This one-way flap entrance design has been used on a number of snake trap designs (reviewed by Rodda, et al., 1999a). Inside the snake compartment a hide is placed: a 300 mm length of 100 mm diameter plastic bottle, covered with 40 mm of water to prevent snake dehydration.

A live mouse, with enough water and food for optimal welfare, is used as attractant. In this trap, the bait is contained in a separate compartment to prevent the snake from ingesting the mouse.

Trapping method

In the 2016 campaign, trap boxes were placed in the area near La Mola (Fig. 3), mostly at the limit of pine forests, near stone walls or at the base of vegetation (Montes, et al., 2015), all of them at ground-level. We covered a total area of 472 ha with 64 traps. Fourteen extra traps were placed in different locations on the island where no snakes had been spotted in the past, as snake indicators. All the traps located in the field were georeferenced.

Every effort was made to keep the mice alive during the whole project, as they are the basis for the operation of the trap (Mateo & Ayllón, 2012). During the coldest months of the year dry grass or similar materials were provided and the boxes were placed in the sunlight, avoiding hypothermia. Conversely, in summer the boxes were moved slightly towards the shade, helping the mice to endure the suffocating heat. Also, for the duration of the rainy season, traps were placed on stones and covered with plastic, preventing contact between the bottom of the box and a waterlogged ground. These measures were taken not just for humane and economic reasons but also because they allowed a longer period between inspections. All traps were checked and bait replaced every nine days, on average.

Capture and data gathering

When an ophidian was captured, it was identified to ensure it was a *R. scalaris* (as opposed to an unknown and possibly venomous snake), so handling did not require

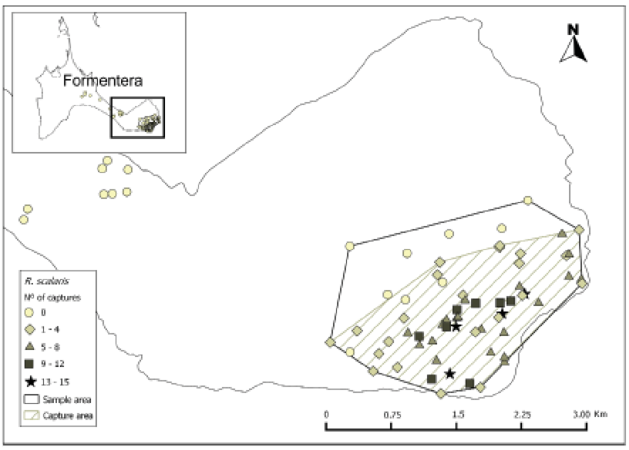

Fig. 3 Distribution of the ladder snake (*Rhinechis scalaris*) in Formentera in 2016. The number of captures per trap is represented with different shapes (see the key). The sample area and the capture area are determined by applying the minimum convex polygon method. All traps on the field are represented on the map but none of the indicator traps are included in the sample area.

Fig. 4 Two specimens of ladder snake (*Rhinechis scalaris*), a juvenile on the left and an adult on the right, caught with the same live-trap in the field on Formentera.

anything special other than a pair of gardening gloves (Fig. 4). We euthanised the snakes using pentobarbital (the approximate dose was 0.1 ml per 100 g) and all injections were performed by an experienced veterinarian, who was the main field technician. Once the specimen was lifeless, it was placed in a 'zip' bag with an identification label including trap number, species, and date. Each captured specimen was stored in a freezer for further investigation; no morphological data was collected on the field. Afterward, all captures were mapped in order to estimate the abundance of *R. scalaris* in La Mola, using the minimum convex polygons tool of Quantum GIS (1.8.0).

RESULTS

A total of 64 traps where placed on La Mola, remaining in the same location for the entire sampling period (Fig. 3). The trapping was conducted between May and late November. The team was based on Ibiza and, for this reason, both sea conditions and vehicle availability restricted the number of possible surveys to 19. The number of traps increased in nearly every survey till the end of July and August, when we had the total number of traps placed in the field. The capture area comprised 321 ha from a sampling area of 472 ha. No indicator trap had any capture.

By the end of the campaign, 314 ladder snakes had been caught, with a total of 7,906 trap-nights (Table 1). It is evident that this was a grass-roots effort, using the best available knowledge to catch as many snakes as possible while keeping costs as low as possible. Therefore, we did not have the time to estimate the density of snakes prior to the trapping. Instead of this, we evaluated the trap effectiveness as defined by captures per unit effort (CPUE). By the end of the project we had an average of 0.040 CPUE.

In May, the first month of trapping, we obtained a trap efficiency of 0.108 captures per trap and night. Next month, June, we got 0.075 captures per unit effort (CPUE), even though the number of traps in the field was more than double. A similar pattern occurred in the following months: the captures per unit effort continued dropping, until we got 0.006 CPUE in November. Therefore, preliminary data seems to indicate an encouraging trap capture decay rate, with a high CPUE at the beginning and a declining recovery from traps as the local supply of snakes depleted. However, seasonal changes in capture success need to be evaluated.

DISCUSSION

The 2016 trapping campaign is the first attempt to remove large numbers of snakes as a step towards controlling the invasion of *R. scalaris* on the island of Formentera. Previous attempts on the neighbouring islands of Ibiza (Montes, et al., 2015) and Mallorca (Mateo, 2015) have tested different methods to capture *R. scalaris* and *Hemorrhois hippocrepis*. After a thorough review of these documents, we decided to use a passive method to trap as many snakes as possible, continuing the work of Montes, et al. (2015) by adapting the wooden box they used. In this regard, we followed the advice and recommendations of previous snake trapping studies. As Rodda, et al. (1999a) showed, it is possible to have higher capture rates using live mice as lures, opaque chambers and flap entrances. Firstly, flap traps have a lower entry rate than open funnel traps, but the former have a higher capture rate. For this reason, we replaced the two open-funnels used by Montes, et al. (2015), with a single frontal flap door, as these are considered to have a negligible escape rate (Rodda, et al., 1999a).

Secondly, in contrast with the lack of a mouse's chamber in the wooden box by Montes, et al. (2015) and the small one that housed the mouse inside the funnel trap described by Mateo & Ayllón (2012), our trap had a proper shelter for the mouse, which was the second big compartment of the cage. With this modification we avoided snake ingestion of the lure and contributed to reducing mouse mortality.

Finally, in order to enhance capture success, refugium bottles were placed inside the snake's compartment as it has been observed that there is a significantly higher number of entries into traps having hiding places, even if the possibility of escape is unlikely (Rodda, et al., 1999a).

Our trap optimises previous designs and the positive capture rates seem to be the result of using both a flap door and a bottle refuge, as these contribute to reducing the number of snakes escaping, along with the separate compartments, which keeps the trap active after a first successful capture. Indeed, our results (0.040 CPUE) confirm a higher efficiency when compared with the study by Montes, et al. (2015) (0.007 CPUE).

All data could have been more accurate had we had a technician exclusively dedicated to checking the traps every other day. Then, not only the number of traps per hectare would have been greater, but the capture rate probably higher. In this case, re-check intervals were determined in relation to care and maintenance of live lures (as the snakes had enough water to avoid death by dehydration during these intervals) instead of capture rate increase. This allowed optimising labour and maximising cost-effectiveness. Even so, trap captures are hypothesised to be higher if the area of trapping is not disturbed (Rodda, et al., 1999a), suggesting normal entrance rates if checks are done within longer intervals. In this study, traps were checked weekly during the summer season but checks were done every 12 days in autumn.

Regarding trap location, Rodda, et al. (1999a), argue that traps should be widely spaced in order to maximise the capture rate when traps are infrequently checked. However, there is still a lack of a mathematic equation describing the relationship between capture rate and trap spacing, as well as a poorly understood interaction between trap design and the environment in which it is used. Taking into account that *R. scalaris* is an active forager (Pleguezuelos, et al., 2007), traps were placed as far apart as it was practical for revisits considering the topography, the trapping area and the number of traps available, resulting in a wide range from 50 m to 600 m apart.

Table 1. Data on ladder snakes (*Rhinechis scalaris*) caught during the 2016 campaign on Formentera.

Survey	Month	No. Traps	No. Captures	Trap/night	CPUE
1	May	9	3	36	0.083
2	May	18	15	90	0.167
3	May	18	12	157	0.076
4	May	43	37	336	0.110
	Average for May	**22**	**67**	**619**	**0.108**
5	June	56	41	306	0.134
6	June	57	40	392	0.102
7	June	57	36	497	0.072
8	June	63	13	532	0.024
	Average for June	**58.25**	**130**	**1,727**	**0.075**
9	July	63	29	635	0.046
10	July	63	11	420	0.026
11	July	64	2	506	0.004
	Average for July	**63.33**	**42**	**1,561**	**0.027**
12	August	64	14	408	0.034
13	August	64	9	408	0.022
	Average for August	**64**	**23**	**816**	**0.028**
14	September	43	6	352	0.017
	Average for September	**43**	**6**	**352**	**0.017**
15	October	43	14	645	0.022
16	October	44	18	602	0.030
17	October	44	7	396	0.018
	Average for October	**43.67**	**39**	**1,643**	**0.024**
18	November	44	5	616	0.008
19	November	44	2	572	0.003
	Average for November	**44**	**7**	**1,188**	**0.006**
		TOTAL	**314**	**7,906**	**0.040**

The first traps distributed on the ground were placed within view of neighbours (on the south of the road to the lighthouse of La Mola), and then a consecutive radial expansion was drawn. As can be seen in Fig. 3, there is a clear ‘hot spot’ to the south of La Mola, with the highest number of captures close to the southern coast. The number of captures decreases the further we move from this high-density area and the traps on the north and west boundaries are characterised by no captures (except for two traps on the west). Considering that the sea is a natural barrier, potential expansion is only possible to the north or to the west of the sample area. As mentioned above, no indicator traps in other parts of the island had any captures. Therefore, our trap array gives an initial indication about the range of *R. scalaris* on Formentera, having a higher density of snakes in the core of the invasion zone than at the edges. Still, a larger array of traps around La Mola, especially on the west boundary, would depict the range of *R. scalaris* more accurately.

It is clear that a population of *R. scalaris* is naturalised in Formentera. Previous extinctions of endemic birds and lizards have been documented as a result of the introduction of an alien snake, such as the well-known case of the *B. irregularis* in the Island of Guam (Savidge, 1987; Rodda, et al., 1997; Fritts & Rodda, 1998). Therefore, the Guam experience should made us wary of the invasive potential that *R. scalaris* could have on the native fauna of the Pityusic islands. It has the potential to affect a wide range of animals, such as the emblematic Ibiza wall lizard, the Balearic shearwater (*Puffinus maruritanicus*), the Scopoli's shearwater (*Calonectris diomedea*), the storm petrel (*Hydrobates pelagicus*) or the garden dormouse of Formentera (*Eliomys quercinus ophiusae*) (Hinckley, et al., 2016), as few predators are present on Formentera and the abundant endemic fauna is an easy and vulnerable target because prey species lack co-evolutionary experience with snakes (Rodda, et al. 1999b).

Successful control of *R. scalaris* is Formentera’s highest conservancy priority (Pleguezuelos, et al., 2015). This is an early invasion, in chronological terms, and the area of invasion seems to be relatively small. The use of this wooden box trap seems to be a useful starting point towards *R. scalaris* control. However, more comprehensive research is required to determine whether the ladder snake’s expansion on Formentera can be stopped by using this capture method. In order to assess this question, the study will continue in future years with a greater trap array.

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REFERENCES

Álvarez, C, Mateo, J.A., Oliver, J. and Mayol, J. (2010). 'Los ofidios ibéricos de introducción reciente en las Islas Baleares'. *Boletín de la Asociación Herpetológica Española* 21: 126–131.

Cabrera-Pérez, M.A., Gallo-Barneto, R., Esteve, I., Patiño-Martínez, C., and López-Jurado, L.F. (2012) 'The management and control of the California kingsnake in Gran Canaria (Canary Islands): Project LIFE+ *Lampropeltis*'. *Aliens: The Invasive Species Bulletin* 32: 20–28.

Carretero, A. and Silva-Rocha, I. (2015). 'La culebra de escalera (*Rhinechis scalaris*) en las Islas Baleares'. *Boletín de la Asociación Herpetológica Española* 26(2): 84–87.

COFIB (Consorci per a la Recuperació de la Fauna de les Illes Balears) (2016). *Análisis de la Efectividad de Métodos de Control de Species Exóticas Invasoras de la Familia Colubridae en Islas.* Unpublished report. Palma: COFIB.

Decret (2005). 'D.75/2005, de 8 de juliol, pel qual es crea el Catàleg Balear d'Espècies Amenaçades i d'Especial Protecció, les Arees Biològiques Crítiques i el Consell Assessor de Fauna i Flora de les Illes Balears'. Boletín Oficial de las Islas Baleares (BOIB) 106. <http://www.caib.es/sites/institutestudisautonomics/ca/n/decret_752005_de_8_de_juliol_pel_qual_es_crea_el_cataleg_balear_despecies_amenacades_i_despecial_proteccio_les_arees_biologiques_critiques_i_el_consell_assessor_de_fauna_i_flora_de_les_illes_balears_-58273/>.

Fritts, T.H. and Rodda, G.H. (1998). 'The role of introduced species in the degradation of island ecosystems: A case history of Guam'. *Annual Review of Ecology, Evolution, and Systematcs* 29: 113–140.

Hinckley, A., Montes, E., Ayllón, E. and Pleguezuelos, J.M. (2016). 'The fall of a symbol? A high predation rate by the introduced horseshoe whip snake *Hemorrhois hippocrepis* paints a bleak future for the endemic Ibiza wall lizard *Podarcis pityusensis*'. *European Journal of Wildlife Research* 63(1): 13.

Mateo, J.A. (2015). 'Los Anfibios y los Reptiles Introducidos en Baleares: un repaso a lo que sabemos y un ejemplo de puerta de entrada'. In: Servei de Protecció d'Espècies (ed.) *Llibre Verd de Protecció d'Espècies a les Balears,* pp. 447–454. Palma: Govern de les Illes Balears. Conselleria d'Agricultura, Medi Ambient i Territori.

Mateo, J. A. and Ayllón, E. (2012). *Viabilidad del Control de Ofidios en Ibiza y Formentera*. Palma: Asociación Herpetológica Española.

Montes, E., Estarellas, J., Ayllón, E., Carretero, M.A., Feriche, M., Hernández, P.L. and Pleguezuelos, J.M. (2015). 'Dades Preliminars del Projecte Pilot de Control de Serps a l'Illa d'Eivissa'. In: Servei de Protecció d'Espècies (ed.) *Llibre Verd de Protecció d'Espècies a les Balears,* pp. 455–464. Palma: Govern de les Illes Balears. Conselleria d'Agricultura, Medi Ambient i Territori.

Pinya, S. and Carretero, M.A. (2011). 'The Balearic herpetofauna: A species update and a review on the evidence'. *Acta Herpetologica* 6: 59–80.

Pleguezuelos, J.M. and Honrubia, S. (2002). '*Elaphe scalaris'*. In: J.M. Pleguezuelos, R. Márquez and M. Lizana (ed.) *Atlas y Libro Rojo de los Anfibios y Reptiles de España,* pp. 281–283. Madrid: Organismo Autónomo de Parques Nacionales.

Pleguezuelos, J.M, Fernández-Cardenete, J.R., Honrubia, S., Feriche, M., Villafranca, C. (2007). 'Correlates between morphology, diet and foraging mode in the ladder snake *Rhinechis scalaris* (Schinz, 1822)'. *Contributions to Zoology* 76(3): 179–186.

Pleguezuelos, J.M., Ayllón, E., Arribas, O., Bertoloero, A., Bosch, J., Cabido, C., Carranza, S., Carretero, M.A., Díaz-Paniagua, C., Egea-Serrato, A., Garin-Barrio, I., Giménez, A., Gosá, A., Garciá, E., Guicking, D., Llorente, G.A., Martínez-Solano, I., Mateo, J.A., Montori, A., Palomar, G., Perera, A., Pinya, S., Pretus, J.L., Pujol-Buxó, E., Rato, C., Recuero, E., Sanz-Azkue, I., Silva-Rocha, I., Vasconcelos, R., Velo-Antón, G., Vörös, J., Santos, X. (2015). 'Conclusiones y propuestas de gestión de las poblaciones y especies de anfibios y reptiles alóctonos en España'. *Boletín de la Asociación Herpetológica. Española* 26 (2): 113–115.

Quammen, D. (1996). *The Song of the Dodo: Island Biogeography in an Age of Extinction*. New York: Simon and Schuster.

Real Decreto (2013). 'RD.630/2013, de 2 de agosto, por el que se Regula el Catálogo Español de Especies Exóticas Invasoras'. Boletín Oficial del Estado (BOE) 185. <https://www.boe.es/buscar/pdf/2013/BOE-A-2013-8565-consolidado.pdf >.

Rodda, G.H., Fritts, T.H., Clark, C.S., Gotte, S.W. and Chiszar, D., (1999a) 'A State-of-the-Art Trap for the Brown Treesnake'. In: G.H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka (ed.) *Problem Snake Management: The Habu and the Brown Treesnake*, pp. 268–305. Ithaca and London: Cornell University Press.

Rodda, G.H.; Fritts, T.H.; McCoid, M.J. and Campbell, E.W. III. (1999b). 'An Overview of the Biology of the Brown Treesnake, *Boiga irregularis*, a Costly Introduced Pest on Pacific Islands'. In: G.H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka (ed.) *Problem Snake Management: The Habu and the Brown Treesnake*, pp. 44–80. Ithaca and London: Cornell University Press.

Rodda, G.H., Fritts, T.H. and Chiszar, D. (1997). 'The disappearance of Guam's wildlife; new insights for herpetology, evolutionary ecology, and conservation'. BioSience 47: 565–574.

Rodríguez-Pérez, J. (2009). 'Pérdidas de interacciones planta-animal en islas: dispersión de semillas por la lagartija balear y sus consecuencias para la regeneración de un arbusto endémico'. *El Indiferente* 20: 36–43.

Savidge, J.A. (1987). 'Extinction of an island forest avifauna by an introduced snake'. *Ecology* 68: 660–668.

SPE (Servei de Protecció d'Espècies) (2007). *Pla de Recuperació del Ferreret (Alytes muletensis)*. Palma: Conselleria de Medi Ambient. Govern de les Illes Balears.

Silva-Rocha, I., Salvi, D., Sillero, N., Mateo, J.A. and Carretero, M.A. (2015). 'Snakes on the Balearic Islands: An invasion tale with implications for native biodiversity conservation'. *PLOS ONE* 10(4): e0121026. doi:10.1371/journal.pone.0121026

Vigne, D. and Alcover, A. (1985). 'Incidence des relations historiques entre l'homme et l'animal dans la composition actuelle du peuplement amphibien, reptilien et mammalien des îles de méditerranée occidentale'. *Actes 110e Congrès national des Sociétés savantes, Section* Sciences: 79–91.

Wiles, G.J., Bart, J., Beck, Jr., R.E., and Aguon, C.F. (2003). 'Impacts of the brown tree snake: patterns of decline and species persistence in Guam's avifauna' *Conservation Biology* 17: 1350–1360.

Spatial dynamics of invasion and distribution of alien frogs in a biodiversity hotspot archipelago

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Abstract The endemic-rich amphibian fauna of the Philippine Archipelago (ca. 350,000 km^2) includes six alien frogs: the American bullfrog (*Lithobates catesbeianus*), Asiatic painted toad (*Kaloula pulchra*), cane toad (*Rhinella marina*), Chinese bullfrog (*Hoplobatrachus rugulosus*), green paddy frog (*Hylarana erythraea*), and greenhouse frog (*Eleutherodactylus planirostris*). The chronological history of their invasion across the Philippines was reconstructed based on historical and geographic data. Subsequently, we estimated their current and potential distribution through species distribution modelling and Gaussian kernel density smoothing species distribution data. Seven known and potential pathways of introduction into and spread throughout the Philippines were identified, namely, intentional introduction as a (1) biocontrol agent and (2) food source; contamination of (3) agriculture trade, (4) aquaculture trade, and (5) ornamental plant trade; (6) stowaway of cargo; and (7) through the exotic pet trade. Spatio-temporal patterns of distribution showed a stratified diffusion process of spread wherein human-mediated jum dispersal is the primary mode followed by diffusion dispersal. The status of the American bullfrog in the Philippines is unresolved, whether it has successfully established. Meanwhile, the other five alien frogs have established populations in the wild, typically the dominant species in both artificial and disturbed habitats, and are continuously spreading throughout the Philippines. Estimates of current and potential distribution indicate that none of the alien frogs has realised its full potential distribution and that the cane toad is the most widespread, occurring in almost all major islands of the Philippines (ca. 85%), while the greenhouse frog is the least distributed, being found so far in eight provinces and on seven islands. In light of these findings, we provide policy and management recommendations for responding to current and future alien frog invasions.

Keywords: frogs, geographic risk assessment, invasion history, invasive alien species, policy and management

INTRODUCTION

The Philippines (Fig. 1) is the second largest archipelago in the world, with ca. 7,641 islands, and is recognised as a megadiverse nation and a global biodiversity conservation hotspot (Heaney & Mittermeier, 1997; Heaney, et al., 1999; Myers, et al., 2000a). A compelling example of its rich biodiversity is exhibited by the country's amphibian assemblage, which is among the most important faunas in the Indomalayan Region in terms of diversity and endemism (Bain, et al., 2008; Diesmos, et al., 2014). Currently, there are 110 native species of amphibians known from the Philippines, 97 of which (ca. 91%) are endemics (Diesmos, et al., 2015). However, ca. 45% of Philippine amphibians are threatened with extinction: the major threats include habitat loss and deforestation, invasive alien species, emerging infectious diseases, and climate change (Alcala, et al., 2012; Brown, et al., 2012; Diesmos, et al., 2014).

Included in the Philippine amphibian fauna are six introduced frogs, namely, the American bullfrog (*Lithobates catesbeianus* [Shaw, 1802]), the Asiatic painted toad (*Kaloula pulchra* Gray, 1831), the cane toad (*Rhinella marina* [Linnaeus, 1758]), the Chinese bullfrog (*Hoplobatrachus rugulosus* [Wiegmann, 1834]), the green paddy frog (*Hylarana erythraea* [Schlegel, 1837]), and the greenhouse frog (*Eleutherodactylus planirostris* [Cope, 1862]) (Fig. 2; Diesmos, et al., 2006; Diesmos, et al., 2014; Olson, et al., 2014; Diesmos, et al., 2015). Preliminary studies and anecdotal reports indicated that these introduced species, particularly the cane toad and the Chinese bullfrog, are harmful invasives, threatening Philippine wildlife through competitive exclusion and direct predation (Rabor, 1952; Alcala, 1957; Soriano, 1964; Espiritu, 1985; Adraneda, et al., 2005; Diesmos, et al., 2006). Diesmos, et al., (2006) provided the first review on the status and distribution of alien frogs in the Philippines (then only five alien frogs were present). However, there remains a large knowledge gap on their history of invasion and no recent attempts have been made to synthesise the growing body of knowledge on their geographic distribution. By assembling and analysing historical and geographical data of the six alien frogs in the Philippines, we reconstructed the chronological history of invasion and updated their status and distribution. We then estimated their current and potential distribution by projecting suitable areas based on two separate species distribution models ("native range models" and "Philippine models") and, subsequently, Gaussian kernel density smoothing distribution data to delineate occupied suitable areas ("current distribution") and unoccupied suitable areas ("potential distribution").

METHODS

Reconstructing history of invasion

We reconstructed the chronological history of invasion of the six alien frog species in the Philippines based on historical and geographical data ("species distribution data") obtained from the following sources: (1) Natural history collections (NHC): data obtained directly from collections managers of local and international institutions or through the Global Biodiversity Information Facility (GBIF); (2) published and (3) unpublished scientific literature; and (4) personal observations of authors and fellow experts.

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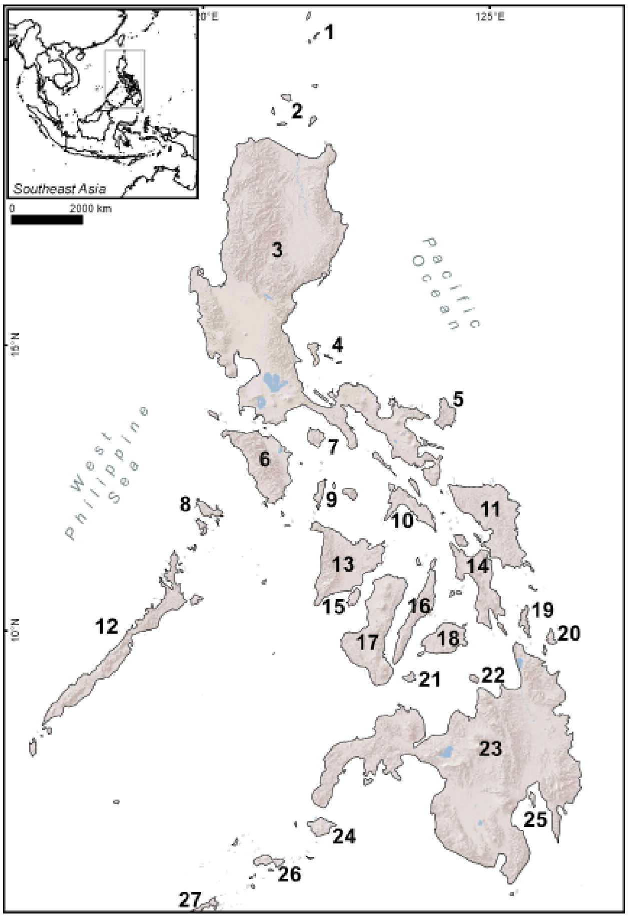

Fig. 1 The Philippine archipelago overlaid on a hypsometric raster shaded-relief. (1) Batanes Island Group, (2) Babyan Island Group, (3) Luzon, (4) Polilio, (5) Catanduanes, (6) Mindoro, (7) Marinduque, (8) Busanga, (9) Romblon Island Group, (10) Masbate, (11) Samar, (12) Palawan, (13) Panay, (14) Leyte, (15) Guimaras, (16) Cebu, (17) Negros, (18) Bohol, (19) Dinagat, (20) Siargao, (21) Siquijor, (22) Camiguin, (23) Mindanao, (24) Basilan, (25) Samal, (26) Jolo, and (27) Tawi-Tawi. Copyright ArcGIS.

Species distribution modelling

Species Distribution Modelling (SDM) involves the quantification of species-environment relationships to define a species' ecological niche. The ecological niche models are then projected into geographic space to visualise and yield an estimate of geographic range or suitable areas where a species can or cannot persist (Guisan & Zimmermann, 2000; Guisan & Thuiller, 2005; Elith & Leathwick, 2009). In studies dealing with invasive alien species, predictions of suitable areas are typically made by extrapolating models fitted with data from the species' native range onto areas that could be invaded (Peterson & Vieglais, 2001; Venette, et al., 2010; Araújo & Peterson, 2012). However, empirical studies have shown evident niche shift in invasive populations, suggesting that species can occupy climatically distinct niche spaces following their introduction into a new area (Broennimann, et al., 2007; Beaumont, et al., 2009).

Here, we developed two separate projections of Philippine-suitable areas for the alien frogs based on models fitted with species distribution data and environmental data (1) from the invaded range in the Philippines (hereafter called "*Philippine models*") and (2) from the alien frogs' native ranges (hereafter called "*Native models*"). Because of the limited amount of species distribution data, we did not develop *Philippine models* for the American bullfrog and the greenhouse frog.

Data collection and calibration. The *Philippine models* were fitted using species distribution data from the Philippines (data used in reconstructing history of invasion). Meanwhile, *Native models* were fitted using species distribution data obtained from the GBIF. Sampling bias was corrected through systematic subsampling neighbouring species distribution data to a resolution of one distribution point per five square kilometres or 2.5 arcminutes and by developing bias files (Elith, et al., 2010; Fourcade, et al., 2014).

The original set of environmental variables includes 19 bioclimatic datasets (Worldclim – Hijmans, et al., 2005) and Global Land Cover 2000 (GLC2000) (Fritz, et al., 2003)

Table 1 Calibration of ecological niche models. Shown are the species distribution data used for model training and testing, model validation approach, number of replicates, and the Maxent features (L – linear; Q – quadratic; P – product) used in fitting *Philippine models* (A) and *Native range models* (B) of the alien frogs. Due to the limited amount of species distribution data viable for model fitting, Philippine models of the American bullfrog and the greenhouse frog were not developed.

Species	Training data	Testing data	Validation	Replication	Maxent Features		
					L	Q	P
A. Philippine model							
American bullfrog	10	-	-	-	-	-	-
Asiatic painted toad	23	-	Crossvalidation	10	✓	✓	-
Cane toad	114	38	Subsampling	10	✓	✓	✓
Chinese bullfrog	79	10	Subsampling	10	✓	✓	✓
Green paddy frog	101	33	Subsampling	10	✓	✓	✓
Greenhouse frog	6	-	-	-	-	-	-
B. Native range model							
American bullfrog	3,704	1,234	Subsampling	10	✓	✓	✓
Asiatic painted toad	93	31	Subsampling	10	✓	✓	✓
Cane toad	1,582	527	Subsampling	10	✓	✓	✓
Chinese bullfrog	83	27	Subsampling	10	✓	✓	✓
Green paddy frog	57	18	Subsampling	10	✓	✓	-
Greenhouse frog	32	-	Crossvalidation	10	✓	✓	-

Fig. 2 Photographs in life of (a) the American bullfrog, (b) the Asiatic painted toad, (c) the cane toad, (d) the Chinese bullfrog, (e) the green paddy frog, and (f) the greenhouse frog. Photographs copyright Tony Gerard (a), Arman N. Pili (b), Emerson Y. Sy (c,d,e,f).

with a spatial resolution of 30 arc seconds. Environmental variables used for fitting the *Philippine models* had a spatial coverage from the Philippines only. Meanwhile, *Native model*s had a spatial coverage equivalent to the native range of the species, based on a convex hull polygon of species distribution data. For both *Philippine* and *Native models* of each species, the environmental variables used for model fitting were pre-selected to only include those that are ecologically relevant (Austin, 2002; Wells, 2007) to the species and are not highly inter-correlated (Dormann, et al., 2013). Correlation between variables were assessed using pair-wise Pearson's correlation coefficient (*stats* R version v.3.3.0 by R Core Team, 2016) and, subsequently, we selected only the putatively ecologically most relevant variable from each group of highly inter-correlated variables ($|r| \geq 0.7$) (Dormann, et al., 2013). The final set of environmental variables used for model fitting included (1) diurnal temperature, (2) temperature seasonality, (3) maximum temperature of warmest month, (4) minimum temperature of coldest month, (5) annual precipitation, (6) precipitation seasonality, (7) precipitation of wettest quarter, and (8) Global Land Cover 2000.

Model fitting. Species distribution modelling was performed using Maximum Entropy Modelling (Maxent v.3.3.3k) (Phillips, et al., 2004; Phillips, et al., 2006a). Maxent is a general-purpose machine learning method premised on the principle of maximum entropy and with a simple and precise mathematical formulation for presence only (i.e., species distribution data) modelling of species distributions from incomplete information (Phillips, et al., 2004; Phillips, et al., 2006a). Maxent has been found to outperform other statistical approaches based on predictive accuracy (Jeschke & Strayer, 2008; Elith & Graham, 2009). Maxent settings used for fitting species distribution models are shown in Table 2. The Maxent features (i.e. linear, quadratic, product) used for each species' models were selected following Phillips (2005), Phillips, et al. (2006b) and Phillips & Dudik (2008) suggestions and were based on the number of species distribution points after systematic subsampling. Developed *bias files* were incorporated in the bias function of Maxent (Table 1). A logistic output was selected to represent the predicted suitable habitats of the species. Pseudo-absence data or background data were generated at random within the Philippines for *Philippine model*s and within the native geographic range of each species for *Native model*s. All other Maxent settings were set to default.

Model evaluation. Model performance of the *Philippine models* of the cane toad, Chinese bullfrog, and green paddy frog, and *Native Models* of all alien frogs except the greenhouse frog was evaluated using the area under receiver operating characteristic (ROC) curve (AUC) by subsampling (randomly splitting presence/pseudo-absences into two subsets with 70% of the records used for model fitting and the remaining 30% to evaluate the models) and was repeated 10 times (Table 1, Pearce & Ferrier, 2000; Allouche et al., 2006; Araújo & Guisan, 2006). Meanwhile, due to the limited amount of species distribution data, model performance of the *Philippine models* of the Asiatic painted toad and *Native Models* of the greenhouse frog was evaluated using the AUC values by 10-fold cross-validation and was repeated five to 10 times (Pearce & Ferrier, 2000; Allouche, et al., 2006; Araújo & Guisan, 2006) (Table 1). The AUC values were interpreted based on Swets (1988) recommendation where 0.5–0.6 = fail, 0.61–0.7 = poor; 0.71–0.8= fair, 0.81–0.9 = good, and 0.91–1.0 = excellent.

Projection. The models were projected to Philippine geographic space to predict suitable areas for the species. The projections were transformed into binary maps of suitable/unsuitable areas, wherein areas above a minimum training presence threshold (no omission) are referred to as "suitable" areas (Liu, et al., 2005).

Table 2 History of invasion and current status and distribution of alien frogs in the Philippines.

Species	Origin of introduced populations	Year and locality of introduction or first detection	Pathway of introduction and spread	Islands Present	Provinces Present
American bullfrog	Louisiana, USA	1966 in Luzon Island	Food source	5	12
Asiatic painted toad	Unknown	2003 in Luzon Island	Cargo Stowaway, Exotic Pet Trade, Ornamental Plant Trade	6	16
Cane toad	Hawaii, USA	1934 in Luzon Island	Biocontrol agent	36	53
Chinese bullfrog	Unknown	1993 in Luzon Island	Food source, Aquaculture trade	7	26
Green paddy frog	Borneo Island[b]	1800s (unknown locality); 1908 Panay Is	Agricultural trade	20	38
Greenhouse frog	Hawaii, USA[b]	2014 in Mindanao Is	Exotic plant trade	8	7

Estimating current and potential distribution

We define the current and potential distribution of invasive alien species as respectively areas occupied and unoccupied by the alien species conditional on areas of suitable habitat (Gormley, et al., 2011). The geographic ranges of the alien frogs in the Philippines were estimated by two-dimensional Gaussian kernel smoothing assembled species distribution data (*kde2d* function of *MASS* v.7.45 R package; Ripley, et al., 2015). This method applies a two-dimensional Gaussian kernel to compute distribution of an animal within its home range/geographic range (Worton, 1989; Venables & Ripley, 2002; Gaston & Fuller, 2009). The solve-the-equation method (*width.SJ* function *MASS* R package; Sheather & Jones, 1991), was used to select the bandwidth for kernel smoothing, and was defined to include 99.5% of species' distribution data. Estimated geographic ranges were then used to delineate the occupied suitable areas ("current distribution") and unoccupied suitable areas ("potential distribution"). Because of the limited amount of species distribution data, we did not estimate the geographic range of the American bullfrog and the greenhouse frog in the Philippines, and, consequently, we did not delineate their current and potential distribution.

RESULTS

History of invasion

A comprehensive review of the history of invasion, including an assembled species distribution database, of the six alien frogs in the Philippines is prepared in a separate study for future publication. The review provided below will suffice as a general overview of their history of invasion.

The American bullfrog

Individuals of the American bullfrog were imported from Louisiana, United States in 1966 and were first reared on Luzon Island (Ugale, 1976; Pascual, 1987b). Frogs were initially bred for the export production of scientific specimens for biomolecular and medical research and other educational activities (Pascual, 1987a; Urbanes, 1988; Urbanes, 1990; Matienzo, 1990). Subsequently, in 1980, through government efforts to boost food security, the American bullfrog breeding shifted to food production. Another eight American bullfrog breeding centres were established across the Philippines (Table 2; Fig 3a; Ministry of Natural Resources, 1981; Buenviaje, 1983; Inovejas, 1985). Breeding centres ceased operation in 1985. The current status of the American bullfrog in the Philippines, whether they were able to successfully establish populations in the wild, is unknown.

The Asiatic painted toad

The Asiatic painted toad was first reported in the Philippines in 2003 on Luzon Island (Diesmos, et al., 2006). It was earlier suggested that the initial introduction of the Asiatic painted toad was through the exotic pet trade (Diesmos, et al., 2006). Introduction as a contaminant of ornamental plant trade or as cargo stowaway is also plausible. From localities of its initial introduction, the Asiatic painted toad has spread in all directions throughout the Philippines and is now recorded in 16 provinces on six islands (Table 2; Fig. 3b). It is likely that the identified introduction pathways may have mediated its spread throughout the Philippines.

The cane toad

The cane toad was intentionally introduced in the Philippines as a part of a national pest control programme (Merino, 1936). Cane toads were secured from the Hawaiian Sugar Planters Association and were brought to the Philippines in 1934 (Merino, 1936). The toads were initially reared on Luzon Island. Since then, they have spread in all directions across islands and onto different islands throughout the Philippines. Their spread is primarily mediated by human movement (deliberate release for biocontrol), as a cargo stowaway, and neighbourhood diffusion dispersal (Rabor, 1952). Today, the cane toad can

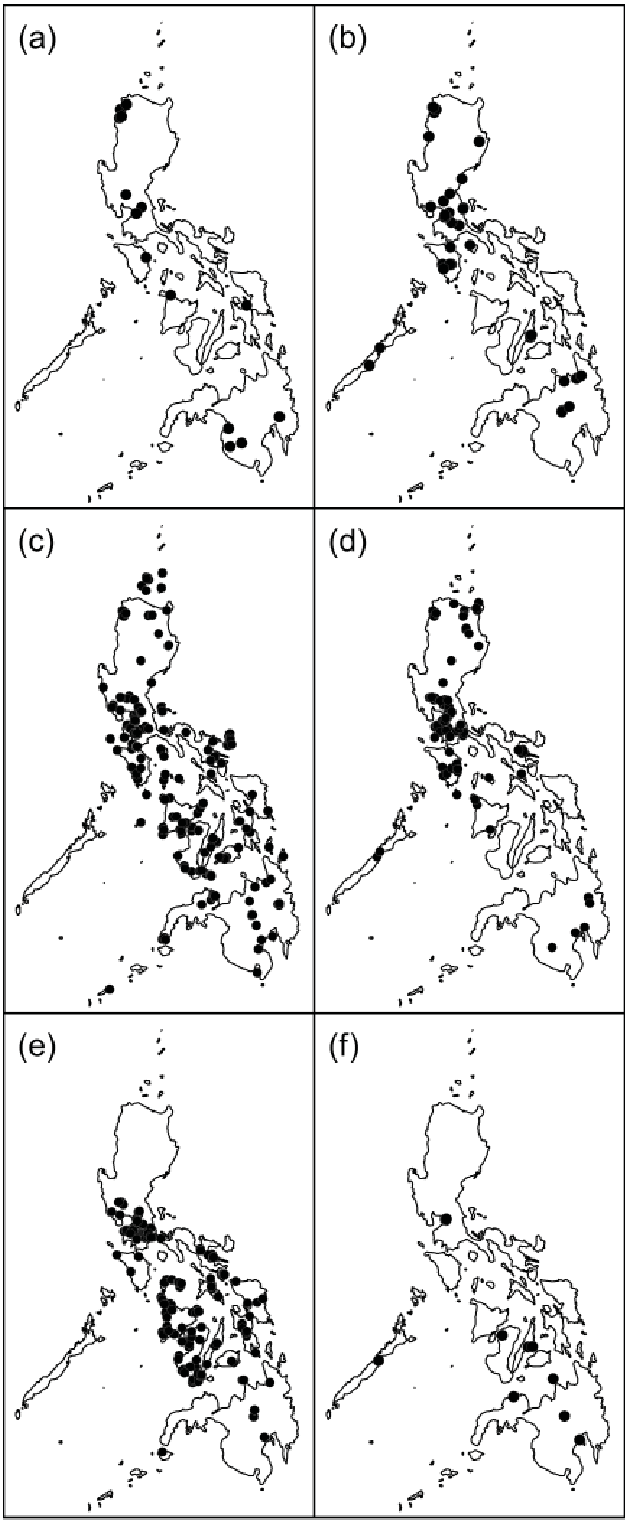

Fig. 3 Geographic distribution of the alien frogs in the Philippines, (a) American bullfrog, (b) Asiatic painted toad, (c) cane toad, (d) Chinese bullfrog, (e) green paddy frog, and (f) greenhouse frog. Points indicate areas where alien frogs were reported present (release sites or areas where bullfrog breeding centres were formerly established for the case of the American bullfrog).

be found on almost all major islands of the Philippines, where it is usually the dominant amphibian species in invaded areas (Table 2; Fig. 3c; Alcala, 1986; Alcala & Brown, 1998; Diesmos, et al., 2006; Diesmos, et al., 2015).

The Chinese bullfrog

The Chinese bullfrog was first reported in the Philippines in 1993 on Luzon Island (Diesmos, 1998; Diesmos et al., 2006). It was speculated that this species was introduced into and spread throughout the Philippines along with American bullfrog breeding in the 1980s (Diesmos, et al., 2006). Other potential pathways of introduction and spread of the Chinese bullfrog throughout the Philippines are contamination of agricultural trade, as for the case of co-occurring alien and native frogs in the Philippines (Inger, 1954; Kuraishi, et al., 2009), and contamination of aquaculture trade, as was the case of its congeneric (Indian bullfrog *Hoplobatrachus tigerinus*) on Andaman Islands, India (Surendran & Vasudevan, 2013). The Chinese bullfrog is now found in 26 provinces on seven islands in the Philippines (Table 2; Fig. 3d).

The green paddy frog

The earliest valid records of the green paddy frog, overlooked in previous discussions regarding its history of invasion (e.g., Inger, 1954), were collections from Panay Island in 1908 (Orrell & Hollowell, 2017). In the early 1900s, the green paddy frog was initially thought to be native to the Philippines with restricted distribution on the islands of Negros, Panay, Sibuyan and Tablas (Taylor, 1920; Taylor, 1922; Inger, 1954). Inger (1954) suggested that the green paddy frog was introduced as a contaminant of agricultural trade owing to its disjunct distribution from the nearest extra-Philippine populations on Borneo Island. The green paddy frog is now found in 38 provinces on 20 islands (Table 2; Fig. 3e). Contamination of agricultural and aquaculture trade may be implicated for its spread throughout the Philippines.

The greenhouse frog

The greenhouse frog was first detected on Mindanao Island in 2013 (Olson, et al., 2014). the propensity of the greenhouse frog to thrive in human-modified environments, especially in gardens (Olson et al., 2014; Sy & Salgo, 2015; Sy, et al., 2015a, b; Sy, 2017a,b), suggests that the trade in exotic ornamental plants is the most plausible pathway of its introduction into and spread throughout the Philippines, as was documented in Hawaii (Kraus, et al., 1999). The greenhouse frog has so far been recorded in eight provinces on seven islands (Table 2; Fig. 3f).

Philippine-suitable areas

Models of the alien frogs indicate fair to excellent training-AUC values (≥70) (Table 3). Based on projections of Philippine-suitable areas of both the *Philippine models* (except American bullfrog and greenhouse frog) and *Native models*, the alien frogs are, to varying extents, suitable to the Philippines. It should be noted that the *Native models* consistently projected a broader range of Philippine-suitable areas (Figs 4 & 5; Table 4). Moreover, both the *Philippine* and *Native models* consistently projected human-modified and disturbed areas to exhibit typical to high probability of suitable conditions for these alien species.

Current and potential distribution

Maps show that the Asiatic painted toad has occupied ca. 30–40% of projected suitable areas (or ca. 20–30% of total Philippine land area), particularly most of central and northern Luzon Island, north-western islands of central Philippines (Cebu, Marinduque, Mindoro, and Palawan Islands), and central Mindanao Island. Potential distribution of the Asiatic painted toad includes islands north of Luzon Island (Babuyanes and Batanes group of islands) areas in north-central (Cordillera Administrative Region), southern (Bicol Region) and most of central Luzon Island, western Mindanao Island, and Sulu Archipelago (Table 5; Fig. 6). The cane toad has occupied almost all projected suitable areas (ca.98–100%) except those on the islands of Batanes Province (northernmost group of islands of the Philippines), islands of Palawan Province (westernmost group of islands), and most of Sulu Archipelago (southernmost islands) (Table 5; Fig. 6). Maps showed that Chinese bullfrog has a disjunct distribution throughout the Philippines, having occupied ca. 40–50% of suitable areas (or ca. 35–40% of total Philippine land area), specifically most of Luzon Island, islands of central

Table 3 Evaluation of the prediction of Species Distribution Models of the six alien frogs. *Philippine models* (**A**) and *Native range models* (**B**) were evaluated by validation of predictions based on the Area Under the Receiver Operating Characteristic (ROC) Curve (AUC).

Species	Training AUC (mean)	Test AUC (mean)	AUC values interpretation	Minimum training presence threshold
A. Philippine model				
American bullfrog	0.78	-	Fair	0.2886
Asiatic painted toad	0.86	-	Good	0.2663
Cane toad	0.72	0.71	Fair	0.0922
Chinese bullfrog	0.84	0.82	Good	0.0858
Green paddy frog	0.82	0.79	Fair	0.1286
Greenhouse frog	0.97	-	Excellent	0.1527
B. Native range model				
American bullfrog	0.70	0.70	Fair	0.0378
Asiatic painted toad	0.86	0.81	Good	0.1243
Cane toad	0.76	0.75	Fair	0.1091
Chinese bullfrog	0.85	0.83	Good	0.0206
Green paddy frog	0.74	0.69	Fair	0.1225
Greenhouse frog	0.78	-	Fair	0.1823

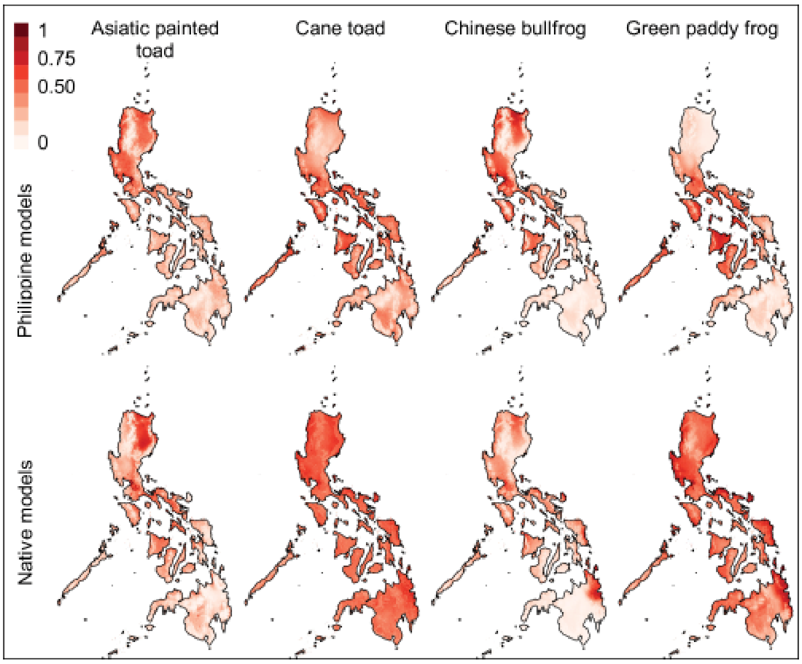

Fig. 4 Projected Philippine-suitable areas for (from left to right) Asiatic painted toad, cane toad, Chinese bullfrog, and green paddy frog, based on (top row) *Philippine model* and (bottom row) *Native model*.

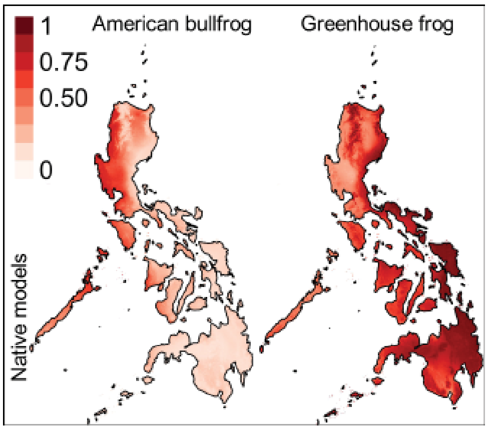

Fig. 5 Projected Philippine-suitable areas for (from left to right) the American bullfrog and the greenhouse frog based on *Native models*.

Philippines (Mindoro and Panay Island), and central and eastern Mindanao Island. Most of its potential distribution are the islands north of Luzon Island (Babuyanes and Batanes group of islands), some areas in central Philippines (Central Visayas Region and Eastern Visayas Region), and most of Mindanao Island including Sulu Archipelago (Table 5; Fig. 6). Despite being present in the Philippines for more than a century, the green paddy frog has only invaded ca. 40–60% of projected suitable areas or 30–40% of the Philippines. The current distribution of the green paddy frog is mainly in Central Philippines, southern and central parts of Luzon Island, disjunct areas in Mindanao Island, and Basilan Island. Potential distribution of the green paddy frog includes most of the islands of Palawan Province, Mindanao Island, and central to northern Luzon Island (Table 5; Fig. 6). Lastly, due to the limited amount of species distribution data, the current and potential distribution of the American bullfrog and the greenhouse frog were not estimated. Interestingly, projections show that almost all of the Philippines is suitable for both species (Fig. 7).

Collectively, maps showed that none of the alien frogs has fully occupied all projected Philippine-suitable areas, and that all alien frogs are on Luzon Island and Mindanao Island, the two largest islands of the Philippines. The islands of Batanes Province (Northernmost group of islands of the Philippines) are the only remaining places in the Philippines with no record of alien frogs.

DISCUSSION

We first discuss here how our study filled knowledge gaps on the invasion history and the status of the alien frogs in the Philippines. At the end of this section, we provide policy and management recommendations.

Invasion history: conceptual background

Our study is the first to reconstruct the invasion history of the six alien frogs in the Philippines. History of invasion refers to the historical, demographical and geographical features of a species' invasion processes. This may include information on the source of propagules and propagule pressure, the dispersal pathways and associated vectors, and the geographical and demographical dynamics of the spread of the adventive populations (Dlugosch & Parker, 2008; Estoup & Guillemaud, 2010). Knowledge of the invasion history forms the foundation of invasion biology by addressing practical and theoretical questions as well as testing different hypotheses concerning the ecology and evolution underlying biological invasions (Estoup & Guillemaud, 2010). More importantly, elucidating invasion history can provide invaluable insights for the

Table 4 Estimates of suitable area in the Philippines (PH) for the six alien frogs. Total area (km²) and percentage (%) of total Philippine land area that is suitable (above minimum training presence threshold) to the alien frogs.

Species	Philippine model		Native range model	
	km²	(%) of total PH	km²	(%) of total PH
American bullfrog	-	-	349,107	99.64
Asiatic painted toad	193,964	55.36	280,825	80.15
Cane toad	344,317	98.27	349,106	99.64
Chinese bullfrog	272,797	77.86	325,179	92.81
Green paddy frog	237,825	67.88	349,104	99.64
Greenhouse frog	-	-	349,107	99.64

Table 5 Estimate of suitable area (current distribution) in the Philippines (PH) occupied by the Asiatic painted toad, the cane toad, the Chinese bullfrog, and the green paddy frog. Shown are total area (km2), percentage (%) of total suitable (Minimum Training Threshold) area, and percentage (%) of PH total land area that is occupied by the alien frogs.

Species	Philippine model			Native range model		
	Occupied suitable area (km^2)	(%) of total suitable	(%) total PH	Occupied suitable area (km^2)	(%) of total suitable	(%) total PH
Asiatic painted toad	81,262	41.90	23.19	106,894	38.064	30.51
Cane toad	293,061	85.11	83.64	296,938	85.057	84.75
Chinese bullfrog	127,185	46.62	36.30	141,797	43.606	40.47
Green paddy frog	141,927	59.68	40.51	160,700	46.032	45.86

development and implementation of sound strategies and science-based policies for the management of invasive alien species, particularly in preventing future introductions and controlling incursions (Hulme, et al., 2008; Estoup & Guillemaud, 2010; Kulhanek, et al., 2011). Here, we reconstructed the chronological history of invasion, identified known and potential pathways involved in introduction, and updated the current status and distribution of invasive frog species in the Philippines. Below we discuss further the dynamics and mechanisms underlying their spread based on spatio-temporal patterns of species distribution.

Invasion history: pathways of introduction of the alien frogs

Identifying the geographical origin, causative pathways, and associated vectors of past introductions can help guide the development of preventive measures, such as monitoring and quarantine schemes, which are most effective when specifically targeted to ports of entry and trade of commodities associated with identified pathways of introduction (Hulme, 2006; Hulme, 2009; Hulme, et al., 2008). Six principal pathways are involved in the global movement of species into new areas: alien species may be commodities (intentionally released and escapees), contaminants of commodities, stowaways on vectors, opportunists exploiting corridors resulting from transport infrastructures, or they may spread naturally (Hulme, et al., 2008). It is noteworthy that the number of total introductions through each pathway may vary among taxonomic groups. For instance, global alien amphibian introductions are most frequently through intentional release as biocontrol agent and food source, contaminant of ornamental plant trade, stowaway of cargo, and escapees from exotic pet trade (Kraus, 2009). In addition to these, herein we identified two other pathways by which alien frogs were introduced into the Philippines: as a contaminant of agricultural trade and aquaculture trade.

Invasion history: dynamics and mechanisms of spread of the alien frogs

Understanding the pattern and rate of spread of invasions are essential components of risk assessment of invasive alien species (Stohlgren & Schnase, 2006;

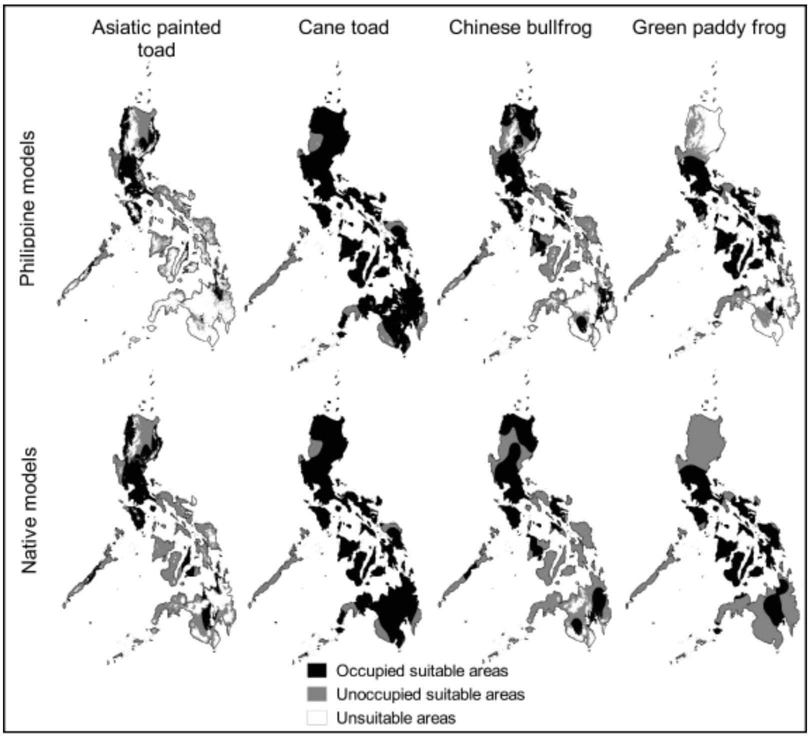

Fig. 6 Current and potential distribution in the Philippines of (from left to right) the Asiatic painted toad, cane toad, Chinese bullfrog, and the green paddy frog based on estimates of geographic range and Philippine-suitable areas projected by (top row) *Philippine models* and (bottom row) *Native models*.

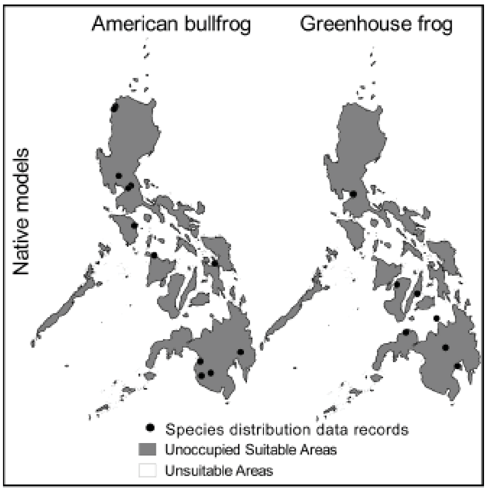

Fig. 7 Current and potential distribution in the Philippines of (from left to right) the American bullfrog and the greenhouse frog based on overlaid species distribution data over Philippine-suitable areas projected by *Native models*. Points indicate areas where alien frogs were reported present (release sites or areas where bullfrog breeding centres were formerly established for the case of the American bullfrog).

Stohlgren & Jarnevich, 2009). For invading organisms, a stratified diffusion process of spread "seems to be the rule rather than the exception" (Higgins & Richardson, 1999). In a stratified diffusion process, initial range expansion occurs though neighbourhood diffusion and new colonies are successively created through jump dispersal events by long-distance migrants, accelerating the rate of overall invasion (Van der Plank, 1967 as cited in Hengeveld, 1989; Shigesada, et al., 1995; Higgins & Richardson, 1999). Jump dispersal events are particularly common for species that are closely associated with humans (Suarez, et al., 2001). For instance, human-mediated jum dispersal has been documented in *Eleutherodactylus* spp. in Hawaii (Kraus & Campbell, 2002), Argentine ants (*Linepithema humile*) in the United States (Suarez, et al., 2001), common ragweed (*Ambrosia artemisiifolia*) in France (Chauvel, et al., 2006).

The reconstructed history of invasion showed that the spread of alien frogs in the Philippines followed a stratified diffusion process wherein human-mediated jump dispersal and neighbourhood diffusion dispersal were the main modes of spread. Given the innate physiological limitations of frogs to cross marine barriers and the close affinity of alien frogs with humans (Wells, 2007), human-mediated jump dispersal is the most plausible primary mode of dispersal of alien frogs inter- and intra-island. Numerous jump dispersal events throughout the course of the invasion of alien frogs in the Philippines can be observed in the spatio-temporal distribution patterns shown in the generated species' distribution maps (Fig. 3). This is particularly evident in the invasion of the cane toad, wherein from founder populations on five islands, it has invaded almost all major islands in the Philippines in a matter of decades (Rabor, 1952). The dispersal of the green paddy frog to Basilan Island, some 300 km from the nearest introduced population in Negros Island in the 1960s and 350 km from nearest native population on Borneo Island, demonstrates a good example of either long-distance jump dispersal or perhaps a secondary introduction event. For the cases of the Asiatic painted toad, the Chinese bullfrog, and the greenhouse frog, it is unclear whether their presence on different islands is caused by jump dispersal events from a single founder population or the result of multiple, independent introduction events.

The same pathways implicated for alien frog introductions may have served as the same pathways that mediated their jump-dispersal throughout the Philippines. Spread of the cane toad was primarily human-mediated, being released deliberately by both government and private individuals with the belief that the frogs would control insect and rodent pests in agricultural fields (Merino, 1936; Rabor, 1952; Soriano, 1964). Observations in the Philippines and on Borneo reported cane toads and the Asiatic painted toads in cargo and vehicles of transport and trade as stowaways (Inger, 1966; A.C. Diesmos personal observation). The greenhouse frog may have spread throughout the Philippines as a contaminant of ornamental plant trade and nursery plants, as happened in Hawaii (Kraus, et al., 1999; Olson, et al., 2012). Similarly, the propensity of the Asiatic painted toad to seek refuge in greenhouse materials (e.g. potted plants, soil, etc.) implicate ornamental plant trade and movement of nursery plants as a potential pathway for its spread (E.Y. Sy personal observation). The American bullfrog, despite having an unresolved status in the Philippines, was dispersed throughout the Philippines as a food source. It was earlier speculated that the Chinese bullfrog may have been introduced and spread throughout the Philippines alongside the proliferation of American bullfrog breeding centres in the 1980s (Diesmos, et al., 2006). Moreover, agricultural trade and aquaculture trade may have served as dispersal pathways for the Chinese bullfrog and as well as the green paddy frog. Agricultural trade has been attributed to recent range expansion of some Philippine native species such as the Philippine common tree frog (*Polpedates leucomystax*), the common mud frog (*Occidozyga laevis*), and the Philippine paddy frog (*Fejervarya vittigera*) (Inger, 1954; Brown, et al., 2010). Meanwhile, the aquaculture trade served as a minor pathway of global introduction for alien frogs and has been well documented in some alien frogs on Guam (Christy, et al., 2007; Kraus, 2007; Kraus, 2009).

Neighbourhood diffusion dispersal also played an invaluable role in the spread of alien frogs within islands. For instance, it was observed that the cane toad has diffused up to 20 km around Dumaguete City, Negros Island in a matter of 15 years (Rabor, 1952). Though this observation was not supported by empirical data, in Australia, the cane toad was observed to travel up to 1.8 km per night, especially during the rainy months (Phillips, et al., 2006b). Moreover, short-distance dispersal may be aided by other "natural" processes such as extensive floods, which are common in most parts of the Philippines.

Policy and management recommendations

Given the potential negative ecological and economic implications of alien frogs (Kraus, 2015), policies and management strategies for alien frog invasions in the Philippines are urgently needed. Our study filled knowledge gaps on the invasion of the alien frogs in the Philippines, which can guide the development and implementation of sound policies and management strategies, particularly the Philippines' National Invasive Species Strategy and Action Plan (NISSAP; DENR-PAWB, 2013).

Prevention of future alien introductions.

Of the six alien frogs currently occurring in the Philippines, three were introduced only in the past three decades, with the greenhouse frog being the most recently reported. Given the lack of measures to prevent invasions in the Philippines, future alien frog introductions seem inevitable. In fact, a recent survey conducted by the authors reported a seventh alien frog is now present (A.C. Diesmos, *for future publication*). A useful preventive measure are early-warning systems (i.e., black-white lists, watch lists, etc.). These systems direct border preventive measures, such as inspection, quarantine, and policies banning entry, by identifying alien species with the potential to threaten native biodiversity (Heger & Trepl, 2003; Hulme, 2006; Maynard & Nowell, 2009). A separate study conducted by the authors for future publication identified alien amphibians that can potentially threaten Philippine biodiversity based on three factors of invasion success, namely history of invasion elsewhere, climate match, and propagule pressure.

To prevent future alien frog introductions, preventive measures are best focused on potential pathways and associated vectors (Perrings, et al., 2005; Hulme, 2006; Hulme, 2009; Hulme, et al., 2008). Some examples of preventive measures include (1) prohibition or developing stricter regulations and standards for the breeding, trading, and keeping of exotic pets (e.g., Taiwan, Australia, and New Zealand) and animals for food production (e.g., European Union States), (2) post-border inspection, quarantine, and treatment of imported commodities such as ornamental plants (e.g., Hawaii and Guam), fish fingerlings, and agricultural products, standardising risk assessment of candidate biocontrol species, and (3) early detection and rapid eradication schemes at ports of entry such as seaports and airports (reviewed in Hulme, 2009; preventive measures focusing on alien amphibians and reptiles are reviewed in Kraus, 2009).

Management of spread between islands

Developing measures to control the inter-island spread of alien frogs is critical in archipelagic systems, such as the Philippines. Like prevention, measures to control the inter-island spread of alien species are best focused on the identified potential pathways of spread and their associated vectors (Hulme, 2006; Hulme, 2009; Hulme, et al., 2008). Some examples of control measures include: for the American bullfrog, Asiatic painted toad, and the Chinese bullfrog, prohibition of release and implementation of standards and regulations for possession or breeding either as pets or for farming (although no bullfrog breeding centres are operational to date); for the Asiatic painted toad and greenhouse frog, quarantine, inspection, and treatment of traded and transported ornamental plants, nursery plants, and greenhouse material; for the Asiatic painted toad and cane toad, early detection and rapid eradication schemes on ports of entry such as seaports and airports and inspection of cargo; for the Chinese bullfrog and green paddy frog, inspection of products of agricultural trade and prohibition of fish fingerling collection for release in novel areas. These control measures should be focused on unoccupied but suitable islands (Leung, et al., 2005), especially in the Batanes Island Group.

It is noteworthy that the Philippines has perhaps the moral responsibility to contain these exotics from spreading into neighboring foreign areas. For example, the southernmost extent of invasion of the cane toad in the Philippines is on Basilan Island, which is about 100 kilometers from Borneo Island (Malaysia) and where the species is alien. The spread of the alien frogs to foreign countries can be prevented by inspection of commodities for export, especially those associated with pathways of introduction and spread. In fact, the Philippine common treefrog was introduced into Ryukyu Archipelago, Japan by contaminated traded agricultural commodities (Kuraishi, et al., 2009).

Maps of current and potential distribution as a guide for management schemes.

Estimating and delineating the potential and current geographical range of alien species is a critical component of risk assessment by providing science-based information that can help guide the strategic allocation of limited resources for the management of invasive alien species (Stohlgren & Schnase, 2006; Stohlgren & Jarnevich, 2009; Venette, et al., 2010). For instance, surveys and monitoring schemes should be focused in areas with no information on the status of the alien frogs or areas of high conservation concern (Wittenberg & Cock, 2001; McGeoch & Squires, 2015), such as in most Protected Areas in the Philippines, on central Luzon Island (Cordillera Administrative Region), western Mindanao Island, and islands in the Batanes Province. More importantly, field surveys are warranted in areas where bullfrog breeding centers were formerly established as well as in release sites, so to confirm the status of the American bullfrog in the Philippines (Fig. 3a). Control and containment of incursions and mitigation of impacts should be focused on invaded areas, especially in areas of high conservation value such as protected areas and nature reserves (Myers, et al., 2000b; Wittenberg & Cock, 2001; Parrish, et al., 2003). Early detection and rapid eradication schemes are best focused on the interface between the potential and current distribution (Hulme, 2006), such as the invasion front of the green paddy frog on central Luzon Island (Fig. 3e).

Recognizing the variability in projected suitable areas between the *Philippine* and *Native models*, and that different modelling techniques yield different results even if calibrated with same set of data, we developed in a separate study for future publication projections of suitable areas based on ensembles of models fitted with data from the entire range (native range and all invaded range/s) and using different statistical techniques. Moreover, evaluation of the accuracy of projections and estimates through ground truthing are underway.

Recommendations for future research

The following recommendations for future research on alien amphibian invasions in the Philippines are suggested: data mining grey literature, conducting interviews, and targeted field work to populate the assembled species distribution database and improve reconstructed invasion history; vector analysis of the pathways so as to understand their importance to current and future alien amphibian invasions; identify 'native exotics' and understand their invasion histories (e.g. dynamics and mechanisms involved in their spatial spread) and impact to ecosystems; comprehensive risk analysis of the alien frogs, specifically research on their ecological and socio-economic impacts; test different hypothesis on the evolution and ecology of alien species invasions.

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REFERENCES

Adraneda, C., Malit, I., Ortiz-Luis, G., Pagaduan, C. and Perez. J. (2005). *Initial Study on the Dietary Preference of Hoplobatrachus rugulosus (Wiegmann 1834)*. BSc thesis. Manila, Philippines: Department of Biological Sciences. University of Santo Tomas.

Alcala, A.C. (1957). 'Philippine notes on the ecology of the giant marine toad'. *Silliman Journal* 6(2): 90–96.

Alcala, A.C. (1986). *Guide to Philippine Flora and Fauna. Vol X: Amphibians and Reptiles*. Quezon City, Philippines: Natural Resources Management Center – Ministry of Natural Resources; Quezon City, Philippines: University of the Philippines.

Alcala, A.C. and Brown, W.C. (1998). *Philippine Amphibians. An Illustrated Field Guide*. Makati City, Philippines: Bookmark, Inc.

Alcala, A.C., Bucol, A.A., Diesmos, A.C. and Brown, R.M. (2012). 'Vulnerability of Philippine amphibians to climate change'. *Philippine Journal of Science* 141(1): 77–87.

Allouche, O., Tsoar, A. and Kadmon, R. (2006). 'Assessing the accuracy of species distribution models: prevalence, kappa and the true skill statistics (TSS)'. *Journal of Applied Ecology* 43: 1223–1232.

Araújo, M.B. and Guisan, A. (2006). 'Five (or so) challenges for species distribution modelling'. *Journal of Biogeography* 33: 1677–1688.

Araújo, M.B. and Peterson, A.T. (2012). 'Uses and misuses of bioclimatic envelope modeling'. *Ecology* 93(7): 1527–1539.

Austin, M.P. (2002). 'Spatial prediction of species distribution: an interface between ecological theory and statistical modelling'. *Ecological Modelling* 157: 101–118.

Bain, R., Biju, S.D., Brown, R., Das, I., Diesmos, A., Dutta, S., Gower, D., Inger, R., Iskandar, D., Kaneko, Y., Lau, M.W.N., Meegaskumbura, M., Ohler, A., Papenfuss, T., Pethiyagoda, R., Stuart, B., Wilkinson, M. and Xie, F. (2008). 'Amphibians of the Indomalayan Realm'. In: S.N. Stuart, M. Hoffmann, J.S. Chanson, N.A. Cox, R. Berridge, P. Ramani and B.E. Young (eds.) *Threatened Amphibians of the World*, pp. 74–49. Barcelona, Spain: Lynx Ediciones; Gland, Switzerland: IUCN; Arlington Virginia, United States: Conservation International.

Beaumont, L.J., Gallagher, R.V., Thuiller, W., Downey, P.O., Leishman, M.R. and Hughes, L. (2009). 'Different climatic envelopes among invasive populations may lead to underestimations of current and future biological invasions'. *Diversity and Distributions* 15(3): 409–420.

Broennimann, O., Treier, U. A., Müller–Shärer, H., Thuiller, W., Peterson, A. T. and Guisan, A. (2007). 'Evidence of climate niche shift during biological invasion'. *Ecology Letter* 10: 701–709.

Brown, R.M., Linkem, C.W., Siler, C.D., Sukumaran, J., Esselstyn, J.A., Diesmos, A.C., Iskandar, D.T., Bickford, D., Evans, B.J., McGuire, J.A. and Grismer, L. (2010). 'Phylogeography and historical demography of *Polypedates leucomystax* in the islands of Indonesia and the Philippines: Evidence for recent human-mediated range expansion?'. *Molecular Phylogenetics and Evolution* 57(2): 598–619.

Brown, R.M., Diesmos, A.C., Sanguila, M.B., Siler, C.D., Diesmos, M.L.D. and Alcala, A.C. (2012). 'Amphibian conservation in the Philippines'. *FrogLog* 20(5), Number 104: 40–43.

Buenviaje, S.V. (1983). 'Bullfrog culture, anyone?'. *Aquascope* 3(4): 4.

Chauvel, B., Dessaint, F., Cardinal-Legrand, C. and Bretagnolle, F. (2006). 'The historical spread of *Ambrosia artemisiifolia* L. in France from herbarium records'. *Journal of Biogeography* 33: 665–673.

Christy, M. T., Savidge, J. A. and Rodda, G. H. (2007). 'Multiple pathways for invasion of anurans on a Pacific island'. *Diversity and Distributions* 13: 598–607.

Department of Environment and Natural Resources – Protected Areas and Wildlife Bureau (DENR–PAWB). (2013). *The National Invasive Species Strategy and Action Plan*. Quezon City, Philippines: DENR–PAWB.

Diesmos, A. C. (1998). *The amphibian faunas of Mount Banahao, Mount Cristobal, and Mount Maquiling, Luzon Island, Philippines.* MSc thesis. Laguna, Philippines: University of the Philippines–Los Baños.

Diesmos, A.C., Diesmos, M.L. and Brown, R.M. (2006). 'Status and distribution of alien invasive frogs in the Philippines'. *Journal of Environmental Science and Management* 9(2): 41–53.

Diesmos, A.C., Alcala, A.C., Siler, C.D. and Brown, R.M. (2014). 'Status and Conservation of Philippine Amphibians'. In: H. Heatwole and I. Das (eds.) *Conservation Biology of Amphibians of Asia – Status of Conservation and Decline of Amphibians: Eastern Hemisphere*, pp. 310–336. Kota Kinabalu, Malaysia: Natural History Publications (Borneo).

Diesmos, A.C., Watters, J.L., Huron, N.A., Davis, D.R., Alcala, A.C., Crombie, R.I., Afuang, L.E., Gee-Das, G., Sison, R.V., Sanguila, M. B., Penrod, M. L., Labonte, M.J., Davey, C.S., Leone, E.A., Diesmos, M. L., Sy, E. Y., Welton, L. J., Brown, R. F. and Siler, C.D. (2015). 'Amphibians of the Philippines, part I: Checklist of the species'. *Proceedings of the California Academy of Science* 62(3) Number 20: 457–539.

Dlugosch, K.M. and Parker, I.M. (2008). 'Founding events in species invasions: Genetic variation, adaptive evolution, and the role of multiple introductions'. *Molecular Ecology* 17: 431–449.

Dormann, C.F., Elith, J., Bacher, S., Buchmann, C., Carl, G., Carré, G., Marquéz, J.R.G., Gruber, B., Lafourcade, B., Leitão, P.J., Münkemüller, T., McClean, C., Osborne, P. E., Reineking, B, Schröder, B., Skidmore, A. K., Zurell, D. and Lautenbach, S. (2013). 'Collinearity: A review of methods to deal with it and a simulation study evaluating their performance'. *Ecography* 36(1): 27–46.

Elith, J. and Graham, C.H. (2009). 'Do they? How do they? Why do they differ? On finding reasons for differing performances of species distribution models'. *Ecography* 32: 66–77.

Elith, J. and Leathwick, J.R. (2009). 'Species distribution models: ecological explanation and prediction across space and time'. *Annual Review of Ecology, Evolution, and Systematics* 40: 677–697.

Elith, J., Kearney, M. and Phillips, S. (2010). 'The art of modelling range-shifting species'. *Methods in Ecology and Evolution* 1: 330–342.

Espiritu, L.M. (1985). *Population Ecology and Distribution of Bufo marinus L. in the Province of Isabela, Philippines*. MSc thesis. Los Baños, Laguna: University of the Philippines–Los Baños.

Estoup, A. and Guillemaud, T. (2010). 'Reconstructing routes of invasion using genetic data: Why, how and so what?'. *Molecular Ecology* 19: 4113–4130.

Fourcade, Y., Engler, J. O., Rödder, D. and Secondi, J. (2014). 'Mapping species distributions with MAXENT using a geographically biased sample of presence data: a performance assessment of methods for correcting sampling bias'. *PLOS ONE* 9(5): e97122.

Fritz, S., Bartholome, E., Belward, A., Hartley, A., Stibig, H.J., Eva, H., Mayaux, P., Bartalev, S., Latifovic, R., Kolmert, S. and Roy, P.S. (2003). *Harmonisation, mosaicing and production of the Global Land Cover 2000 database (Beta Version)*. Ispra, Italy: Joint Research Center (JRC).

Gaston, K.J. and Fuller, R.A. (2009). 'The sizes of species' geographic ranges'. *Journal of Applied Ecology* 46(1): 1–9.

Gormley, A.M., Forsyth, D.M., Griffioen, P., Lindeman, M., Ramsey, D.S., Scroggle, M.P. and Woodford, L. (2011). 'Using presence-only and presence-absence data to estimate the current and potential distributions of established invasive species'. *Journal of Applied Ecology* 48: 25–34.

Guisan, A. and Zimmermann, N.E. (2000). 'Predictive habitat distribution models in ecology'. *Ecological Modelling* 135: 147–186.

Guisan, A. and Thuiller, W. (2005). 'Predicting species distribution: Offering more than simple habitat models'. *Ecology Letters* 8: 993–1009.

Heaney, L.R. and Mittermeier, R.A. (1997). 'Philippines'. In: R. A. Mittermeier, G. Robles, and C. G. Mittermeier (eds.) *Megadiversity. Earth's Biologically Wealthiest Nations,* pp. 236–255. Monterey, Mexico: CEMEX.

Heaney, L.R., Mittermeier, R.A. and Mittermeier, C.G. (1999). 'Philippines'. In: R.A. Mittermeier, N. Myers, G. Robles and C.G. Mittermeier (eds.) *Hotspots. Earth's Biologically Richest and Most Endangered Terrestrial Ecosystems*, pp. 308–317. Monterrey, Mexico: CEMEX.

Heger, T. and Trepl, L. (2003). 'Predicting biological invasions'. *Biological Invasions* 5: 313–321.

Hengeveld, R. (1989). *Dynamics of Biological Invasions*. Springer Science & Business Media.

Higgins, S.I. and Richardson, D.M. (1999). 'Predicting plant migration rates in a changing world: The role of long-distance dispersal'. *The American Naturalist* 153(5): 464–475.

Hijmans, R.J., Cameron, S.E., Parra, J.L., Jones, P.G. and Jarvis, A. (2005). 'Very high resolution interpolated climate surfaces for global land areas'. *International Journal of Climatology* 25: 1965–1978.

Hulme, P.E. (2006). 'Beyond control: wider implications for the management of biological invasions'. *Journal of Applied Ecology* 43: 835–847.

Hulme, P. E. (2009). 'Trade, transport and trouble: Managing invasive species pathways in an era of globalization'. *Journal of Applied Ecology* 46: 10–18.

Hulme, P.E., Bacher, S., Kenis, M., Klotz, S., Kühn, I., Minchin, D., Nentwig, W., Olenin, S., Panov, V., Pergl, J., Pyšek, P., Roques, A., Sol, D., Solarz, W. and Vilá, M. (2008). 'Grasping at the routes of biological invasions: A framework for integrating pathways into policy'. *Journal of Applied Ecology* 45: 403–414.

Inger, R.F. (1954). 'Systematics and zoogeography of Philippine amphibia'. *Fieldiana Zoology* 33: 183–531.

Inger, R.F. (1966). 'Systematics and zoogeography of the amphibia of Borneo'. *Fieldiana Zoology* 52: 1–402.

Inovejas, W.P. (1985). 'The American bullfrog: Tapped for lucrative farming in the Philippines'. *Habitat Philippines* 5(1):30–44.

Jeschke, J. M. and Strayer, D. L. (2008). 'Usefulness of bioclimatic models for studying climate change and invasive species'. *Annals of the New York Academy of Science* 1134: 1–24.

Kraus, F. (2007). 'Using Pathway Analysis to Inform Prevention Strategies for Alien Reptiles and Amphibians'. In: G.W. Witmer, W. C. Pitt and K. A. Fagerstone (eds.) *Managing Vertebrate Invasive Species: Proceedings of an International Symposium*, pp. 94–103. Colorado, United States: USDA/APHIS Wildlife Services, National Wildlife Research Center.

Kraus, F. (2009). *Alien Reptiles and Amphibians. A Scientific Compendium and Analysis* (Vol. 4). Springer Science & Business Media.

Kraus, F. (2015). 'Impacts from invasive reptiles and amphibians'. *Annual Review of Ecology, Evolution, and Systematics* 46: 75–97.

Kraus, F., Campbell, E.W., Allison, A. and Pratt, T. (1999). '*Eleutherodactylus* frog introductions to Hawaii'. *Herpetological Review* 30(1): 21–25.

Kraus, F. and Campbell, E.W. III (2002). 'Human–mediated escalation of a formerly eradicable problem: The invasive Caribbean frogs in the Hawaiian Islands'. *Biological Invasions* 4: 327–332.

Kulhanek, S.A., Ricciardi, A. and Leung, B. (2011). 'Is invasion history a useful tool for predicting the impacts of the world's worst aquatic invasive species?'. *Ecological Applications* 21(1): 189–202.

Kuraishi, N., Matsui, M. and Ota, H. (2009). 'Estimation of the origin of *Polypedates leucomystax* (Amphibia: Anura: Rhacophoridae) introduced to the Ryukyu Archipelago'. *Japan Pacific* 63: 317–325.

Leung, B., Finnoff, D., Shogren, J.F. and Lodge, D. (2005). 'Managing invasive species: Rules of thumb for rapid assessment'. *Ecological Economics* 55: 24–36.

Liu, C., Berry, P.M., Dawson, T.P. and Pearson, R.G. (2005). 'Selecting thresholds of occurrence in the prediction of species distribution'. *Ecography* 28: 385–393.

Matienzo, L.H. Jr. (1990). 'The bullfrog farmers of San Mateo'. *Greenfields* 20(3): 8–13.

Maynard, G. and Nowell, D. (2009). 'Biosecurity and Quarantine for Preventing Invasive Species'. In: M.N. Clout and P.A. Williams (eds.) *Invasive Species Management. A Handbook of Principles and Techniques*, pp. 1–18. New York, USA: Oxford University Press Inc.

McGeoch, M.A. and Squires, Z.E. (2015). *An Essential Biodiversity Variable Approach to Monitoring Biological Invasions: Guide for Countries.* GEO BON Technical Series 2.

Merino, G. (1936). 'Introduction of the giant toad, *Bufo marinus*, into the Philippines'. *The Philippine Journal of Agriculture* 7: 283–286.

Ministry of Natural Resources (1981). *Annual Report of the Ministry of Natural Resources*. Quezon City, Philippines: Ministry of Natural Resources.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G.A. and Kent, J. (2000a). 'Biodiversity hotspots for conservation priorities'. *Nature* 403: 853–858.

Myers, J.H., Simberloff, D., Kuris, A.M. and Carey, J.R. (2000b). 'Eradication revisited: Dealing with exotic species'. *Trends in Ecology and Evolution* 15(8): 316–320.

Olson, C.A. Beard, K.H. and Pitt, W.C. (2012). 'Biology and impacts of Pacific island invasive species. 8. *Eleutherodactylus planirostris,* the greenhouse frog (Anura: Eleutherodactylidae)'. *Pacific Science* 66(3): 255–270.

Olson, C., Diesmos, A. and Beard, K. (2014). '*Eleutherodactylus planirotris* (greenhouse frog): Philippines, Mindanao Island, Davao City, Ladislawa Villages'. *Herpetological Review* 45(4): 652–653.

Orrell, T. and Hollowell, T. (2017). *NMNH Extant Specimen Records. V. 1.7*. National Museum of Natural History, Smithsonian Institution. Occurrence Dataset. <https://doi.org/10.15468/hnhrg3>. Accessed via GBIF.org on 27 May 2017.

Parrish, J.D., Braun, D.P. and Unnasch, R.S. (2003). 'Are we conserving what we say we are? Measuring ecological integrity within protected areas'. *BioScience* 53(9): 851–860.

Pascual, R. (1987a). 'Bullfrogs for export: Not for meat but for laboratory use'. *Agribusiness Weekly Mr & Ms Special Edition* (August 28-3): 6–8.

Pascual, R. (1987b). 'Bullfrogs for meat (part 2): Inside the farm'. *Agribusiness Weekly Mr & M. Special Edition* (September 11-17): 8–9.

Pearce, J. and Ferrier, S. (2000). 'Evaluating the predictive performances of habitat models developed using logistic regression'. *Ecological Modelling* 133: 225–245.

Perrings, C., Dehnen-Schmutz, K., Touza, J. and Williamson, M. (2005). 'How to manage biological invasions under globalization'. *Trends in Ecology and Evolution* 20(5): 212–215.

Peterson, A.T. and Vieglais, D.A. (2001). 'Predicting species invasions using ecological niche modeling: New approaches from bioinformatics attack a pressing problem'. *BioScience* 51(5): 363–371.

Phillips, S.J. (2005). *A Brief Tutorial on Maxent*. AT&T Research. <https://biodiversityinformatics.amnh.org/open_source/maxent/Maxent_tutorial2017.pdf>.

Phillips, S.J., Dudik, M. and Schapire, R.E. (2004). 'A maximum entropy approach to species distribution modeling'. *Proceedings of the Twenty-First International Conference on Machine Learning*: 655–662.

Phillips, S.J. Anderson, R.P. and Schapire, R.E. (2006a). 'Maximum entropy modeling of species geographic distributions'. *Ecological Modelling* 190(3): 231–259.

Phillips, B.L., Brown, G.P., Webb, J.K., and Shine. R. (2006b). 'Invasion and the evolution of speed in toads'. *Nature* 439: 803.

Phillips, S.J. and Dudík, M. (2008). 'Modeling of species distributions with Maxent: New extensions and a comprehensive evaluation'. *Ecography* 31: 161–175.

R Core Team (2016). *R: A language and environment for statistical computing.* Vienna, Austria: R Foundation for Statistical Computing. <http://www.R–project.org/>.

Rabor, D.S. (1952). 'Preliminary notes on the giant toad, *Bufo marinus* (Linn.), in the Philippine Islands'. *Copeia* 4: 281–282.

Ripley, B., Venables, B., Bates, D.M., Hornik, K., Gebhardt, A. and Firth, D. (2015). *Support Function and Datasets for Venables and Ripley's MASS*. R package version 7.3-45.

Sheather, S.J. and Jones, M.C. (1991). 'A reliable data-based bandwidth selection method for kernel density estimation'. *Journal of the Royal Statistical Society Series B* 53: 683–690.

Shigesada, N., Kawasaki, K., and Takeda, Y. (1995). 'Modeling stratified diffusion in Biological Invasions'. *American Naturalist* 146 (2): 229–251.

Soriano, P. (1964). 'Notes on the food habits of the giant toad (*Bufo marinus* Linn.) in the Philippines'. *Philippine Journal of Plant Industry* 29 (3-4): 79–86.

Stohlgren, J.S. and Schnase, J.L. (2006). 'Risk analysis for biological hazards: What we need to know about invasive species?'. *Risk Analysis* 26(1): 163–173.

Stohlgren, T.J. and Jarnevich, C.S. (2009). 'Risk Assessment of Invasive Species'. In: M.N. Clout and P.A. Williams (eds.) *Invasive Species Management. A Handbook of Principles and Techniques*, pp. 19–35. New York, United States: Oxford University Press.

Suarez, A.V., Holway, D.A. and Case, T.J. (2001). 'Patterns of spread in biological invasions dominated by long-distance jump dispersal: Insights from Argentine ants'. *Proceedings of the National Academy of Sciences* 98(3): 1995–1100.

Surendran, H. and Vasudevan, K. (2013). 'Recent introduction and spread of Indian bullfrog *Hoplobatrachus tigerinus* (Daudin, 1802) into the Andaman Islands'. *Aliens* 33: 42–43.

Swets, J.A. (1988). 'Measuring the Accuracy of Diagnostic Systems'. *Science* 240: 1285–1293.

Sy, E.Y. and Salgo, J. (2015). 'Geographic distribution: *Eleutherodactylus planirostris* (greenhouse frog): Philippines, Cebu Island, Cebu City, Barangay Bulacao'. *Herpetological Review* 46(2): 212.

Sy, E.Y., Eleazar, B.I., Achacoso, R. and Diesmos, A.C. (2015a). 'Geographic distribution: *Eleutherodactylus planirostris* (greenhouse frog): Philippines, Luzon Island, Metropolitan Manila, Quezon City'. *Herpetological Review* 46(1): 56.

Sy, E.Y., Martyr, J.C. and Diesmos, A.C. (2015b). 'Geographic distribution: *Eleutherodactylus planirostris* (greenhouse frog): Philippines, Negros Island, Negros Occidental, Bacolod City, Barangay Villamonte'. *Herpetological Review* 46 (1): 56.

Sy, E.Y. (2017a). 'Geographic distribution: *Eleutherodactylus planirostris* (greenhouse frog): Philippines, Palawan Island'. *Herpetological Review* 48(3): 583.

Sy, E.Y. (2017b). 'Geographic distribution: *Eleutherodactylus planirostris* (greenhouse frog): Philippines, Mactan Island'. *Herpetological Review* 48(3): 583.

Taylor, E.H. (1920). 'Philippine amphibia'. *The Philippine Journal of Science* 16 (3): 213–359.

Taylor, E.H. (1922). 'Additions to the herpetological fauna of the Philippine Islands, II'. *The Philippine Journal of Science* 21(2): 257–304.

Ugale, C.T. Jr. (1976). 'Big profits from bullfrogs'. *Greenfields* 6(4): 4–7.

Urbanes, P.O. Jr. (1988). 'Raising bullfrogs brings on profits by leaps and bounds'. *Animal Husbandry and Agricultural Journal* 22(6): 29, 37.

Urbanes, P.O. Jr. (1990). 'Export of cultured, preserved bullfrogs a lucrative business'. *Animal Husbandry and Agricultural Journal* 24(2): 38.

Van Der Plank, J E. (1967). 'Spread of Plant Pathogens in Space and Time'. In: P.H., Gregory and P.A. Monteith (eds.) *Airborne Microbes*, pp. 227–247. Cambridge, United Kingdom: Cambridge University Press.

Venables, W.N. and Ripley, B.D. (2002). *Modern Applied Statistics with S*. Fourth edition. Springer.

Venette, R.C., Kriticos, D.J., Magaret, R.D., Koch, F.H., Baker, R.H.A., Worner, S.P., Gómez Raboteaux, N.N., McKenney, D.W., Dobesberger, E.J., Yemshanov, D., De Barro, P.J., Hutchinson, W.D., Fowler, G., Kalaris, T.M. and Pedlar, J. (2010). 'Pest risk maps for invasive alien species: A roadmap for improvement'. *BioScience* 60(5): 349–362.

Wells, K.D. (2007). *The Ecology and Behaviour of Amphibians*. Chicago, United States: The University of Chicago Press.

Wittenberg, R. and Cock, M.J.W. (eds.) (2001). *Invasive Alien Species: A Toolkit of Best Prevention and Management Practices*. Wallingford, Oxon, United Kingdom: CABI International.

Worton, B.J. (1989). 'Kernel methods for estimating the utilization distribution in home-range studies'. *Ecology* 70(1): 164–168.

In situ evaluation of an automated aerial bait delivery system for landscape-scale control of invasive brown treesnakes on Guam

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Abstract After decades of biodiversity loss and economic burden caused by the brown treesnake invasion on the Pacific island of Guam, relief hovers on the horizon. Previous work by USDA Wildlife Services (WS) and its National Wildlife Research Center (NWRC) demonstrated that brown treesnake numbers in forested habitats can be dramatically suppressed by aerial delivery of dead newborn mouse (DNM) baits treated with 80 mg of acetaminophen. However, manual bait preparation and application is impractical for landscape-scale treatment. WS, NWRC, and the US Department of the Interior have collaborated with Applied Design Corporation to engineer an automated bait manufacturing and delivery system. The core technology is an aerially delivered biodegradable "bait cartridge" designed to tangle in the tree canopy, making the acetaminophen bait available to treesnakes and out of reach of terrestrial non-target organisms. When mounted on a rotary- or fixed-wing airframe, the automated dispensing module (ADM) unit can broadcast 3,600 bait cartridges at a rate of four per second and can treat 30 hectares of forest at a density of 120 acetaminophen baits per hectare within 15 minutes of firing time. We conducted the first *in situ* evaluation of the ADM in July 2016. Initial acetaminophen bait deployment rates (proper opening of the bait cartridge for canopy entanglement) were low, and mechanism jams were frequent due to internal friction and wind forces; on-site remedial engineering improved these performance measures. Bait cartridge placement and spacing were accurate (average 8.9 m along 9 m swaths) under various flight heights and speeds. Canopy entanglement of properly-deployed acetaminophen baits was high (66.6%). Although only a small proportion (5.9%) of radio transmitter-equipped acetaminophen baits were confirmed to have been taken by brown treesnakes, the baiting density was high enough to make it likely that a significant proportion of brown treesnakes in the area had taken acetaminophen baits. With subsequent improvements in system reliability, the automated bait cartridge manufacturing and delivery system is poised to transition from research and development to operational field implementation. Applications include reduction of brown treesnake numbers around transportation infrastructure and within core habitats for the reintroduction of native birds extirpated by this troublesome invasive predator.

Keywords: invasive species suppression, invasive vertebrate predator, public-private partnership, scaling up, technical innovation, toxic baits

INTRODUCTION

The brown treesnake (*Boiga irregularis*) is a nocturnal, arboreal predator that was probably introduced on the island of Guam after World War II as a passive stowaway in cargo from the Admiralty Islands north of New Guinea (Rodda & Savidge, 2007; Richmond, et al., 2014). Lacking natural predators on Guam, the population of brown treesnakes irrupted, reaching as many as 50–100 brown treesnakes per hectare in some areas (Rodda, et al., 1999). Brown treesnakes colonised the entire island of Guam (54,930 ha) in about 20–30 years (Savidge, 1987). The brown treesnake has been – and continues to be – a threat to the economy and ecology of Guam, and is currently the subject of a cooperative programme to control brown treesnake populations on the island and prevent its spread throughout the Pacific Basin and other vulnerable locations (Clark, et al., 2018). Owing to the significant ecological and economic damages caused by the brown treesnake on Guam, the potential for the brown treesnake to be spread to other Pacific Islands, including Hawai`i, is of great concern (Shwiff, et al., 2010).

Landscape-scale suppression of brown treesnakes is desirable in habitats adjacent to transportation network infrastructure (e.g., cargo terminals), to reduce the risk of accidental transport to other vulnerable ecosystems, and within key habitats for the recovery of Guam's native wildlife. Because of the great amount of inaccessible and topographically challenging forest habitat on Guam, aerial delivery of brown treesnake suppression tools is key to the management of this species on a landscape scale. Dead newborn mice (DNM) dosed with 80 mg of acetaminophen have proven to be safe and effective baits for lethal control of brown treesnakes (Savarie, et al., 2001; Johnston, et al., 2002; Clark, et al., 2012) and are registered with the US Environmental Protection Agency (EPA) as an approved pesticide (Registration No. 56228-24, Revised 06/2018). To be effectively delivered to the forest canopy where they are available to foraging brown treesnakes and inaccessible by terrestrial non-targets, the baits must be coupled with a 'flotation device' intended to entangle in foliage (Savarie & Tope, 2004).

Through a previous project, the US Department of Agriculture (USDA) Animal and Plant Health Inspection Service (APHIS) Wildlife Services National Wildlife Research Center (NWRC) has demonstrated that brown treesnake abundance in Guam's forests can be suppressed via the aerial application of DNM baits adhered to paper streamers (Dorr, et al., 2016). During this prior study, baits were hand-prepared and hand-broadcast from a helicopter. While this method of treatment proved effective on a small scale (two 55 ha plots), manual bait preparation and application is economically impractical for larger landscape-scale treatments. In scaling up to meet the challenge of landscape-scale control of brown treesnakes, one of the principal logistical concerns is the obvious need to automate both bait production and the aerial dispensing of baits. In response to this need, NWRC, primarily funded by the US Department of the Interior Office of Insular Affairs, has partnered with a private, small business engineering company (Applied Design Corporation, Boulder, Colorado) to develop a brown treesnake suppression system that offers the capability to achieve precise distribution of thousands of baits in a matter of minutes, through a fully-integrated

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solution that encompasses bait cartridge production, an aerial bait cartridge dispensing system, and supporting infrastructure and logistics for practical manufacturing, storing, and flight-line handling of bait cartridges.

Automated Bait Manufacturing System (ABMS)

Many of the functional details of the three-stage ABMS are currently considered proprietary information pending application for US and foreign patent protection. The descriptions provided below will suffice as a basic functional explanation.

The first of three bait cartridge manufacturing stations is the Gluer/Placer Station (Station 1) where the DNM are distributed on moulded pulp paper trays and an 80 mg acetaminophen tablet is adhered to the DNM via a hot-melt adhesive. At the final stage of Station 1, the individual capsules containing the acetaminophen tablet and DNM are cut from the paper trays and fed into a transport cassette for transfer to the Assembly/Winder Station (Station 2). Hereafter, a DNM with an adhered acetaminophen tablet will be referred to as an “acetaminophen bait”.

At Station 2, the capsule is folded and held closed by pinching at the paper hinge between the two capsule halves. This pinched paper hinge, hereafter referred to as the ‘tang,’ is inserted into a slotted pressed pulp paper end cap. One end of a biodegradable plastic ribbon is adhered to the endcap and the entire assembly is rotated until the ribbon is wound around the length of the capsule in a ‘barber pole’ fashion. The terminal end of the ribbon is then adhered to the paper capsule. An exterior cardboard tube is placed over the wound assembly, with the end cap tightly pressed into the tube; this entire resulting assembly, comprised of the acetaminophen bait, capsule, streamer, and end cap, enclosed within the external tube, is referred to as a “bait cartridge”. The entire bait cartridge (Fig. 1) is biodegradable.

The final manufacturing station is Packaging (Station 3). Completed bait cartridges are automatically fed to the packaging station, where they are gathered and placed into a corrugated plastic case (900 bait cartridges per case). Filled cases are shrink wrapped, placed on a shipping pallet, and frozen. A complete pallet of 40 cases holds 36,000 bait cartridges, enough to treat 300 ha at the current EPA-approved maximum application rate of 120 acetaminophen baits/ha.

Fig. 1 When deployed, the bait capsule and outer bait cartridge tube are joined by a length of unfurled ribbon intended to entangle in the forest canopy when applied aerially.

Automated Dispensing Module (ADM)

The Automated Dispensing Module (ADM; Fig. 2) is comprised of three main components: 1) four magazines; 2) an electro-mechanical firing unit on a tilt-plate; and 3) a frame, which holds the power supply battery, the computer control module, and integrates the other components into a single functional ADM. The frame is mounted within the hold of the aircraft.

Each magazine is comprised of a body with two halves hinged at the back, allowing the payload area to be fully exposed, and a faceplate. The opened magazine receives the contents of one case (900 bait cartridges). Upon loading, the bait cartridges receive a final inspection for manufacturing imperfections or shipping damage which may adversely affect smooth feeding through the magazine and into the firing unit (Fig. 3).

Fig. 2 The ADM is comprised of four firing units and four 900-cartridge magazines along with an onboard battery and control electronics (not visible).

Fig. 3 Bait cartridges are inspected for manufacturing imperfections or shipping damage that might impede smooth feeding and ejection.

A magazine can be loaded with a case of bait cartridges and prepared for flight in two to three minutes. Once four magazines are prepared, they are loaded into the aircraft-mounted ADM frame (Figs. 4 and 5). The bait cartridge exit door on each magazine is then opened, allowing bait cartridges to flow into the firing unit feed chute. A full payload of 3,600 bait cartridges is sufficient to treat 30 ha of forest at the maximum application rate. At full performance, this area can be treated at 120 acetaminophen baits/ha within 15 minutes of firing time (Fig. 6). An additional set of magazines allows for the next payload to be prepared while the current payload is being applied.

A payload manager and the pilot are the only personnel aboard the aircraft. As directed by the payload management software, the computer control module engages the firing units within the ADM to fire bait cartridges at the proper rate to match the aircraft's current ground speed and intended acetaminophen bait application rate. The payload management software detects when a port is jammed or a magazine is empty and increases the firing rate of the other three ports to maintain the desired bait cartridge delivery rate.

Fig. 4 Magazines are loaded into the helicopter-mounted ADM frame.

Fig. 5 Complete ADM with loaded magazines mounted in a McDonnell-Douglass MD 500D helicopter.

Aerial navigation is achieved by following a preprogrammed mission plan in the payload management software, which details the transects to be flown. The pilot is provided with an LCD display, a "virtual lightbar," that provides realtime feedback as to whether the aircraft is on the prescribed flight path and what corrective movements are needed to return to the path. The payload manager manually toggles on bait cartridge firing when over the treatment area, and toggles it off when the flight path is complete. After an 'ag turn' (an aerial maneuver to quickly reverse directions) the next flight path is flown in the opposite direction. This is repeated until the payload is expended or the treatment area has been fully covered.

Objectives

This report describes the first *in situ* evaluation of this system through the experimental treatment of 110 hectares of forest on Guam. The major objectives were to evaluate: 1) the ground support work flow and performance of the automated dispensing module in-flight; 2) the precision of spatial coverage of the treatment area; and 3) the proper deployment of bait cartridges into the forest canopy and the fate of acetaminophen baits once distributed into the environment.

MATERIALS AND METHODS

Study site

The evaluation was conducted over 110 ha of secondary forest on the Marbo Annex of Andersen Air Force Base (typically referred to as "Andy South") in Yigo, Guam, at approximately 13.508°N, 144.873°E. This site was selected because: 1) there is low risk to threatened or endangered species; 2) the habitat is representative of much of Guam's forests; and 3) it is on a closed military facility with restricted public access.

ADM performance

We assessed the performance of the ADM through two trial applications of acetaminophen baits, simulating operational applications for brown treesnake control. The first application was initially scheduled to be completed on 19 July 2016, during which 13,200 acetaminophen

Fig. 6 Bait cartridges dispensed in flight.

baits would be applied over the 110 ha treatment area (120/ha). A second application was scheduled to occur three days later. For the purposes of this report, we define an "application" as a treatment of an area with aerially-distributed acetaminophen baits within the usage restrictions described in the EPA label.

A McDonnell-Douglas MD 500D (Fig. 5) and pilot were contracted from Hansen Helicopters (Tamuning, Guam) to perform aerial bait cartridge delivery. GoPro video cameras (GoPro, Inc., San Mateo, California) were positioned at various locations on the helicopter to document and evaluate bait cartridge ejection and deployment success.

On the night prior to flight operations, bait cartridge cases required for the next day's application were removed from the freezer to thaw and were stored overnight in an air-conditioned workspace to minimise condensation. The plastic wrapping on the cases were left intact to ensure that all condensation would occur on the external surface of the plastic wrap rather than on the paper-based bait cartridges themselves.

Bait cartridge coverage

Bait cartridge spacing trials were conducted to determine the accuracy and evenness of bait cartridge distribution at varying flight heights and airspeeds. Three lanes of approximately 200 m were delineated with orange traffic cones within an open grassy area at the treatment site. The helicopter, traveling at 50 knots, distributed bait cartridges along each flight line at heights of 25 m, 50 m, and 100 m above ground level. A ground crew attempted to locate all bait cartridges and measured their distance from the ideal flight path and the distance to the next bait cartridge along that path. A second round of transects was flown, this time at 60 knots, to determine the effect of airspeed on accuracy and spacing.

The completeness and the evenness of the spatial coverage of the treatment area was determined by recording the GPS flight paths in the payload management software, and generating coverage maps. Flight path segments were highlighted where the ADM unit was firing.

Acetaminophen bait fate

Methods for monitoring of radio transmitter-equipped baits were modified from procedures established by Dorr, et al. (2016). During each treatment, a subset of baits was prepared containing small 1.0 g VHF radio transmitters (Holohil BD-2H with internal helical antennae, Holohil Systems Ltd., Carp, Ontario, Canada) implanted in the acetaminophen bait DNM. Transmitter-equipped bait cartridges were placed directly in the ADM firing port unit so that they would be deployed simultaneously at the beginning of the flight path, to be followed by bait cartridges without transmitters.

An acetaminophen bait is considered properly "deployed" when the inner capsule assembly slides out of the outer cardboard tube, unfurling the ribbon to allow entanglement in the forest canopy. While some acetaminophen baits may deploy on impact with treetops, the system is designed for the acetaminophen bait to deploy in the air immediately upon ejection of the bait cartridge from the ADM.

Immediately after being aerially distributed, field technicians with handheld VHF receivers located the transmitter-equipped baits and recorded: bait cartridge location, position (in tree/vegetation or on ground), type of vegetation the bait cartridge was suspended from, height above ground, whether the bait cartridge was actually seen or its location was estimated, whether the acetaminophen bait was properly deployed and the DNM available for take by a brown treesnake, whether the acetaminophen tablet was still adhered to the mouse, and other notes about the circumstances of the condition and location of the acetaminophen bait and its availability for take by a brown treesnake.

If a DNM became separated from the bait cartridge and was on the ground but still had the acetaminophen tablet attached, it was considered intact and available for take by a brown treesnake. If the acetaminophen bait did not deploy properly and the DNM was not available to be taken, the bait cartridge and transmitter were recovered and that trial was ended. After deployment-day data were collected, the transmitter-equipped baits were left to determine the fate of acetaminophen baits over the next 48–72 hours. On each day following the application, each transmitter was re-located and the following data were recorded: whether the acetaminophen bait was still present and viable, whether the acetaminophen tablet was still attached, whether the acetaminophen bait was consumed by a brown treesnake or a non-target, whether the brown treesnake or non-target was alive or dead, whether the transmitter had moved to a new location, and other notes about acetaminophen bait location and condition.

If acetaminophen baits were unconsumed and still viable, they were left for another night and located again the next day. If acetaminophen baits had been consumed by a brown treesnake or non-target that was still alive, it was left undisturbed and relocated daily to establish survival or time to death. While tracking transmitters, technicians were alert for carcasses of any dead organisms, including those that had ingested transmitter-equipped baits. Global Positioning System (GPS) locations and notes on the location and condition of carcasses were recorded. Carcasses were collected and stored frozen for future analytical chemistry to verify acetaminophen exposure.

RESULTS

ADM performance

The first application of bait cartridges commenced on schedule on 19 July 2016. Ground operations and logistical support proceeded according to plan. However, crew and video observations indicated poor ADM performance in two primary categories: 1) bait cartridge feed/ejection reliability and 2) percentage of bait cartridges properly deploying in flight. These problems with system performance resulted in frequent flight stoppages to resolve bait cartridge jams and address other engineering challenges. As a result, additional flight days on 20, 22, 23, 25, and 26 July were required to achieve the first complete coverage of the treatment area.

Reliable bait cartridge ejection was hampered in three primary manners: 1) mechanical jams in the firing unit; 2) "starvation" of the firing unit feed ramp (bait cartridges not arriving at the firing position from the magazine); and 3) impediment of ejection by aerodynamic forces. These are not distinct processes, with multiple possible interactions among them. These issues were resolved with a variety of on-the-fly field improvements, with the causes and effects noted for future ADM design improvements.

Acetaminophen baits that do not deploy from the bait cartridge constitute a waste of resources (because the toxicant is inaccessible to snakes) and a fruitless toxic input into the environment. While we did not expect 100% deployment, observations by ground crew and video camera evidence indicated that initial acetaminophen bait deployment rates were unacceptably low at far less than 50%. Acetaminophen bait deployment issues generally fell into two categories: inadequate rotational energy imparted by the firing unit to overcome external air resistance effects and internal friction between the sliding components of the bait cartridge.

Air resistance effects were largely mitigated by employing adjustable baffles near the firing unit ejection ports to disrupt ejection-inhibiting air currents. Internal friction effects were traced to excessive friction between the bait cartridge capsule 'tang' and end cap. As manufactured, the tang (folded paper hinge) of the interior clamshell is seated in the slot of the end cap to prevent rotation of the internal assembly during manufacturing and unwinding of the ribbon during shipping and handling. However, it was discovered that the tension of the ribbon wound around the clamshell capsule caused the inner assembly to rotate slightly and the tang to twist against the sides of the slot in the end cap. This friction, along with the taut wind of the ribbon, created a 'locking' force, holding the entire assembly together and resisting the available centrifugal force which would otherwise deploy the acetaminophen bait properly. We determined that tearing off the paper tang would relieve the friction against the end cap slot, greatly increasing the deployment rate. For the second application, all bait cartridges were prepared by manual removal of the paper tang.

After system modifications were made, the second application was re-scheduled for 29 July 2016 (three days after the completion of the first application in accordance with EPA label restrictions). During this application, bait cartridge ejection and acetaminophen bait deployment were far more reliable. Bait cartridge jams in firing ports were less frequent and were promptly cleared. The only significant delay occurred when an ejector unit bearing broke; a temporary bushing replacement was fabricated and the ADM was returned to service within a few hours. Aside from this stoppage, the entire second application was completed within 2.5 hours.

Even after the above-mentioned modifications, only 37.3% of acetaminophen baits (571 out of a sample of 1,528 bait cartridge ejections observed on video) deployed immediately, as intended. Bait cartridges could only reliably be observed for about a third of their trajectory to the canopy, and some certainly deployed lower in the airstream. Still more would have deployed upon impact with the canopy or the ground. Nonetheless, we determined that improvement is needed in the reliability of aerial deployment of acetaminophen baits. Though there is no way to be certain of the actual deployment rates, we presume the realised acetaminophen bait deployment rate to be <50% for the overall acetaminophen bait application period.

Bait coverage

Bait cartridge placement and spacing was tested on 28 July 2016. Wind conditions during all flights were recorded at 0 to 1 on the Beaufort scale (0 = < 1 km/h, calm, smoke rises vertically; 1 = 1–5 km/h, light air, wind motion visible in smoke). When air movement was detectable, it was moving north to north-northwest. Flight direction was west to east or east to west. Bait cartridge distributions over trial flight paths are depicted in Fig. 7.

Placement along target flight paths and within 9-m swaths was very accurate and consistent. The one exception was the run at 100 m flight height at 50 knots airspeed; these results are inconsistent with the other five, and we consider this to be an anomalous lapse in pilot flight accuracy. Results do not appear to be influenced by the difference between 50 and 60 knots airspeed. Likewise, accuracy of placement along paths did not appear to be influenced by flight height. The most challenging combination of higher flight speed (60 knots) and highest flight height (100 m) resulted in an acceptable distribution pattern. Spacing between bait cartridges along a given flight path was highly variable, but the mean overall spacing of 8.9 m was virtually identical to the target spacing of 9 m.

Due to frequent flight stoppages during the first application, the full site coverage was achieved piecemeal over several days, with the entire area being treated by the 6th flight day. While the appropriate number of bait cartridges was deployed, the evenness of transect spacing was of reduced importance compared to overcoming the engineering challenges. The second application of acetaminophen baits was relatively uninterrupted. Flight paths were flown as planned which, along with increased pilot and payload manager experience, resulted in a much more even treatment (Fig. 8).

Acetaminophen bait fate

On 26 July 2016, 28 transmitter-equipped baits were broadcast over the treatment site. On 29 July 2016, an additional 23 were broadcast, for a total of 51 transmitter-equipped baits. The conditions of acetaminophen baits on the day of deployment are summarised in Table 1. Of the

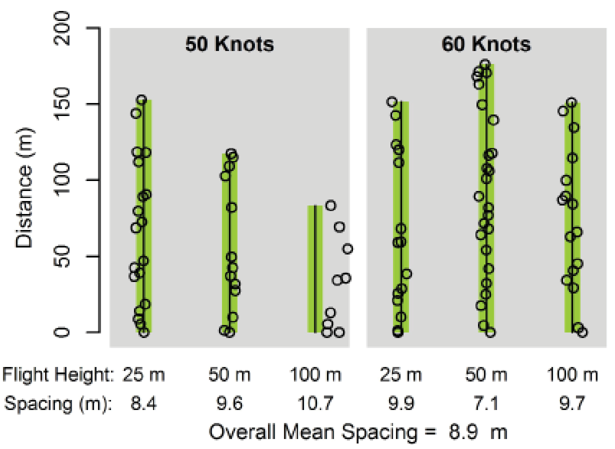

Fig. 7 Bait cartridge spacing and placement results. The centre line for each flight path indicates the target line, over which the pilot flew and bait cartridges were dispensed. Green boxes around the centre lines indicate 4.5 m on each side of the centre line, for a 9 m swath (the ideal flight path spacing for applications at 120 baits/ha). "Spacing" is the average distance from one bait cartridge to the next one along the flight path (target spacing was 9 m).

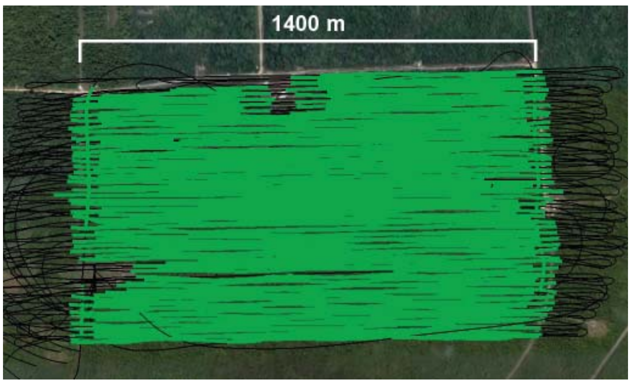

Fig. 8 Flight paths from the second bait application. Green swaths (portion of the flight paths where bait firing was actuated) are depicted at 9 m width, the optimal bait cartridge spacing for the 120 acetaminophen baits/ha application rate.

51 transmitter-equipped baits, 92.2% deployed from the bait cartridges, with the acetaminophen baits available to brown treesnakes.

Thirty-four bait cartridges (66.6%) tangled in the canopy as intended (Fig. 9). Thirteen (25.5%) were on the ground, but open and available to be taken by ground-foraging brown treesnakes. Four (7.8%) were not deployed (closed) on the ground, making the bait and toxicant unavailable to the brown treesnake. During each application, one transmitter-equipped cartridge was in the canopy but could not be confirmed to have deployed; we consider it unlikely that an unopened bait cartridge would be caught in the canopy, so assumed that these acetaminophen baits deployed.

Fig. 9 Desired canopy entanglement and acetaminophen bait exposure.

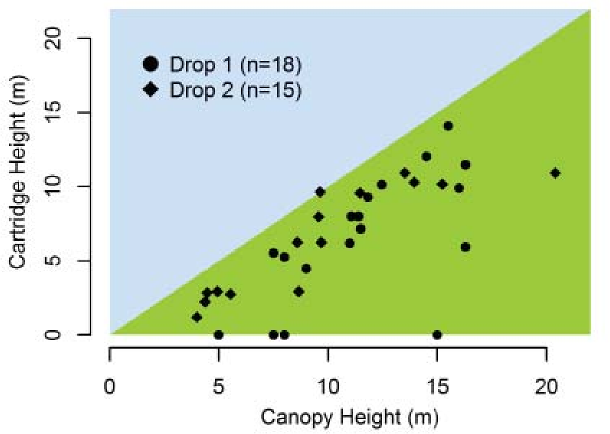

Fig. 10 Hanging height of the bait cartridge (y-axis) in relation to the height of the canopy at that location (x-axis).

Of the 47 opened bait cartridges, two from each application had the acetaminophen tablet detached from the DNM, making it an ineffective acetaminophen bait. In total, 41 of the 51 acetaminophen baits (80.4%) had acetaminophen tablets attached to the DNM and were available for take by a brown treesnake. This should be viewed as the overall successful bait deployment rate for this sample of baits.

Of the 51 transmitter-equipped baits, canopy height and acetaminophen bait height data were available on 33 acetaminophen baits that successfully deployed (18 from Application 1 and 15 from Application 2). The hanging height of the acetaminophen bait with respect to the canopy height is represented graphically in Fig. 10. In a linear regression, canopy height and acetaminophen bait height were significantly correlated ($p << 0.001$, adjusted $R^2 = 0.694$; the four bait cartridges on the ground were not included in the regression). These results show that the majority of deployed acetaminophen baits were entangled within a few metres of the top of the canopy.

Deployed and intact acetaminophen baits were re-checked daily, with very few confirmed takes by brown treesnakes or non-target organisms (Table 2). Of the 51 transmitter-equipped baits, three (5.9%) were confirmed by visual sighting to have been taken by brown treesnakes, or 7.3% of the 41 transmitter-equipped baits known to be available and intact. The 95% binomial confidence interval (logit parameterisation) for the estimated take rate of 5.9% is 1.9% to 16.7%; given the small number

Table 1 Status of transmitter-equipped bait cartridges following ejection from ADM. "Deployed" means the inner capsule completely exited the outer tube and the acetaminophen bait was available for take by a brown tree snake. "Intact" means the acetaminophen tablet was still attached to the bait mouse and available to be taken by a brown treesnake.

Bait cartridge status	Application 1 (n=28)	Application 2 (n=23)	TOTAL (n=51)
Opened in canopy*	19 (67.9%)	15 (65.2%)	34 (66.6%)
Opened on ground	7 (25.0%)	6 (26.1%)	13 (25.5%)
Not deployed	2 (7.1%)	2 (8.7%)	4 (7.8%)
Unknown	1 (3.6%)	1 (4.3%)	2 (3.9%)
Total deployed*	26 (92.3%)	21 (91.3%)	47 (92.2%)
Total known deployed and intact**	23 (82.1%)	18 (78.3%)	41 (80.4%)

*Assumes that "unknown" bait cartridges in canopy were deployed; **Does not assume "unknown" baits were intact.

Table 2 Transmitter-equipped acetaminophen baits taken by target (brown treesnake) or non-target species.

Species	Application 1	Application 2	Total
Brown tree snake	1	2	3
Monitor lizard	1*	0	1
Marine toad	0	2*	2
Unknown	0	1	1

*Transmitter recovered in faeces

of acetaminophen baits equipped with transmitters, the actual rate of acetaminophen bait take by brown treesnakes could vary widely. All three transmitters were regurgitated prior to death, so no transmitters were recovered in brown treesnake carcasses. All three transmitters taken by non-targets were later found in faeces; it is unclear whether any of these animals succumbed to acetaminophen toxicosis.

All vertebrate carcasses encountered during field activities were collected. This included three brown treesnakes and one marine toad (*Rhinella marina*).

DISCUSSION

ADM performance

Future improvements to the ADM will focus on: baffling of the airstream around the ejector ports to prevent interference with ejection; improved feeding of bait cartridges from redesigned magazines and; increased energetic impact imparted to the bait cartridge at the instant of firing in order to improve ejection and deployment reliability. Engineering modifications to the ABMS will further address the non-deployment issue through tighter quality control on bait cartridge imperfections and abatement of tang friction through a redesigned end cap.

Bait cartridge coverage

The accuracy of bait cartridge placement along flight lines was encouraging. There was very little air movement during these trials; under windier conditions, bait cartridges distributed from greater heights will be more likely to drift further from the intended flight path.

We attribute high variability in bait cartridge spacing along the flight lines to variability in the times at which acetaminophen baits deployed after being ejected from the ADM. When the acetaminophen bait deploys, wind drag increases greatly and the forward momentum is quickly attenuated, causing the bait cartridge to drop straight down. Acetaminophen baits that deploy later maintain forward momentum longer and will move farther along the flight path before landing. It is expected that bait cartridge modifications that improve acetaminophen bait deployment will also result in less variability in time of deployment, leading to more consistent spacing along flight paths.

Variability in spacing along the flight path does have the potential to affect bait cartridge placement accuracy at the edges of treatment areas where bait cartridge application begins and ends, potentially leading to a small number of bait cartridges landing outside of the desired treatment area. To make up for the inconsistency of bait cartridge density at these edges, it is advisable that another application flight should occur along these edges, perpendicular to the original flight paths, ensuring that the edges get a full treatment in a more controlled fashion, similar to coastal aerial rodenticide applications during island rodent eradications.

Variability in bait cartridge placement along and perpendicular to the flight path will add apparently random "noise" to the locations, as opposed to placing bait cartridges precisely on an idealised 9 x 9 m grid. This variability will not affect the ability to get acetaminophen baits into the movement areas of every brown treesnake. The greatest risk of gaps in coverage might arise from strong changes in wind direction, which might introduce strong biases in bait cartridge drift patterns. This will likely factor in with other considerations leading to recommendations not to apply baits during high wind conditions.

Wind effects at bait cartridge ejection ports and direct sunlight on the bait cartridge counter photogates resulted in unreliable bait cartridge counts as tabulated by the ADM onboard software. We ensured that EPA label application rate restrictions were not exceeded by confirming that no more than 14.66 cases (13,200 bait cartridges) were applied throughout the treatment area during each application period.

Acetaminophen bait fate

The proportion of transmitter-equipped baits taken by brown treesnakes was low (5.9%); however, only a very small portion of the bait cartridges distributed (0.19%) were equipped with transmitters. If we assume that half of the acetaminophen baits applied during both applications properly deployed and were viable, then there were 13,200 acetaminophen baits available for take by brown treesnakes. If 5.9% of those acetaminophen baits were taken by brown treesnakes, we would expect approximately 779 brown treesnakes to have taken an acetaminophen bait. If we assume a density of 25 brown treesnakes per hectare in this area (a conservative estimate based on the 25-50/ha range reported by Rodda, et al. 1999), 2,750 brown treesnakes would have been exposed to the treatment. If 779 brown treesnakes took acetaminophen baits and were killed, this would be a brown treesnake mortality of approximately 28% in what was effectively a single treatment (given the low deployment rate). The three acetaminophen baits visually confirmed to have been taken by brown treesnakes were found on the ground, apparently regurgitated. In previous NWRC lab efficacy trials of acetaminophen baits with acetaminophen tablets internally-implanted in the DNM (rather than glued to the exterior), 26% were regurgitated, but 100% of the caged brown treesnakes that regurgitated the acetaminophen bait died within 12 to 36 hours (Savarie, 2002). Based on that result, it is reasonable to assume that the brown treesnakes that had taken and regurgitated acetaminophen baits with transmitters in this study also died.

With respect to deployment and entanglement rates, caution should be taken in considering transmitter-equipped cartridges to be representative of the standard bait cartridges distributed during this evaluation. Machine-assembled bait cartridges were manually unwound and rewound by hand after the implantation of the radio transmitter; this may explain why transmitter-equipped cartridges deployed at a higher rate than those observed on video. The added mass of the transmitter may also have an effect on the forces exerted on various parts of the bait cartridge and acetaminophen bait assembly. However, it is also possible that unopened bait cartridges without transmitters actually did deploy lower in the air column (out of view of the video cameras) or upon impact with the canopy.

The overall reduction of brown treesnake abundance in the treatment area – as inferred from a foraging activity index based on take rates of non-toxic DNM from bait stations – is currently being monitored as a separate study for future publication.

CONCLUSION

Upon firing from the ADM, bait cartridge ejection and acetaminophen bait deployment reliability was initially low. Performance was improved dramatically with field-improvised remedial measures. It is estimated that <50% of acetaminophen baits deployed from the bait cartridges, resulting in an under-treatment compared to the target application rate of 120/ha. Canopy entanglement of acetaminophen baits that properly deployed was high. Aerial bait cartridge placement and spacing were satisfactorily accurate. Reliability of bait cartridge ejection and acetaminophen bait deployment will be a critical focus of bait manufacturing and delivery system improvements, increasing per-cartridge effectiveness. Future advancements of this technology may include adaptation for payload management by the pilot alone, incorporation of a longer-lasting artificial bait to replace the DNM, and increases in ejector unit and magazine capacity for greater payloADM.

With this evaluation – and subsequent improvements in system reliability – we consider the concept of automated bait production and aerial delivery to be fundamentally sound. For the first time in the decades-long saga of the brown treesnake invasion of Guam, the prospect of landscape-scale suppression hovers on the horizon.

DISCLAIMER

The use of trade or corporation names within this report is for the convenience of the user in identifying products. Such use does not constitute an official endorsement or approval of any product by the U.S. Department of Agriculture.

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AUTHOR CONTRIBUTIONS

S. Siers was the principal investigator for the field evaluation, establishing the study design, coordinating site access and environmental compliance, curating data, executing analyses and data visualisation, and writing the original draft of the manuscript. S. Siers, M. Messaros, C. Clark, and R. Gosnell supervised field evaluation activities. W. Pitt and M. Messaros conceptualised the automated bait cartridge manufacturing and delivery systems. M. Messaros was the chief engineer and developer of the intellectual property associated with the bait cartridge and associated manufacturing and delivery systems. W. Pitt, L. Clark, A. Shiels, J. Eisemann, and S. Siers contributed to programme administration, funding acquisition, and other matters associated with research and development of the automated systems. J. Eisemann coordinated pesticide registration and technology transfer matters, and contributed to on-site coordination of evaluation activities. All authors contributed to review and editing of the manuscript.

REFERENCES

Clark, L., Savarie, P.J., Shivik, J.A., Breck, S.W. and Dorr, B.S. (2012). 'Efficacy, effort, and cost comparisons of trapping and acetaminophen-baiting for control of brown treesnakes on Guam'. *Human–Wildlife Interactions* 6: 222–236.

Clark, L., Clark, C.S. and Siers, S.R. (2018). 'Brown tree snakes: methods and approaches for control.' In: W.C. Pitt, J.C. Beasley and G.W. Witmer (eds.) *Ecology and Management of Terrestrial Vertebrate Invasive Species in the United States*, pp. 107–134. New York: Taylor & Francis.

Dorr, B.S., Clark, C.S. and Savarie, P. (2016). *Aerial application of acetaminophen-treated baits for control of Brown treesnakes (RC-200925; NWRC study number: QA-1828)*. Environmental Security Technology Certification Program Final Report, Virginia, USA: Department of Defense, Alexandria.

Johnston, J.J., Savarie, P.J., Primus, T.M., Eisemann, J.D., Hurley, J.C. and Kohler, D.J. (2002). 'Risk assessment of an acetaminophen baiting program for chemical control of brown tree snakes on Guam: Evaluation of baits, snake residues, and potential primary and secondary hazards'. *Environmental Science and Technology* 36: 3827–3833.

Richmond, J.Q., Wood, D.A., Stanford, J.W. and Fisher, R.N. (2014). 'Testing for multiple invasion routes and source populations for the invasive brown treesnake (*Boiga irregularis*) on Guam: Implications for pest management'. *Biological Invasions* 17: 337–349.

Rodda, G.H., McCoid, M.J., Fritts, T.H. and Campbell III, E.W. (1999). 'Population trends and limiting factors in *Boiga irregularis*.' In: G.H. Rodda, Y. Sawai, D. Chiszar and H. Tanaka (eds.) *Problem Snake Management: the Habu and the Brown Treesnake*, pp. 236–254. New York: Cornell University Press.

Rodda, G.H. and Savidge, J.A. (2007). 'Biology and impacts of Pacific island invasive species: 2. *Boiga irregularis*, the brown tree snake (Reptilia: Colubridae)'. *Pacific Science* 61: 307–324.

Savarie, P.J., Shivik, J.A., White, G.C., Hurley, J.C. and Clark, L. (2001). 'Use of acetaminophen for large-scale control of brown tree snakes'. *Journal of Wildlife Management* 65: 356–365.

Savarie, P.J. (2002). *Acute oral toxicity of acetaminophen tablets to brown treesnakes, Unpublished report – QA-636*. Fort Collins, Colorado: National Wildlife Research Center.

Savarie, P.J. and Tope, K.L. (2004). 'Potential flotation devices for aerial delivery of baits to brown treesnakes'. In: R.M. Timm and W.P. Gorenzel (eds.) *Proceedings of the 21st Vertebrate Pest Conference*, pp. 27–30. Davis, California: University of California.

Savidge, J.A. (1987). 'Extinction of an island forest avifauna by an introduced snake'. *Ecology* 68: 660–668.

Shwiff, S.A., Gebhardt, K., Kirkpatrick, K.N. and Shwiff, S.S. (2010) 'Potential economic damages from introduction of brown tree snakes, *Boiga irregularis* (Reptilia: Colubridae), to the islands of Hawai'i'. *Pacific Science*, 64: 1–10.

The potential detrimental impact of the New Zealand flatworm to Scottish islands

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Abstract: The New Zealand flatworm, *Arthurdendyus triangulatus*, is an alien invasive species in The British Isles and the Faroes. It was probably first introduced after WWII and is an obligate predator of our native earthworms. It was initially considered a curiosity until observations in the 1990s in Northern Ireland found it could significantly reduce earthworm numbers. In 1992, it was scheduled under the Countryside and Wildlife Act 1981 then transferred to the Wildlife and Natural Environment (Scotland) Act in 2011 which makes it an offence to knowingly distribute the flatworm. A retrospective survey in Scotland showed that it was detected in botanic gardens, nurseries and garden centres in the 1960s but then spread to domestic gardens then finally to farms in the 1990s. Although the geographical distribution of *A. triangulatus* was initially confined to mainland Scotland it was subsequently found established on 30 Scottish Islands. Most of the islands are to the north and west of Scotland and have cool damp climates which are favoured by the New Zealand flatworm. These islands also generally have relatively poor soils that support grassland farming systems. Evidence from both Northern Ireland and Scotland suggests anecic species of earthworm which occur predominantly in grassland, which help drainage and are a source of food for both animals and birds are at particular risk from the flatworm. The detrimental impact of the flatworm on soil processes and wildlife has yet to be quantitatively evaluated but unlike many other invasive species there is currently no known means of control. The precautionary principle must be therefore applied wherever possible and every opportunity taken to stop its further spread.

Keywords: *Arthurdendyus triangulatus*, earthworms, invasive alien species, predator, Scotland

INTRODUCTION

Many of the Scottish islands have impoverished acidic soils (Boyd, 1957; Glentworth, 1979; Hudson, et al., 1982) which have been improved in the past by the addition of seaweed, lime or occasionally imported soil (Magnusson, 1997; Entwhistle, et al., 2000). Wind-blown calcareous sandy soils with a high shell content occurring in the north and west of Scotland, known as "machair" (Angus, 2001), are characterised with a defined flora and low input agriculture (Hudson, et al., 1982) and is a fragile ecosystem listed under the EU Habitats Directive Annex 1. Earthworms prefer soils with a near neutral pH and are therefore typically found in improved soils compared with those with a pH levels< 4 (Guild, 1951; Edwards & Bohlen, 1996). Boyd (1956, 1957) surveyed earthworms in the Hebrides and recorded a complex of 15 species from the machair, dominated by *Lumbricus rubellus, Aporrectodea caliginosa* and *Dendrobaena octaedra*. This contrasted with only six species under acidic heather soils, the dominant species being *L. rubellus and D. octaedra.* The genus of one of the recorded species, *Bimastos,* has since been removed from the British list (Sims & Gerard, 1985).

Stop-Bowitz (1968), suggested that in Norway some species e.g. *D. octaedra* and *L. rubellus* could have survived the Quaternary ice age and this may also have occurred in Scotland. However, many of the other recorded earthworm species were probably introduced by man to enhance the productivity of the land as occurred in New Zealand where productivity was increased significantly by the addition of European earthworm species (Stockdill, 1982). The only other place in the world where the New Zealand flatworm has become established is the Faroe Islands where earthworms have been found in closed association with human settlements possibly due to the inhabited areas being on fertile land (Enckell & Rundgren, 1988).

The New Zealand flatworm (*Arthurdendyus triangulatus*) was probably first accidentally introduced into the British Isles just after WWII but not officially recorded in Scotland until 1965 when it was considered a curiosity (Wakeman & Vickerman, 1979). However, Blackshaw (1990) reported that the presence of the New Zealand flatworm was associated with a decline in earthworm populations to below detectable levels in Northern Ireland. The results of a survey undertaken in Scotland during 1991–1992 (Boag, et al., 1994) indicated that the New Zealand flatworm was initially confined to botanic gardens, garden centres and nurseries but then spread to domestic gardens in the 1970s and finally to farms in the 1980s. It also showed that by 1992 it had spread to many parts of Scotland including the islands of Skye and Orkney. In the last 30 years the flatworm has been recorded from several other islands off the west and north coast of Scotland.

Further research indicated that the presence of the New Zealand flatworm reduced the abundance of anecic earthworm species (Jones, et al., 2001), populations of which were unlikely to fully recover (Murchie & Gordon, 2013). Anecic earthworm species are those which make vertical burrows and consume dead organic matter on the soil surface (Fraser and Boag, 1998), thus play a key role in soil nutrient processes and are considered ecosystem engineers (Lavelle, et al., 1997; Blouin, et al., 2013). Furthermore, they are also a major component of food for some mammals and birds (Boag & Neilson, 2006).

The New Zealand flatworm prefers cool damp conditions to survive (Boag, et al., 1998a) and this may have contributed to it being a problem predominantly in the north and west of Scotland, Ireland and the Faroe Islands compared with the east of Scotland and England (Jones & Boag, 1996). The flatworm is also dependent on the presence of earthworms which potentially restricts its distribution in Scotland as earthworms rarely occur in soils with a pH < 4 (Boag, et al., 1998b).

The aim of the present paper is to document the extent to which the New Zealand flatworm has become established in the Scottish islands and to consider the detrimental impact its presence might, in the future, have on island agriculture and wildlife.

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MATERIAL AND METHODS

The data used for this paper were the records of where the New Zealand flatworms were found by the general public in their gardens, parks etc. and these records over time have also shown how it spread. The New Zealand flatworm can easily be recognised by the general public as different from earthworms as it is flat, covered by a sticky mucus and pointed at both ends. Initially the records were collated by the National Museum of Scotland until this was taken over by the senior author after a survey financed by the Scottish Government (Boag, et al., 1994) and a 1995 BBC TV survey (Jones & Boag, 1996). Subsequent records have continued to be collected by staff at the James Hutton Institute and submitted to and curated by the National Biodiversity Network from where three additional island records were gleaned for this paper. More recently The Open Air Laboratory (OPAL) has run a citizen science survey for the New Zealand flatworm (<https://www.opalexplorenature.org>). The records are stored by the National Biodiversity Network and the senior author is the national expert on this species and verifies all records.

RESULTS

Most flatworm records were from individual households with a few from farms, garden centres and schools. Scotland has 790 offshore islands of which 95 are inhabited of which 30 islands recorded New Zealand flatworm (Table 1). These were distributed from Arran in the south of Scotland to Shetland in the north. Many of the infested islands had few inhabitants, but in general the number of flatworm records reflected the population size (Table 1).

This was demonstrated across the Orkney archipelago (Fig. 1) where there were 41 flatworm records, from a population > 17,000 on mainland Orkney compared with the outlying islands of Burray, Egilsay, Hoy, North Ronaldsay, Rousay and South Ronaldsay which had a combined population of > 1,700 and had only seven flatworm records. The Orkney mainland had the most records of all Scottish Islands even though it has a smaller land area than Skye, Shetland, Mull or Lewis. Of these larger islands Orkney is by far the most fertile with a large proportion covered with arable crops or permanent pasture (Dry & Robertson, 1982). Most island records only reported the presence of the flatworm but others reported a reduction or absence of native earthworms while others reported that large numbers of flatworms had been collected e.g. a householder from Baleshare killed 1,445 flatworms over a period between May 2015 and January 2016. Another householder from Skye regularly killed 20-40 flatworms daily with a reported maximum of 150, and an estimated total kill of 15,000 over a period of one year.

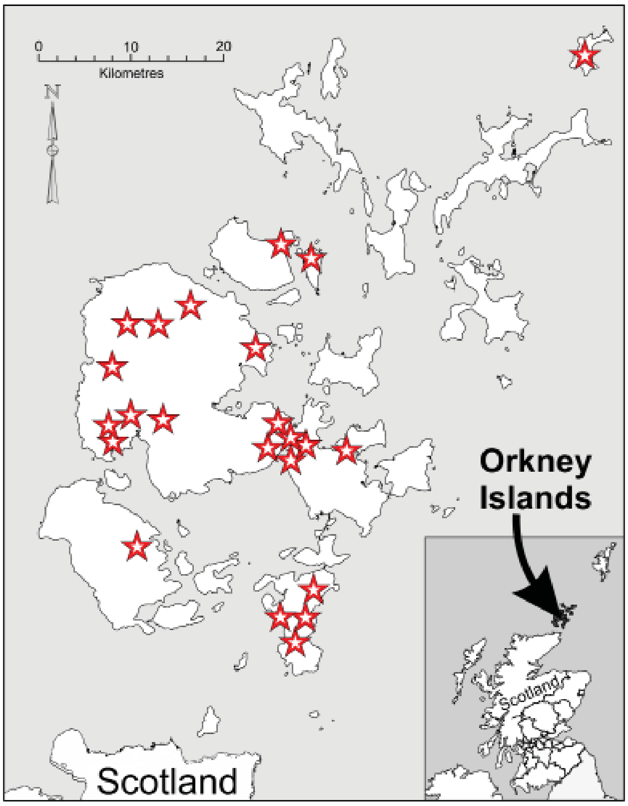

Fig. 1 Distribution of records of *Arthurdendyus triangulatus*, the New Zealand flatworm, in Orkney.

Table 1 Scottish islands infested with *Arthurdendyus triangulatus,* the New Zealand flatworm: the number of records; the human population and; area of the islands.

Island	No of records	Population	Hectares
Arran	5	5,058	43,201
Baleshare	1	58	910
Barra	3	1,078	5,875
Bressay	4	360	2,805
Burray	1	409	903
Bute	13	7,228	12,217
Coll	1	164	7,685
Easdale	1	59	25
Egilsay	1	26	650
Eriskay	1	143	703
Fair Isle	1	55	768
Gigha	3	110	1,305
Greater Cumbrae	2	1,376	1,168
Harris	3	1,916	50,119
Hoy	1	272	14,320
Iona	1	120	877
Islay	2	3,228	61,956
Isle of Seil	1	21	1,329
Lewis	14	18,500	163,695
Lismore	4	146	2,351
Mull	6	2,667	87,535
North Ronaldsay	1	72	690
North Uist	1	1,271	30,305
Orkney Mainland	41	17,162	52,325
Rousay	1	26	4,860
Shetland	13	22,000	96,879
Skye	12	10,008	165,625
South Ronaldsay	1	909	4,980
South Uist	1	1,754	32,026
Whalsay	1	14	1,970

DISCUSSION

The records received over the last 25 years show that the New Zealand flatworm is now widely distributed in the Scottish islands. Since it is an obligate feeder on earthworms, the presence of the flatworm also indicates that these islands must have had an abundance of earthworms. A possible reason for Orkney having a disproportionately greater number of New Zealand flatworm records compared with Lewis, Skye, Mull or Shetland is probably the fact that Orkney is formed from sedimentary rock while the others are igneous or metamorphic in origin (Dry & Robertson, 1982) thus more conducive to earthworm establishment and survival.

The New Zealand flatworm is known to have a deleterious impact on earthworms in mainland Scotland and Ireland (Jones, et al., 2001; Murchie & Gordon, 2013) and it can probably be assumed that is also the case on these islands. Apart from earthworms playing an important role in delivering soil function and ecosystem services (Lavelle, et al., 1997; Blouin, et al., 2013) they are an important constituent of the diets of some mammals and birds which live in the islands and are, in some cases, declining in number e.g. the lapwing (*Vanellus vanellus*). To help revive the decrease in lapwing numbers it has been proposed that lime should be added to increase the soil pH and hence encourage the build-up of earthworms upon which lapwing feed (McCallum, et al., 2015). Studies have also shown that earthworms are a major constituent of the diet of chough (*Pyrrhocorax pyrrhocorax*) (Meyer, 1990), a rare breeding corvid found on Islay and Colonsay with an estimate of c. 50 breeding pairs in 2014 (https://scotlandsnature.wordpress.com/2014/09/25/good-news-from-islay-as-population-grows/). It is therefore concerning that flatworms have been recorded from Islay as this may confound the conservation of chough on the island.

Apart from the direct impact of New Zealand flatworm on wildlife it has been estimated that it could have a potential detrimental economic effect on agriculture (Boag & Neilson, 2006). This is particularly relevant to small holdings with tight farm unit margins such as crofts. Circumstantial evidence from an area north of Dunoon infested with flatworm suggested that in addition to an accumulation of dead organic matter on the soil surface, undesirable plants such as rushes became established as a result of frequent flooding after rainfall events. The New Zealand flatworm may also become a problem where there are large amounts of arable land and permanent grassland which occurs in mainland Orkney. Agricultural land in Scotland can have a wide range of earthworm species including the anecic species which would be particularly at risk (Boag, et al., 1997).

No investigations have been undertaken to ascertain the actual impact of the New Zealand flatworm on either wildlife or agricultural production in the Scottish islands. Assumptions on the detrimental impact of the New Zealand flatworm must therefore be made based on the knowledge gleaned from the literature on the benefits that earthworms have on agricultural production and wildlife (Schmidt & Curry, 1999; Bartlett, et al., 2010). Unlike many other invasive plants and animals which have been successfully removed from Scottish islands e.g. mink from the Uists and rats from Canna (Bell, et al., 2011; Roy, et al., 2015) there are no prospects of the New Zealand flatworm being controlled on the Scottish islands once it has become established. Given the only mechanism known to spread the New Zealand flatworm is the human mediated movement of plant material every effort must be made to stop infested material reaching the islands by informing the general public of the threat that New Zealand flatworm poses.

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REFERENCES

Angus, S. (2001). 'Machair'. In: S. Angus (ed.) *The Outer Hebrides, Moor and Machair*, pp. 195–243. Cambridge: The White Horse Press.

Bartlett, M., Briones, M.J.I., Neilson, R., Schmidt, O., Spurgeon, D. and Creamer R.E. (2010). 'A critical review of current methods in earthworm ecology: From individuals to populations'. *European Journal of Soil Biology* 46: 67–73.

Bell, E., Boyle, D., Floyd, K., Garner-Richards, P., Swann, B., Luxmoore, R., Patterson, A. and Thomas, R. (2011). 'The ground-based eradication of Norway rats (*Rattus norvegicus*) from the Isle of Canna, Inner Hebrides, Scotland'. In: C.R. Veitch; M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 269–274. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Blackshaw, R.P. (1990). 'Studies on *Artioposthia triangulata* (Dendy) (Tricladida: Terricola), a predator of earthworms'. *Annals of Applied Biology* 118: 698–694.

Blouin, M., Hodson, M.E., Delgado, E.A., Baker, G., Brussaard, L., Butt, K.R., Dai, J., Dendooven, L., Peres, G., Tondon, J.E., Cluzeau, D. and Brun J. (2013). 'A review of earthworm impact on soil function and ecosystem services'. *European Journal of Soil Science* 64: 161–182.

Boag, B., Palmer, L.F., Neilson, R. and Chambers, S.J. (1994). 'Distribution and prevalence of the predatory planarian *Artioposthia triangulata* (Dendy) (Tricladida: Terricola) in Scotland'. *Annals of Applied Biology* 124: 165–171.

Boag, B., Palmer, L.F., Neilson, R., Legg, R. and Chambers, S.J. (1997). 'Distribution, prevalence and intensity of earthworm populations in arable land and grassland in Scotland'. *Annals of Applied Biology* 130: 153–165.

Boag, B., Yeates, G.W. and Johns, P.M. (1998a). 'Limitations to the distribution and spread of terrestrial flatworms with special reference to the New Zealand flatworm (*Artioposthia triangulata*)'. *Pedobiologia* 42: 495–503.

Boag, B., Jones, H.D., Evans, K.A., Neilson, R., Yeates, G.W. and Johns, P.M. (1998b). 'The application of GIS techniques to estimate the establishment and potential spread of *Artioposthia triangulata* in Scotland'. *Pedobiologia* 42: 504–510.

Boag, B. and Neilson, R. (2006). 'Impact of the New Zealand flatworm on agriculture and wildlife in Scotland'. *Proceedings Crop Protection in Northern Britain 2006*, pp. 51–56. Dundee, UK: Page Bros, Norwich Ltd.

Boyd, J.M. (1956). 'The Lumbricidae in the Hebrides II Geographical distribution'. *Scottish Naturalist* 68: 165–172.

Boyd, J.M. (1957). 'Ecological distribution of the Lumbricidae in the Hebrides'. *Proceedings of the Royal Society Edinburgh B* 66: 311–338.

Dry, F.T. and Robertson, J.S. (1982). *Soils and Land Capability for Agriculture, Orkney and Shetland.* Aberdeen, UK: University Press.

Edwards, C.A. and Bohlen, P.J. (1996). *Biology and Ecology of Earthworms*. London, UK: Chapman and Hall.

Enckell, P.H. and Rundgren, S. (1988). 'Anthropochorous earthworms (Lumbricidae) as indicators of abandoned settlements in the Faroe Islands'. *Journal of Archeaological Science* 15: 439–351.

Entwhistle, J.A., Dodgshon, R.A. and Abrahams, P.W. (2000). 'An investigation of former land-use activity through the physical and chemical analysis from soils from the Isle of Lewis, Outer Hebrides'. *Archaeological Prospection* 7: 171–188.

Fraser, P.M. and Boag, B. (1998). 'The distribution of lumbricid earthworm communities in relation to flatworms: a comparison between New Zealand and Europe'. *Pedobiologia* 42: 642–553.

Glentworth, R. (1979). 'Observations on the soils of the Outer Hebrides'. *Proceedings of the Royal Society of Edinburgh B* 77: 123–137.

Guild, Mc.L.W.J. (1951). 'The distribution and population density of earthworms (Lumbricidae) in Scottish pasture fields'. *Journal of Applied Ecology* 20: 88–97.

Hudson, G., Towers, W., Bibby, J.S. and Henderson, D.J. (1982). *The Outer Hebrides: Soil and Land Capability for Agriculture, with a Contribution from J. S. Robertson. Handbook of the Soil Survey of Scotland*. Aberdeen, UK: University Press.

Jones, H.D. and Boag, B. (1996). 'The distribution of the New Zealand and Australian flatworms (Platyhelminthes: Turbelliaria: Tricladida: Terricola) in the British Isles – the Scottish survey and MEGALAB WORMS'. *Journal of Natural History* 30: 955–975.

Jones, H.D., Santoro, G., Boag, B. and Neilson, R. (2001). 'The diversity of earthworms in 200 Scottish fields and the possible effect of the New Zealand flatworms (*Arthurdendyus triangulatus*) on earthworm populations'. *Annals of Applied Biology* 139: 75–92.

Lavelle, P., Bignell, D., Lepage, M., Wolters, V., Roger, P., Ineson, P., Heal, O.W. and Dhillion, S. (1997). 'Soil function in a changing world: The role of invertebrate ecosystem engineers'. *European Journal of Soil Biology* 33: 159–193.

Magnusson, M. (1997). *Rum: Nature's Island*. Edinburgh, UK: Luath Press.

McCallum, H.M., Park, K.J., O'Brien, M.G., Gimona, A., Poggio, L. and Wilson, J.D. (2015). 'Soil pH and organic matter content add explanatory power to the northern lapwing *Vanellus vanellus* distribution models and suggest soil amendment as a conservation measure on upland farmland'. *Ibis* 167: 677–687.

Meyer, R.M. (1990). 'Observations on two red-billed choughs *Pyrrhocorax pyrrhocorax* in Cornwall: Habitat use and food intake'. *Bird Study* 37: 199–209.

Murchie, A.K. and Gordon, A.W. (2013). 'The impact of the "New Zealand" flatworm, *Arthurdendyus triangulatus*, on earthworm populations in the field'. *Biological Invasions* 15: 469–586.

Roy, S.S., Chauvenet, A.I.M. and Robertson, P.A. (2015). 'Removal of American mink (*Neovison vison*) from the Uists, Outer Hebrides, Scotland'. *Biological Invasions* 17: 2811–2820.

Schmidt, O. and Curry, J.P. (1999). 'Effects of earthworms on biomass production, nitrogen allocation and nitrogen transfer in wheat-clover intercropping model systems'. *Plant and Soil* 214: 187–198.

Sims, R.G. and Gerard, B.M. (1985). *Earthworms: Keys and Notes for the Identification and Study of Species. Synopss of the British Fauna.* London, UK and Leiden, The Netherlands: E.J. Brill.

Stop-Bowitz, C. (1968). 'A contribution to our knowledge of the systematics and zoogeography of Norwegian earthworms (Annelida Oligochaeta: Lumbricidae)'. *Norwegian Journal of Zoology* 17: 169–280.

Stockdill, M.J. (1982). 'Effects of introduced earthworms on the productivity of New Zealand pastures'. *Pedobiologia* 24: 29–35.

Wakeman, D. and Vickerman, K. (1979). 'The land planarian *Artioposthia triangulata*, a new species record for the West of Scotland'. *Glasgow Naturalist* 19: 499–501.

Management of an invasive avian parasitic fly in the Galapagos Islands: is biological control a viable option?

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Abstract The bird-parasitic fly, *Philornis downsi*, was first recorded in the Galápagos Islands in 1964 where it likely invaded from mainland Ecuador. This muscid fly is now the leading cause of recent declines in endemic landbird populations as its larvae feed on the nestlings of at least 19 bird species in the Galápagos, including many species of Darwin's finches. As yet, no long-term control method has been implemented for *P. downsi*, but importation (also known as classical) biological control may be a viable option. Due to historically high-profile examples of biological control agents attacking non-target species, some consider biological control to be too risky to be compatible with conservation aims. However, since biosafety practices were implemented beginning in the 1990s, these risks have been drastically reduced, and biological control is now an important tool for suppressing invasive species that are difficult to control using other means. We investigated the safety of a potential biological control agent, the parasitoid wasp, *Conura annulifera,* that attacks *P. downsi* in its native range. Here we summarise the results of a series of field, laboratory and comparative studies on *C. annulifera* (methods and results are not reported here) and outline future directions. We used a field experimental paradigm involving nest boxes baited with non-target hosts, and quarantine laboratory no-choice trials in which non-target hosts were exposed to *C. annulifera.* Our work to-date suggests that *C. annulifera* is restricted to attacking species within the genus *Philornis*. Furthermore, a phylogenetically controlled comparative study suggests that *C. annulifera* is evolutionarily constrained in its host range. These results lead us to conclude that *C. annulifera* demonstrates promise as an ecologically safe agent for the long-term biological control of *P. downsi.* Studies will now focus on an evaluation of risks to endemic and native species in the Galápagos.

Keywords: biological control, conservation, *Conura annulifera*, invasive species, *Philornis downsi*

THE HISTORY OF BIOLOGICAL CONTROL

When non-native species invade novel ecosystems, the consequences can be extreme. Anthropogenic change has increased the frequency and the success of such invasions; increased traffic to previously isolated sites increases the chance that non-native species will arrive, and the chances of colonisation are often increased in areas that are altered by human activity (Sax, et al., 2002). When colonisation occurs, non-native species are often far more successful than in their native range. Although the reasons for this are likely complex, one often-cited reason is the 'enemy release' hypothesis (ERH) (Liu & Stiling, 2006). Enemy release occurs when invaders colonise a new area, free from the natural enemies (predators, parasites and pathogens) with which they co-evolved, and are released from the effects that these enemies have on population suppression. Importation (also known as classical) biological control involves reconstructing (at least in part) the assemblage of co-evolved natural enemies present in the native range of the problematic species in order to control it (Heimpel & Mills, 2017). Although more commonly known from agricultural systems, importation biological control for conservation is a developing sub-discipline and shows promise as a long-term strategy for dealing with harmful invasive species (Van Driesche, et al., 2010; Van Driesche & Reardon, 2017). One noteworthy example is the introduction of the specialised ladybeetle, *Rodolia cardinalis,* which effectively controlled populations of the cottonycushion scale, *Icerya purchasi*, in the Galápagos archipelago. This particular introduction has more than likely been the saviour of endemic plant species that are attacked by *I. purchasi* (Hoddle, et al., 2013).

Unfortunately, historically high-profile cases of biological control failures that led to non-target effects on threatened species have received significant media attention and these examples have hampered progress in the sub-discipline of conservation-focused biological control (Van Driesche & Reardon, 2017). In order for importation biological control to be safe and successful, it is paramount that we understand the ecology, in particular the host specificity of the putative natural enemy set for release. A majority of examples of biological control, for conservation or for agriculture, have demonstrated both its success and its safety, particularly since the 1990s when concerns over biosafety gained momentum (Barratt, et al., 2010; Van Driesche, et al., 2010; Van Driesche, 2012; Heimpel & Mills, 2017; Van Driesche & Reardon, 2017; Heimpel & Cock 2018). However, the negative reputation of biological control persists due to the memorable nature of failures that have caused detrimental effects on native fauna (see Clarke, et al., 1984; Howarth 1991). Sadly, it is these examples that are more publicly well-known due to the strong emotions that they elicit (Van Driesche, & Reardon, 2017). Yet there is still hope for the discipline, and conservation-focused biological control has initiated a paradigm shift, demonstrating that biological control can be more than just compatible with conservation aims, it can actually promote them (Van Driesche, et al., 2010; Heimpel & Cock, 2018). In order for these techniques to be incorporated into the conservation 'tool box' it is imperative that we build trust between practitioners of biological control, conservationists and the public. To do so, the biological control community must demonstrate the pragmatism and caution that go into designing safe and effective biological control programmes.

Philornis downsi

Philornis downsi (Dodge & Aitken) is a bird-parasitic muscid fly that is native to mainland South America but is invasive in the Galápagos Islands where it likely invaded from mainland Ecuador (Bulgarella, et al., 2015). It was first reported in the archipelago in 1964 and, in the last 15–20 years has become a major threat to the persistence of many passerine bird species in the Galápagos, including

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the majority of species of Darwin's finches (Fessl, et al., 2018). This threat occurs because of the way the larvae of *P. downsi* feed: the adults are free-living but the larvae are obligate ectoparasitic blood-feeders on young nestlings, leading to blood loss and death (Fessl, et al., 2006; O'Connor, et al., 2010; Kleindorfer, et al., 2014; Koop, et al., 2016; Heimpel, et al., 2017). *P. downsi* is considered the greatest threat to the persistence of many land-bird species in Galápagos. The critically endangered mangrove finch (*Camarhynchus heliobates*) and medium tree finch (*C. pauper)* are particularly at risk, with any nestlings of the former now being protected by 'head-starting' (hand-rearing any eggs collected in the wild; Cunninghame, et al., 2012). The ramifications of any such extinction would be extensive. Not only would this represent a tragic loss of iconic species in a well-protected environment, it would also be a terrible loss of evolutionary history and the opportunity to study it. Rosemary and Peter Grant spent the last forty years studying the evolution of Darwin's finches and have commented on the importance and uniqueness of Darwin's finches for work of this nature:

'A final reason that makes them (Darwin's finches) *so suitable* (for studying evolution) *is that none of the species has become extinct as a result of human intervention. This cannot be said for many other radiations elsewhere in the world.'* Grant & Grant (2009)

The Grants' work has demonstrated the power of evolution and speciation and the underlying mechanisms, but much more remains to be discovered (e.g. Abzhanov, 2010 and articles therein). Losing even a single species of Darwin's finch would represent a terrible loss for evolutionary biology and could have a profound impact on future phenomena that the species' radiation may reveal to us. Moreover, losing large numbers of individuals of any of these species will have considerable impacts on the functioning of Galápagos ecosystems due to the critical roles that they play in pollination and seed dispersal (Causton, et al., 2013; Traveset, et al., 2015; Nogales, et al., 2017).

Need and potential for biological control of *Philornis downsi*

Infestation by *P. downsi* results in extreme nestling mortality in Galápagos, which has not been observed in the native range of the fly (Fessl, et al., 2018). The ERH (Liu & Stiling, 2006) – a paucity of co-evolved natural enemies in the invaded compared to the native range – is one likely reason for the increased abundance of *P. downsi* in Galápagos compared to the mainland (Bulgarella, et al., 2015; 2017; Boulton & Heimpel, 2017). The ERH serves as the theoretical underpinning of modern importation biological control, whereby one or a suite of co-evolved natural enemies is liberated into the invasive range to control the target species (Heimpel & Mills, 2017). The scarcity of natural enemies of *Philornis* spp. in the Galápagos compared to the mainland suggests that importation biological control may be a valuable tool to control *P. downsi* (Bulgarella, et al., 2017).

Although several control strategies are currently being explored and considered, importation biological control may be critical in protecting Darwin's finches and other endemic bird species in Galápagos from *P. downsi.* Other possible control methods include short-term strategies, such as nest treatment with insecticide and mass trapping using lures (Fessl, et al., 2018). The short-term approaches are considered mainly as stop-gap measures, whilst long-term measures, such as biological control and sterile male release, are developed and implemented. Of the long-term measures considered so far, biological control using natural enemies from the native range is currently the most promising solution. The release of sterile males is another potential long-term solution but this is currently hampered by difficulties in laboratory breeding of *P. downsi* (Lahuatte, et al., 2016; Fessl, et al., 2018).

In 2012, a workshop was organised by the Charles Darwin Foundation and the Galápagos National Park Directorate in order to form an action plan for conservation of Darwin's finches and other small land birds due to the ever-increasing threat from *P. downsi* (Causton, et al., 2013). One priority research goal recognised at this workshop was to identify natural enemies in the fly's native range and investigate the potential for biological control (Causton, et al., 2013). Over the last four years, we have discovered several parasitoid wasp species attacking species of *Philornis* in mainland Ecuador (Bulgarella, et al., 2015; 2017). Before any of these parasitoids can be considered as suitable biocontrol agents, in-depth studies of their host range need to be conducted. To address this question, we have been using a holistic approach consisting of a novel field experimental paradigm, comprehensive literature review, detailed study of the physiology and evolutionary ecology of the putative biological control agents, and traditional laboratory host range tests. In this manuscript we review and summarise our published work so far and outline future directions.

FIELD OBSERVATIONS AND EXPERIMENTS

Field work at two field sites in western mainland Ecuador between 2013 and 2017 has revealed a number of parasitoid species attacking *Philornis* spp. pupae collected from nest boxes (Bulgarella, et al., 2015; 2017). In addition, we have developed a novel field experimental paradigm over the last two years that can be used as a preliminary assay to test whether the parasitoid wasp species that we have recovered are specific to *Philornis* spp. in the field.

The experimental set-up was as follows. Nest boxes that we monitor throughout the bird breeding season for *P. downsi* pupae and their parasitoid wasps were paired with bait boxes. These bait boxes contained a number of non-target host species that had been reared from the local area. We also placed pupae of non-target species inside active bird nests. Any parasitoid wasp species that attacked *Philornis* spp. in the nest boxes and nests also had the opportunity to attack non-target hosts in the adjacent bait boxes and inside active nests. Using this experimental paradigm, we were able to determine which (if any) species of parasitoid wasp did not exclusively attack *Philornis* spp. We have concentrated our further efforts on *Conura annulifera* (Hymenoptera: Chalcididae), a parasitoid that has been recorded attacking only *Philornis* spp. in these field experiments. We will concentrate on this species for the remainder of the manuscript but note that we are also considering other species for biological control of *P. downsi*, such as an unidentified species of *Trichopria* (see Bulgarella, et al. (2017) and Boulton & Heimpel (2017) for details). This study is in progress at the time of writing and the results will be published elsewhere.

Life history and evolutionary ecology of *Conura annulifera*

Previous work on the natural host range of *C. annulifera* supports our assertion that it is a specialist on the genus *Philornis*. It has been recorded in previous studies throughout South and Central America where is has been reported as parasitising only *Philornis* spp. (including *P. downsi* and *P. deceptivus;* Burks, 1960; De Santis, 1979; Delvare, 1992; Couri, et al., 2006). Moreover, studies where pupae were reared from other Diptera (Muscidae, Calliphoridae and Sarcophagidae) in regions where *C. annulifera* has been reported never yielded this parasitoid

(Bulgarella, et al., 2017). However, only five studies have reported finding *C. annulifera* in the field (Burks, 1960; De Santis, 1979; Delvare, 1992; Couri, et al., 2006; Bulgarella, et al., 2017), and so more data were needed in order to determine whether this species might constitute a *Philornis*-specific biological control agent. In the sections below, we present the evidence that we have accumulated so far in support of the possibility that *C. annulfiera* is a *Philornis* genus specialist.

Conura annulifera is a solitary pupal ectoparasitoid (Bulgarella, et al., 2017). It attacks pupae of *Philornis* spp., laying a single egg on the outside of the developing pupa. More specifically, *C. annulifera* is a 'gap-layer', a parasitoid that deposits its egg between the hard external puparium and the soft body of the developing pupa. We hypothesised that the specificity of this oviposition site is likely to restrict the range of suitable hosts that *C. annulifera* can parasitise to the cyclorrhaphan Diptera, an unranked taxon that contains families such as the Muscidae, Calliphoridae, Sarcophagidae and Syrphidae (Griffiths, 1972; Boulton & Heimpel, 2017). The Cyclorrhapha are the only group of holometabolous insects that exhibit this gap (Whitten, 1957), and so it is unlikely that species outside this taxon are physiologically viable hosts for *C. annulifera*. We tested this possibility using phylogenetically controlled comparative studies for all known species of gap-layers in the superfamily Chalcidoidea and the results support our hypothesis: gap-laying species exhibit narrower host ranges than 'true' ectoparasitoids (Boulton & Heimpel, 2018). Moreover, these analyses revealed that gap-laying as a strategy may constitute an evolutionary dead-end. Compared to endoparasitoids and other ectoparasitoids, evolutionary transitions towards gap-laying were more likely than transitions away from it (Boulton & Heimpel, 2018).

This comparative work has implications for biological control in general and for the specific case of control outlined here. Our findings suggest that (1) gap-layers such as *C. annulifera* are likely to be more host specific, and so safer putative biological control agents, than 'true' ectoparasitoids, and (2) gap-layers including *C. annulifera* may represent particularly useful agents for importation biological control as they are less likely to transition, or diversify, to attack novel hosts after release outside their native range. With regards to the specific case of using *C. annulifera* to control *P. downsi* in the Galápagos, this work improves our understanding of the most at-risk non-target organisms were a release to be attempted, but it does not explicitly tell us whether *C. annulifera* is likely to be a safe species for importation biological control. To test this, more traditional host range studies were conducted, the results of which we outline in the section below.

Laboratory host range studies

Bulgarella, et al. (2017) exposed a range of non-target host pupae to *C. annulifera* that were maintained in the laboratory. This included five cyclorrhaphan Diptera (*Musca domestica, M. autumnalis, Stomoxys calcitrans* (Muscidae), *Sarcophaga bullata* (Sarcophagidae), *Calliphora vicina* (Calliphoridae)), three Lepidoptera (*Epiphyas postvittana* (Tortricidae), *Manduca sexta* (Sphingidae), *Plodia interpunctella* (Pyralidae)) and a hymenopteran (*Habrobracon hebetor* (Braconidae)). These species were chosen due to their likely physiological compatibility with parasitism by *C. annulifera* (Diptera) and because other species in the genus *Conura* have been shown to attack various Lepidoptera and Hymenoptera (see Bulgarella, et al., 2017).

In no case did the wasp produce offspring on any of these non-target species: in these lab studies, *C. annulifera* only reproduced successfully on *P. downsi*. This suggests that, of the species presented so far, only *P. downsi* represents a viable host for *C. annulifera* (Bulgarella, et al., 2017). However, this experimental design did not allow us to address the mechanism underlying this apparent specificity. It could either be that *C. annulifera* does not attempt to attack any species other than *Philornis* (i.e., behavioural specificity) or the wasp attempts to parasitise these species but their offspring fail to develop and emerge (i.e., only physiological specificity; see Desneux, et al., 2009). For an importation biological control programme with *C. annulifera* to be truly considered safe, it is important that we rule out the possibility that *C. annulifera* would attack non-target hosts, and cause their mortality by envenomation or oviposition.

To do this, we carried out additional analyses to test whether exposure to *C. annulifera* had any influence on the successful emergence of non-target pupae compared with controls. We found no evidence that exposure to *C. annulifera* resulted in elevated mortality for non-target hosts (see Bulgarella, et al., 2017). In contrast, when *P. downsi* pupae were exposed to the wasp, mortality increased independently of successful parasitism (i.e. more unparasitised fly pupae failed to emerge in the exposed treatment than in the control), perhaps as a result of host-feeding or envenomation/attempted parasitism. This finding, plus behavioural observations, suggests that *C. annulifera* does not attempt to sting or probe any potential host other than *Philornis* spp. pupae (Boulton & Heimpel, 2017; Bulgarella, et al., 2017).

FUTURE DIRECTIONS

Although all the evidence accumulated so far suggests that *C. annulifera* is a specialist parasitoid of *Philornis* spp. and should be seriously considered as a potential agent for the biological control of *P. downsi* in the Galápagos, one crucial question regarding the host range remains. It is critical to know whether *C. annulifera* is able to attack and develop on native or endemic non-target species present in the archipelago. As is common for most oceanic islands, the Galápagos exhibits high rates of endemism in insects (Peck, 1996). Island endemics may be particularly vulnerable to the introduction of a non-native parasitoid due to their lack of shared co-evolutionary history and the necessary adaptations to evade or resist parasitism. Before we can consider biological control in the Galápagos using any natural enemy, we must evaluate the host specificity of the putative biological control agent in the context under which it is intended for use. The studies that we have conducted using *C. annulifera* thus far represent a vital first step, suggesting that importation of *C. annulifera* into a quarantine facility in the Galápagos for further host range testing is justifiable. The results of these studies also allow us to narrow down the list of most at-risk non-target organisms in the Galápagos, due to the limitations imposed by its evolutionary and behavioural ecology.

Importation biological control of *P. downsi* in the Galápagos constitutes a promising means of population suppression that may ultimately serve to protect the extremely vulnerable bird species that the fly attacks (Boulton & Heimpel, 2017). Establishment of a biological control agent such as *C. annulifera*, may, in addition to ameliorating the current situation, serve as a preventative measure from future invasions of *P. downsi* and other bird parasitic species in the genus *Philornis* that are found in Ecuador. Preventative measures such as this may be deemed particularly judicious given the probability of further invasions under the high tourism pressure that the islands currently face (Toral-Granda, et al., 2017).

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REFERENCES

Abzhanov, A. (2010). 'Darwin's Galápagos finches in modern biology'. *Philosophical Transactions of the Royal Society of London B: Biology* 365: 1001–1007.

Barratt, B.I., Howarth, F.G., Withers, T.M., Kean, J.M. and Ridley, G.S. (2010). 'Progress in risk assessment for classical biological control'. *Biological Control* 52: 245–254.

Boulton, R.A. and Heimpel, G.E. (2017). 'Potential for Biological Control of a Parasite of Darwin's Finches'. In: R.G. Van Dreische and R.C. Reardon (eds.). *Supressing Over-abundant Invasive Plants and Insects in Natural Areas by Use of Their Specialized Natural Enemies*, pp. 23–28, USA: FHTET, USDA.

Boulton, R.A. and Heimpel, G.E. (2018). 'Mind the gap: The evolution of oviposition site and specialisation in the parasitoid superfamily Chalcidoidea'. *Biological Journal of the Linnean Society* 124: 213–227.

Bulgarella, M., Quiroga, M.A., Dregni, J.S., Cunninghame, F, Muñoz, D.A.M., Monje, L.D., Causton, C.E. and Heimpel, G.E. (2015). '*Philornis downsi* (Diptera: Muscidae), an avian nest parasite invasive to the Galápagos islands, in mainland Ecuador'. *Annals of the Entomological Society of America* 108: 242–250.

Bulgarella, M., Quiroga, M.A., Boulton, R.A., Ramirez, I., Moon, R.D., Causton, C.E. and Heimpel, G.E. (2017). 'Life cycle and host specificity of the parasitoid *Conura annulifera* (Hymenoptera: Chalcididae), a potential biological control agent of *Philornis downsi* (Diptera: Muscidae) in the Galápagos Islands'. *Annals of the Entomological Society of America* 110: 317–328.

Burks, B.D. (1960). 'A *Spilochalcis* parasitic on flies that infest birds' nests (Hymenoptera, Chalcididae)'. *Acta Hymenopterologica* 1: 95–97.

Causton, C.E., Cunninghame, F. and Tapia, W. (2013). 'Management of the avian parasite *Philornis downsi* in the Galápagos Islands: A collaborative and strategic action plan'. *Galápagos Report 2011–2012*: 167–173.

Clarke, B., Murray, J. and Johnson, M.S. (1984). 'The extinction of endemic species by a program of biological control'. *Pacific Science* 38: 97–104.

Couri, M.S., Tavares, M.T. and Stenzel, R.R. (2006). 'Parasitoidism of chalcidid wasps (Hymenoptera, Chalcididae) on *Philornis* sp. (Diptera, Muscidae)'. *Brazilian Journal of Biology* 66: 553–557.

Cunninghame, F., Switzer, R., Parks, B., Young, G., Carrión, A., Medranda, P. and Sevilla, C. (2013). 'Conserving the critically endangered mangrove finch: head-starting to increase population size'. *Galápagos Report* 2014*:* 151–157.

De Santis, L. (1979). *Catálogo de los Himénopteros Calcidoideos de América al sur de los Estados Unidos.* La Plata, Buenos Aires, Argentina: Publicación Especial de la Comisión de Investigaciones Científicas de la Provincia de Buenos Aires.

Delvare, G. (1992). 'A reclassification of the Chalcidini with a checklist of the New World species'. *Memoirs of the American Entomological Institute* 53: 315.

Desneux, N., Barta, R.J., Hoelmer, K.A., Hopper, K.R. and Heimpel, G.E. (2009). 'Multifaceted determinants of host specificity in an aphid parasitoid'. *Oecologia* 160: 387–398.

Fessl, B., Sinclair, B.J. and Kleindorfer, S. (2006). 'The life-cycle of *Philornis downsi* (Diptera: Muscidae) parasitizing Darwin's finches and its impacts on nestling survival'. *Parasitology* 133: 739–747.

Fessl, B., Heimpel, G.E. and Causton, C.E. (2018). 'Invasion of an Avian Nest Parasite, *Philornis downsi,* to the Galápagos Islands: Colonization History, Adaptations to Novel Ecosystems, and Conservation Challenges'. In: P.G. Parker (ed.) *Disease Ecology: Social and Ecological Interactions in the Galapagos Islands*, pp. 213–266., Dordrecht, The Netherlands: Springer.

Grant, B.R. and Grant, P.R. (2009). 'Darwin's Finches: Studying Evolution in Action'. In: T. de Roy (ed.) *Galápagos: Preserving Darwin's Legacy.* pp. 122–129. Auckland, New Zealand: David Bateman.

Griffiths, G.C.D. (1972). *The Phylogenetic Classification of Diptera Cyclorrhapha*. Dordrecht, The Netherlands: Springer.

Heimpel, G.E., Hillstrom, A., Freund, D., Knutie, S.A. and Clayton, D.H. (2017). 'Invasive parasites and the fate of Darwin's finches in the Galápagos Islands: The case of the vegetarian finch'. *Wilson Journal of Ornithology* 129: 345–349.

Heimpel, G.E. and Mills, N.J. (2017). *Biological Control: Ecology and Applications*. Cambridge, UK: Cambridge University Press.

Heimpel, G.E. and Cock, M.J.W. (2018). 'Shifting paradigms in the history of importation biological control'. *BioControl* 63: 27–37.

Hoddle, M.S., Ramírez, C.C., Hoddle, C.D., Loayza, J., Lincango, M.P., Van Driesche, R.G. and Causton, C.E. (2013). 'Post release evaluation of *Rodolia cardinalis* (Coleoptera: Coccinellidae) for control of *Icerya purchasi* (Hemiptera: Monophlebidae) in the Galápagos Islands'. *Biological Control* 67: 262–274.

Howarth, F.G. (1991). 'Environmental impacts of classical biological control'. *Annual Review of Entomology* 36: 485–509.

Kleindorfer, S., Peters, K.J., Custance, G., Dudaniec, R.Y. and O'Connor, J.A. (2014). 'Changes in *Philornis* infestation behavior threaten Darwin's finch survival'. *Current Zoology* 60: 542–550.

Koop, J.A., Kim, P.S., Knutie, S.A., Adler, F. and Clayton, D.H. (2016). 'An introduced parasitic fly may lead to local extinction of Darwin's finch populations'. *Journal of Applied Ecology* 53: 511–518.

Lahuatte, P.F., Lincango, M.P., Heimpel, G.E. and Causton, C.E., (2016). 'Rearing larvae of the avian nest parasite, *Philornis downsi* (Diptera: Muscidae), on chicken blood-based diet'. *Journal of Insect Science* 16(1): 84.

Liu, H. and Stiling, P. (2006). 'Testing the enemy release hypothesis: a review and meta-analysis'. *Biological Invasions* 8: 153–1545.

Nogales, M., Gonzalez-Castro, A., Rumeu, B., Traveset, A., Vargas, P., Jaramillo, P., Olesen, J.M. and Heleno, R.H. (2017). 'Contribution by vertebrates to seed dispersal effectiveness in the Galapagos Islands: a community-wide approach'. *Ecology* 98: 2049–2058.

O'Connor, J.A., Sulloway, F.J., Robertson, J. and Kleindorfer, S. (2010). '*Philornis downsi* parasitism is the primary cause of nestling mortality in the critically endangered Darwin's medium tree finch (*Camarhynchus pauper*)'. *Biodiversity Conservation* 19: 853–866.

Peck, S.B. (1996). 'Origin and Development of an Insect Fauna on a Remote Archipelago: The Galapagos Islands, Ecuador'. In: A. Keast and S.E. Miller (eds.) *The Origin and Evolution of Pacific Island Biotas, New Guinea to Eastern Polynesia: Patterns and Processes*, pp. 91–122. Amsterdam, The Netherlands: SPB Academic Publishers.

Sax, D.F., Gaines, S.D. and Brown, J.H. (2002). 'Species invasions exceed extinctions on islands worldwide: A comparative study of plants and birds'. *American Naturalist* 160: 766–783.

Toral-Granda, M.V., Causton, C.E., Jaeger, H., Trueman, M., Izurieta, J.C., Araujo, E., Cruz, M., Zander, K.K., Izurieta, A. and Garnett, S.T. (2017). 'Alien species pathways to the Galapagos Islands, Ecuador'. *PLOS ONE* 12: e0184379.

Traveset, A., Olesen, J.M., Nogales, M., Vargas, P., Jaramillo, P., Antolín, E., Trigo, M.M. and Heleno, R. (2015). 'Bird-flower visitation networks in the Galapagos unveil a widespread interaction release'. *Nature Communications* 6: Article Number 6376.

Van Driesche, R.G., Carruthers, R.I., Center, T., Hoddle, M.S., Hough-Goldstein, J., Morin, L., Smith, L., Wagner, D.L., Blossey, B., Brancatini, V. and Casagrande, R. (2010). 'Classical biological control for the protection of natural ecosystems'. *Biological Control Supplement* 1: S2–S33.

Van Driesche, R.G. (2012). 'The role of biological control in wildlands'. *BioControl* 57: 131–137.

Van Driesche, R.G. and Reardon, R.C. (2017). 'Supressing Over-abundant Invasive Plants and Insects in Natural Areas by Use of Their Specialized Natural Enemies'. USA: FHTET, USDA. <https://www.fs.fed.us/foresthealth/technology/pdfs/FHTET-2017-02_Biocontrol_Natural_Areas.pdf>.

Whitten, M.J. (1957). 'The supposed pre-pupa in cyclorrhaphous Diptera'. *Quarterly Journal of Microscopical Science* 98: 241–250.

Feasibility of eradicating the large white butterfly *(Pieris brassicae)* from New Zealand: data gathering to inform decisions about the feasibility of eradication

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Abstract *Pieris brassicae*, large white butterfly, was first found in New Zealand in Nelson in May 2010. The Ministry for Primary Industries (MPI) responded with a monitoring programme until November 2012 when the Department of Conservation (DOC) commenced an eradication programme. DOC was highly motivated to eradicate *P. brassicae* by the risk it posed to New Zealand endemic cress species, some of which are already nearly extinct. DOC eliminated the butterfly from Nelson in less than four years at a cost of ca. NZ$5 million. This is the first time globally that a butterfly has been purposefully eradicated. Variation in estimates of benefits, costs, the efficacy of detection and control tools, and the probability of eradication success all contributed to uncertainty about the feasibility. Cost benefit analyses can contribute to assessing feasibility but are prone to inaccurate assumptions when data are limited, and other feasibility questions are equally important in considering the best course of action. Uncertainty does not equate to risk and reducing uncertainty through data gathering can inform feasibility and decision making while increasing the probability of eradication success.

Keywords: biodiversity, cost-benefit analysis, eradication success, extinction risk, invasive species, non-native species

INTRODUCTION

Biological invasions by insects, including Lepidoptera, are increasing worldwide (Liebhold, et al., 2016; Suckling, et al., 2017). Insect invaders can cause significant biodiversity, economic, social and health impacts, which makes eradication an attractive management strategy (Liebhold, et al., 2016). Expanding international trade and travel have increased the numbers of exotic organisms entering New Zealand (Biosecurity Council, 2003; MPI, 2016).

Pieris brassicae (Lepidoptera: Pieridae), a Northern Hemisphere species native to Eurasia, was first found in the wild in New Zealand in Nelson (41°27′S, 173°28′E), a coastal city at the north of the South Island, in May 2010 (Toft, et al., 2012). An Unwanted Organism under the Biosecurity Act 1993, and a known pest of cultivated brassicas, it was referred to locally as 'great white butterfly' (GWB), or 'great white cabbage butterfly' (GWCB'). In this paper we use the scientific name *Pieris brassicae*.

Pieris brassicae can migrate hundreds of kilometres to new locations within a season (Spieth & Cordes, 2012). Together with the species' cold tolerance, its dispersal ability would put most New Zealand endemic brassica populations at risk (Kean & Phillips, 2013). However, the rate of *P. brassicae* spread in Nelson was uncertain. It was found at eight sites spread over 10–12 km in urban Nelson five months after the initial detection, but its distribution appeared not to have changed significantly after a further two years (Phillips, et al., 2016). This suggested unexpectedly slow dispersal for this species, perhaps impeded by parasitic wasps, predation or other factors.

DOC considered that *P. brassicae* had potential to cause extinctions of New Zealand endemic cresses, many of which occur in isolated, small populations; this makes them vulnerable to a wide range of threats and expensive to protect. New Zealand has 79 native cress species within the Brassicaceae family, most of them endemic and two already presumed extinct. Fifty-five species are currently threatened by extinction: 18 listed as nationally critical (the closest ranking to extinction), four nationally endangered, five nationally vulnerable, one declining, eight naturally uncommon, and 19 threatened though not yet ranked (Townsend, et al., 2008; de Lange, et al., 2013; S. Courtney, DOC, pers. comm.).

After mating, a *P. brassicae* female lays a cluster of 50–150 eggs on a host plant, and can lay a total of ca. 500 eggs (Gardiner, 1963; Spieth & Schwarzer, 2001). After hatching, larvae feed together and can wander up to 350 m in search of food plants. *Pieris brassicae* develop through five larval stages, usually defoliating several host plants in the process. Larvae at the fifth stage crawl away from their host plants to form pupae. The time required for *P. brassicae* to complete its lifecycle depends both on temperature and day length. It had two to three generations per year in Nelson (Kean & Phillips, 2013).

The Ministry for Primary Industries (MPI) is New Zealand's lead biosecurity agency with responsibilities to protect New Zealand's environment, economy, health and socio-cultural values under the Biosecurity Act 1993. MPI responded quickly to the 2010 detection of *P. brassicae* in Nelson by alerting the public and establishing a monitoring and surveillance programme. However, they terminated their response in November 2012 based on the results of the final of several cost benefit analyses (CBA) (Dustow, 2010; Dustow & van Eyndhoven, 2012; Manning, 2012). MPI predicted costs would outweigh benefits and that the probability of success was low (Manning, 2012).

The Department of Conservation (DOC) has a responsibility to protect native biodiversity under the Conservation Act 1987. DOC has a strong track record of pest management and successful eradication of (mostly) vertebrate pests from islands (Diamond, 1990; Simberloff, 2002; Howald, et al., 2007). On 19 November 2012, DOC initiated an eradication attempt to eliminate the risk that *P. brassicae* posed to New Zealand endemic cresses, primarily using systematic ground-based searching (Phillips, et al., 2016). The attempt succeeded: the last *P. brassicae* was captured near central Nelson on 16 December 2014, and the eradication programme closed on 4 June 2016. MPI and DOC declared *P. brassicae* eradicated from New Zealand on 23 November 2016, at a cost of NZ$4.97 million (€3.22 million).

DOC and MPI both had very pressing competing priorities and were acutely aware that spending money on an eradication attempt would take resources from other high priority work. To spend limited taxpayer dollars wisely, MPI responds to incursions according to carefully

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considered priorities using CBA to prioritise management responses. When considering eradication, MPI calculates a Benefit Cost Ratio (BCR) by estimating: the pest's impact over 20 years, the predicted cost of the eradication attempt, and the probability of eradication success. A BCR over 3:1 is required for MPI to initiate an eradication attempt.

Unfortunately, elements of a CBA can be difficult to quantify. Accurately predicting the impacts of invasive species can be difficult (Andersen, et al., 2004; Paterson, et al., 2015; Simberloff, et al., 2013; Simberloff, 2015). There is no universally accepted way of quantifying the benefit of conserving biodiversity in dollar terms (Spash, 2008; Parks & Gowdy, 2013; Barkowski, et al., 2015). Predicting eradication costs (Donlan & Wilcox, 2007; Holmes, et al., 2015) and the probability of eradication success also pose challenges, especially when there are few precedents and limited field data (Pluess, et al., 2012; Brown & Brown, 2015; Phillips, et al., this publication).

A feasibility study that considers eradication costs and benefits is a routinely used decision support tool in DOC (Broome, et al., 2005). Before starting the *P. brassicae* eradication attempt, DOC also considered costs, benefits and probability of success, though in a proposed eradication strategy rather than a CBA (Toft, et al., 2012). After commencing it, DOC revised costings, procured an independent CBA (East, 2013a) and developed additional feasibility criteria (Phillips, et al., this publication)

In this paper we explore uncertainties in the feasibility and economics of eradicating *P. brassicae* and suggest ways of reducing them to help inform future decision-making.

METHODS

We examined the question of when to attempt or abandon eradication when faced with high uncertainty and discuss ways to assist future decisions in such circumstances.

Cost Benefit Analysis

Four CBAs were developed, three by MPI and one by the University of New England for DOC. CBA is a systematic process for calculating and comparing the costs and benefits of a decision. Written as a formula it would read: (discounted benefits × probability of success)/ discounted costs. Costs and benefits were discounted at a rate of 8% for 20 years based on New Zealand Treasury advice (NZ Treasury, 2005).

The costs of aerial and ground-based eradication were predicted using known or estimated costs of service providers. Predictions also drew on experience with previous eradication operations regarding the activities required and their likely timeframes. Costs were included for active surveillance, passive surveillance (media, public), organism management (insecticide spraying, etc.), vegetation (host plant) movement controls, host plant removal and science support (developing a lure, augmenting natural enemy populations by releasing parasitic wasps, developing the sterile insect technique, data analysis, genetics and modelling).

The benefits of eradication are the avoided impacts. The impacts on brassica seed production, vegetable growing, and livestock forage production were calculated based on the cost of applying additional insecticide to control *P. brassicae*. These purely monetary impacts were estimated using several assumed rates of *P. brassicae* dispersal that were based on previous observations of *P. brassicae* spread in Chile (400 km in seven years), South Africa (350 km in two years) and Japan (400 km in five years or less) (Manning, 2012).

The biodiversity impacts (i.e. the cost of applying insecticide to endemic cresses to control *P. brassicae*) were considered by two CBAs (Dustow & van Eyndhoven, 2012; East, 2013a). In both analyses, 'willingness to pay' – a non-market valuation method which is based on a New Zealand community's willingness to avoid local extinction of a native plant – was also used to estimate biodiversity impacts (Dustow & van Eyndhoven, 2012; East, 2013a: East, 2013b). However, neither Dustow (2010) nor Manning (2012) used the cost of applying insecticide to endemic cresses for controlling *P. brassicae* in their 'willingness to pay' calculations.

Criteria used to evaluate eradication feasibility

Feasibility analysis aims to scope the size of the project, decide if eradication is possible and identify issues that require resolution to maximise the chance of eradication success (Pacific Invasives Initiative, 2013) and thereby estimate the probability of eradication success. MPI estimated the probability of success of ground-based eradication at approximately 30% based on overseas examples and expert opinion (Manning, 2012). The feasibility criteria used by MPI when considering eradication probability are based on Bomford & O'Brien (1995). They are:

- Rate of removal exceeds rate of increase at all population densities
- Immigration is zero
- All reproductive pests must be at risk
- Target pest can be detected at low densities
- Cost benefit analysis must favour eradication
- Suitable socio-political environment.

DOC assembled a Technical Advisory Group (TAG) to support the eradication attempt. The TAG developed a modified set of nine criteria, which built on the six criteria above, to evaluate feasibility (including the probability of success) and guide the eradication attempt (Phillips, et al., this publication). The criteria used by MPI are discussed below.

Technical advice and decision making

Both MPI and DOC used in-house and external expertise to inform decision making. DOC's TAG comprised three senior animal pest technical advisors from DOC including an entomologist, three senior scientists from two government research institutes (AgResearch and Plant and Food Research), and a private insect ecology consultancy (Entecol Ltd). DOC's TAG had considerable experience in ground-based eradication having advised or been directly involved in multiple animal and weed pest eradications nationally and worldwide. MPI consulted in-house technical staff, some of whom had been involved in previous insect eradication programmes, and also held a day-long meeting to consult with an external group of insect ecologists and industry stakeholders about the feasibility of eradicating *P. brassicae*. An MPI Governance Group reviewed the evidence provided by technical staff and decided not to attempt eradication in September 2012. DOC senior managers decided in November 2012 to attempt eradication based on the technical advice they received (Toft, et al., 2012).

RESULTS

The greatest variation between the four CBAs is in the predicted costs of eradication and discounted benefits (Table 1). The former due to differences in method and labour unit cost, and the latter due to the presence or absence of biodiversity benefit. Probability of success estimates were relatively similar, although the Manning CBA, which MPI ultimately used in their decision to abandon eradication, was somewhat less optimistic.

Eradication feasibility and cost benefit uncertainty

There was uncertainty about *P. brassicae*'s New Zealand distribution, reproductive rate, seasonality, rate of emigration, and host plant range. Similarly, it was difficult to predict the response of the public to control measures, efficacy of control measures, efficacy of detection methods, ability to monitor progress towards eradication, eradication costs, eradication benefits and probability of success.

Technical assessment of eradication feasibility criteria

Rate of removal exceeds rate of increase at all population densities

This was unknown at the outset given the potentially high reproductive capacity. No pheromones or other chemical attractants were available for *P. brassicae*, therefore trapping could not be used as a control tool, nor as a surveillance tool to monitor changes in population density. Aerial insecticide application was considered a potential method of maximising *P. brassicae* mortality at all population densities but was not pursued due to its likely unacceptability to Nelson residents (see criterion 6) and some uncertainty over just how vulnerable eggs and larvae would be to aerial spraying of large-leaved host plants. The large, conspicuous larvae feeding in groups on the same host plant did, however, suggest ground-based searching may be effective if the scale of operation could match the scale of infestation. Also, most *P. brassicae* host plants were likely to be low-growing, which would facilitate ground-based searching.

Immigration is zero

There was concern that the high dispersal potential of *P. brassicae* would make delimiting the population expensive and unreliable (given the unavailability of effective lures) and could result in undetected populations occurring outside the operational area that could reinvade. However, large commercial brassica crops on arable land near Nelson city were routinely monitored and by 2012 were still not showing evidence of *P. brassicae*.

All reproductive pests must be at risk

As described in more detail below, most potential control methods depended on visually detecting *P. brassicae*, but search efficacy was initially unknown. Thus, the possibility that some individuals would evade detection and avoid control was a major concern.

If *P. brassicae* populations occurred outside the operational area and remained undetected, those individuals would not be at risk, therefore violating criteria 2 and 3 above. *Pieris brassicae* adults are highly mobile and can cover long distances in search of larval food plants and nectar sources. Individuals are known to fly up to 5 km a day searching for host plants for egg-laying (Schutte, 1966, cited in Feltwell, 1982). Given the high dispersal potential of *P. brassicae* and the observed rapid spread of the closely related *P. rapae* when it appeared in New Zealand (at least 160–190 km within two years of detection) (Muggeridge, 1942), it was assumed *P. brassicae* would be widespread in Nelson and that undetected populations existed. It was considered that *P. brassicae* was capable of moving outside Nelson city's boundaries in the first season post-detection. There was also the risk that *P. brassicae* could escape Nelson in association with human transport, perhaps as larvae on infested vegetation or as pupae on inanimate objects including vehicles. However, despite the potential for rapid dispersal beyond Nelson, by 2012 there was still no evidence that it had occurred. Possibly dispersal was density-dependent (Toft, et al., 2012).

There was concern that wild brassicas and other food hosts in less accessible places would act as refugia if they could not be found and searched.

There was also concern that some life stages would not be susceptible to control. For example, eggs can occur under leaves making them difficult to see and less vulnerable to insecticide sprays. The cryptically coloured pupae can attach to man-made structures such as fences and it seemed they would often be difficult to find. However, every individual could be put at risk during one or more stages of its lifecycle through human search effort.

In addition, not all tools depended on people detecting *P. brassicae.* There was published evidence that eggs and larvae were vulnerable to storm events, and eggs, larvae and pupae would be susceptible to parasitism or predation by various species of parasitic wasps and paper wasps that were already present in New Zealand (Muggeridge, 1943; Bonnemaison, 1965; Gould & Jeanne, 1984; Richards, et al., 2016). Moreover, detection was not an essential prerequisite for applying control measures such as insecticides and destroying host plants (e.g. garden brassicas).

Target pest can be detected at low densities

There was concern that visually searching for *P. brassicae* without a lure would be costly, labour intensive and ineffective at detecting all individuals at low population densities. All previously successful eradications of Lepidoptera used pheromone lures (Tobin, et al., 2014). Pheromones can be used to detect and monitor populations, and also to disrupt mating, which can be a particularly effective control method at low pest densities. However, pheromones and other chemical attractants were unavailable for *P. brassicae*. Detection probabilities could be calculated but only through data gathering and analysis during an eradication attempt (Phillips, et al., 2014a).

Cost benefit analysis must favour eradication

Four separate CBAs were carried out, three before the eradication attempt commenced and one a year after the

Table 1 Eradication method, cost, benefit and probability of success.

Reference	Method	Discounted cost (NZ$ m)	Discounted benefit (NZ$ m)	Benefit: cost ratio	Probability of success (%)
Dustow (2010)	Aerial	25–73	21.7–60.9	0.3–2.44	50–75%
Dustow & van Eyndhoven (2012)	Aerial	25–73	21.7–123.2	0.3–4.93	50–65%
	Ground	13.3		1.64–9.28	
Manning (2012)	Ground	8.9	13.2–26.5	1.5–3	30–60%
East (2013a)	Ground	3.9	17.4–70.8	4.8–19.7	56–76%

eradication attempt started. All used different estimates of costs, benefits and probability of success and, therefore, all obtained different BCRs and reached different conclusions (Table 1).

Dustow (2010) concluded that "the analysis strongly suggests that it is not economically beneficial to attempt to eradicate great white butterfly [using the aerial application of insecticide]". Dustow & van Eyndhoven (2012) concluded that "the CBA analysis indicates favourable benefit cost ratios for all but the most conservative ground-based eradication when biodiversity values are excluded", and "relatively low biodiversity values are required to generate favourable benefit cost ratios for many scenarios". Manning (2012) concluded that "given the level of uncertainty surrounding the development of effective control tools, low probabilities of successfully eradicating the GWCB, and the uncertainty surrounding biodiversity benefits, it is unlikely to be technically or economically feasible to eradicate the GWCB". Subsequently, East (2013a) concluded that "The high expected impacts of the GWB on New Zealand's native brassicas, the agricultural industry and home gardeners result in high net present values and benefit cost ratios [which suggests] that a GWB eradication programme in Nelson is warranted".

Manning (2012) stated that 'the ground-based eradication option is considered to have a probability of success of approximately 30% based on overseas examples and expert opinion". MPI used the probability of 30% when decision making. The probability of success value (mean 56%; range 50–60%) used in the fourth CBA (East 2013a) a year after eradication commenced was determined by DOC's TAG who had the benefit of some hard data on which to make their estimate.

Cost estimates varied greatly among the four CBAs (Table 1). Aerial spraying costs were based on previous experience of using this method against white tussock moth (*Orgyia leucostigma*) and painted apple moth (*Orgyia anartoides*) in Auckland (Ashcroft, et al., 2010) and assumed substantial social mitigation costs for affected residents of Nelson. Ground-based cost estimates were little more than guesses given uncertainty around method efficacy, delimitation boundaries and detection probabilities (which strongly influence the length of time ground crews must remain operational beyond the last detection to have confidence in declaring eradication success). MPI contractor costs were also estimated at three times higher than DOC staff costs. Again, East (2013a) had some actual data to work with and consequently her cost estimate came closer than the others to the final actual cost.

Suitable socio-political environment

An aerial application of the bio-pesticide bacterium Btk (*Bacillus thuringiensis kurstaki*) was thought likely to raise considerable public opposition as it did in Auckland for white tussock moth and painted apple moth (Ashcroft, et al., 2010). Ground-based control, on the other hand, was assumed likely to gain public and political support. This was evidenced shortly after the initial detection by positive public responses to official requests for reports of *P. brassicae* sightings.

DISCUSSION

When assessing the feasibility of eradication, three basic questions must be answered (Broome, et al., 2005): Why do it? Can it be done? What will it take to succeed?

Why attempt eradication?

It was impossible to precisely predict the impact of *P. brassicae* on New Zealand endemic brassicas (and predicting impacts on cultivated brassicas under different management regimes was also problematic). New Zealand's biodiversity has been geographically isolated for millions of years from Northern Hemisphere plants, herbivores, predators and parasitoids, which makes it hard to predict impacts. This is a generic issue for incursion response management in New Zealand. If the New Zealand native plants that a non-native herbivore will feed on cannot be immediately identified, then estimating impacts can only be achieved either through difficult, expensive and imperfect laboratory testing, or by watching them unfold in the wild. Laboratory testing of the suitability of threatened native cresses for herbivory by *P. brassicae* was impractical as most are not cultivated due either to the difficulty of obtaining seed, or to their complex cultivation requirements.

The risk of extinction to endemic cresses from herbivory by *P. brassicae* was considered significant even without multiple other threats. Other threats to endemic cresses include herbivory and disturbance by a range of pests, viral and fungus attack, weed competition, sea-level rise and the loss of seabird-driven ecosystem processes which all impact on different cresses. As Quammen (1996) pointed out, extinction often results from multiple causes and "to be rare is to have a lower threshold of collective catastrophe".

Preventing extinction of native biodiversity is a core function of DOC and is fundamental to the Department's legislative mandate (Conservation Act 1987). Given the multiple threats facing endemic cresses in addition to *P. brassicae*'s potential to access all endemic brassicas, dietary preference for brassicas, tendency to deposit large numbers of eggs on individual plants and voracious feeding on individual plants by clusters of caterpillars, DOC's senior botanists and entomologist concluded there was a high risk that *P. brassicae* would drive at least some New Zealand endemic cresses to extinction. Knowledge of this risk strongly motivated DOC to attempt eradication, despite the uncertainties, while using a 'learn by doing' approach.

Can it be done?

The value attributed to the probability of success can significantly influence the benefit value obtained (i.e. benefit = discounted benefit × probability of success) and therefore the BCR. Estimating the probability of success is a subjective process based on evidence from previous eradication attempts and expert opinion. This becomes problematic when eradication of the taxon in question has not been attempted before, and where factors including the ecology, physiology and behaviour of the non-native species in the new environment are poorly understood. Accurately estimating the probability of eradication success is impossible without knowledge of the effectiveness of control tools, pest population rate of increase, pest distribution, and risk of immigration (Bomford & O'Brien, 1995; Tobin, et al., 2014; Phillips, et al., this publication). The challenge is to gather enough quality data quickly enough to inform decisions. Choosing a threshold of certainty – where there is enough information to make a decision – can be partly based on an assessment of the consequences of not deciding. As Harvey Cox (1968) puts it "not to decide is to decide".

If the pest can be killed at the same time as it is being surveyed for distribution and abundance, then eradication may gain a 'head start' while critical feasibility information is being collected. Pre-defined stopping rules can be used to trigger reassessments of feasibility, thus limiting the risk of over-investing in eradication attempts that cannot succeed. For example, the DOC TAG defined the following triggers for re-evaluating the *P. brassicae* eradication attempt (Phillips, et al., 2014b):

- If established *P. brassicae* populations are detected outside the residential Nelson operational area
- If the population has expanded outwards after 12 months of being subjected to control
- If *P. brassicae* has not been eradicated by 30 June 2015
- If no *P. brassicae* have been detected for two consecutive years.

Triggers clearly indicate when the objectives in the plan are or are not being achieved. The initial response gathered some information about *P. brassicae*'s distribution prior to commencing the eradication attempt, but not about its rate of increase or the efficacy of visual searching. Once the attempt was underway, however, distribution data and the effectiveness of control tools was gathered in a systematic way that was used to inform management decisions, reduce uncertainty, reassess feasibility through time, measure progress and eventually provide confidence that eradication had been achieved (Phillips, et al., 2016).

What will it take to succeed?

The 'learn by doing' approach informed the technical assessment of the probability of success (described above). It also allowed the level of resourcing and capability that was needed for the eradication to succeed to be accurately quantified and adjusted as the programme progressed. For example, in the early stages of the programme in 2012, the ground control team was limited to a team of four. However, by April 2013 it had become clear that, although the methods might be effective, more resources were needed to achieve success (Table 2). The field team size was increased to 10 (and up to 30 later in the programme) and the consequent increased costs were factored into the final CBA (East, 2013a). By constantly reassessing resource allocations to different aspects of the project, efficiencies were gained without jeopardising the probability of success. Crucial in this decision making was expert analyses of incoming data by DOC's TAG that supported the project.

CONCLUSIONS

Delays in attempting eradication can increase the programme's duration, cost and risk of failure.

Table 2 Sites searched, sites infested with *Pieris brassicae* and proportion infested by financial year (July to June).

Year	Sites searched	Sites infested	Proportion of sites infested
2009–10	3	3	1
2010–11	88	30	0.341
2011–12	76	71	0.934
2012–13	23,923	1,121	0.047
2013–14	80,263	1,490	0.019
2014–15	83,118	170	0.002
2015–16	76,507	0	0
Total	263,978	2,885	0.015

Quick, proactive responses can help to achieve eradication while simultaneously gathering data to inform decision making. Stopping rules can be used to assess if an eradication should cease to minimise the waste of resources.

In the absence of reliable information about costs, biodiversity benefits and probability of success, CBAs should not be relied on as the sole decision making tool.

A TAG can be a powerful tool for providing ongoing well-structured advice to assess feasibility and assist eradication decision making.

Close engagement with research agencies facilitates research support for eradication attempts, which can help to provide critical analyses, information and management tools.

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REFERENCES

Andersen, M. C., Adams, H., Hope, B., & Powell, M. (2004). 'Risk assessment for invasive species'. *Risk analysis* 24(4), 787–793.

Ashcroft, T., Richardson, E., Voice, D. and Dustow, K. (2010). *Strategies and Feasibility to Undertake Eradication of Pieris brassicae (GWCB) in New Zealand*. Unpublished report. Wellington: Ministry for Primary Industries.

Barkowski, B., Lienhoop, N. and Hansjurgens, B. (2015). 'Capturing the complexity of biodiversity: A critical review of economic valuation studies of biodiversity'. *Ecological Economics* 113: 1–14.

Biosecurity Council, (2003). *Tiakina Aotearoa: Protect New Zealand: The Biosecurity Strategy for New Zealand.* Wellington: Ministry of Agriculture and Forestry.

Bomford, M. and O'Brien, P. (1995). 'Eradication or control for vertebrate pests?'. *Wildlife Society Bulletin* 23: 249–255.

Bonnemaison, L. (1965). 'Insect pests of crucifers and their control'. *Annual Review of Entomology* 10: 233–256.

Broome, K.G., Cromarty, P. and Cox, A. (2005). 'Rat Eradications – How to Get it Right Without a Recipe'. *Proceedings of the 13th Australasian Vertebrate Pest Conference, Te Papa Wellington, New Zealand, 2–6 May 2005*, pp. 152–157. Lincoln, NZ: Manaaki Whenua – Landcare Research.

Brown, K. and Brown, D. (2015). *Control to Eradication of* Tradescantia fluminensis *on Stephens Island (Takapourewa): The Importance of Systematic and Persistent Effort*. DOC Research and Development Series 346. Wellington: Department of Conservation.

Cox, H. (1968). *On not Leaving it to the Snake.* UK: SCM Press.

de Lange, P.J., Rolfe, J.R., Champion, P.D., Courtney, S.P., Heenan, P.B., Barkla, J.W., Cameron, E.K., Norton, D.A. and Hitchmough, R.A. (2013). *Conservation Status of New Zealand Indigenous Vascular Plants, 2012.* New Zealand Threat Classification Series 3. Wellington: Department of Conservation.

Diamond, J.M. (1990). 'New Zealand as an Archipelago: An International Perspective'. In: D.R. Towns, C.H. Daugherty and I.A.E. Atkinson, (eds.) *Ecological restoration of New Zealand Islands*, pp. 3-8. Conservation Sciences Publication No. 2. Wellington: Department of Conservation.

Donlan, C.J. and Wilcox, C. (2007). 'Complexities of costing eradications'. *Animal Conservation* 10: 154–156.

Dustow, K. (2010). *Great White Cabbage Butterfly Cost Benefit Analysis*. Unpublished report. Wellington: Ministry for Primary Industries.

Dustow, K. and van Eyndhoven, E. (2012). *Great White Cabbage Butterfly Cost Benefit Analysis*. Unpublished report. Wellington: Ministry for Primary Industries.

East, M. (2013a). *A Benefit Cost Analysis for the Great White Butterfly Incursion in New Zealand. Final report to Department of Conservation*. Armidale, NSW: Institute of Rural Futures, University of New England.

East, M. (2013b). *Impact of the Great White Butterfly (*Pieris brassicae*) on New Zealand's Native Cress Species. Final report to Department of Conservation*. Armidale, NSW: Institute of Rural Futures, University of New England.

Feltwell, J. (1982). *Large White Butterfly the Biology, Biochemistry and Physiology of* Pieris brassicae *(Linnaeus).* The Hague-Boston-London: Dr W. Junk Publishers.

Gardiner, B. (1963). 'Genetic and environmental variation in *Pieris brassicae*'. *Journal of Research on the Lepidoptera* 2: 127–136.

Gould, W.P. and Jeanne, R.L. (1984). '*Polistes* wasps (Hymenoptera: Vespidae) as control agents for lepidopterous cabbage pests. *Environmental Entomology* 13: 150–156.

Holmes, N.D., Campbell, K.J., Keitt, B.S., Griffiths, R., Beek, J., Donlan, C.J. and Broome, K.G. (2015). 'Reporting costs for invasive vertebrate eradications'. *Biological Invasions.* 17: 2913–2925.

Howald, G.C., Donlan, J., Galvan, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y. Veitch, C.R., Genovesi, P., Pascal, M., Saunders, A., Tershey, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Kean, J. and Phillips, C. (2013). *Phenology and Diapause Research for Great White Butterfly (*Pieris brassicae*). Milestone 4: Detailed Spreadsheet Model for Phenology (Report for Ministry for Primary Industries No. RE400/2013/481).* MPI 16341. Lincoln: AgResearch.

Liebhold, A. M., Berec, L., Brockerhoff, E. G., Epanchin-Niell, R. S., Hastings, A., Herms, D. A., & Yamanaka, T. (2016). 'Eradication of invading insect populations: from concepts to applications'. *Annual Review of Entomology* 61: 335–352.

Manning, R. (2012). *Biosecurity Response Cost Benefit Analysis: Great White Cabbage Butterfly.* Unpublished report. Wellington: Ministry for Primary Industries.

MPI. (2016). *Biosecurity 2025 Direction Statement for New Zealand's biosecurity system*. Wellington.

Muggeridge, J. (1942). 'The white butterfly (*Pieris rapae* L.). Its establishment, spread and control in New Zealand'. *New Zealand Journal of Science and Technology* 24 (A): 107–129.

NZ Treasury. (2005). *Cost Benefit Analysis Primer*. Wellington: New Zealand Government.

Pacific Invasives Initiative. (2013). *Resource kit for rodent and cat eradication*. <http://www.pacificinvasivesinitiative.org/rk/intro/index.html>.

Parks, S. and Gowdy, J. (2013). 'What have economists learned about valuing nature? A review essay'. *Ecosystem Services* 3: e1–e10.

Paterson, R.A., Dick, J.T.A., Pritchard, D.W., Ennis, M., Hatcher, M.J., and Dunn, A.M. (2015). 'Predicting invasive species impacts: a community module function response approach reveals context dependencies'. *Journal of Animal Ecology.* 84: 453–463.

Phillips, C., Kean, J., and van Koten, C. (2014a). *Detection Rates of* Pieris brassicae *Eggs and Larvae (No. RE400/2014/589*). Unpublished report. Lincoln: AgResearch.

Phillips, C.B., Brown, K., Green, C., Walker, G., Broome, K., Toft, R., Vander Lee, B., Shepard, M., Bayly, S. and Rees, J. (2014b). Pieris brassicae *(Great White Butterfly) Eradication Annual Report 2013/14*. Unpublished report. Wellington: Department of Conservation.

Phillips, C.B., Brown, K., Green, C., Broome, K., Toft, R., Walker, G., Shepard, M., Murphy, J. and Rees, J. (2016). Pieris brassicae *(Great White Butterfly) Eradication Annual Report 2015/16*. Unpublished report. Wellington: Department of Conservation.

Phillips, C.B., Brown, K., Broome, K., Green, C., and Walker, G. (this publication). 'Criteria to Help Evaluate and Guide Attempts to Eradicate Terrestrial Arthropod Pests'.

Pluess, T.; Cannon, R.; Jarošík, V.; Pergl, J.; Pyšek, P.; Bacher, S. (2012). 'When are eradication campaigns successful? A test of common assumptions'. *Biological Invasions* 14: 1365–1378.

Quammen, D. (1996). *The Song of the Dodo: Island Biogeography in an Age of Extinction*. UK: Pimlico.

Richards, N.R., Hardwick, S., Toft, R., and Phillips, C.B. (2016). 'Enhancing parasitism of *Pieris brassicae* pupae by *Pteromalus puparum*'. *New Zealand Plant Protection* 69: 126–132.

Simberloff, D. (2002). 'Today Tiritiri Matangi, tomorrow the world! are we aiming too low in invasives control? Keynote Address'. In: C.R. Veitch, and M.N. Clout, (eds.) *Turning the tide: the eradication of invasive species*, pp. 4–12. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN Gland, Switzerland and Cambridge, UK.

Simberloff, D. (2015). 'Non-native invasive species and novel ecosystems'. *F1000Prime Reports* 7: 47.

Simberloff, D., Martin, J., Genovesi, P., Maris, V., Wardle, D.A., Aronson, J., Courchamp, F., Galil, B., Garcia-Berthou, E., Pascal, M., Pysek, P., Sousa, R., Tabacchi, E. and Vila, M. (2013). 'Impacts of biological invasions: what's what and the way forward'. *Trends in Ecology and Evolution.* 28: 58–66.

Spash, C. L. (2008). 'How much is that ecosystem in the window? The one with the bio-diverse tail'. *Environmental Values* 17: 259–284.

Spieth, H.R. and Schwarzer, E. (2001). 'Aestivation in *Pieris brassicae* (Lepidoptera: Pieridae): Implications for parasitism'. *European Journal of Entomology* 98: 171–176.

Spieth, H.R. and Cordes, R. (2012). 'Geographic comparison of seasonal migration events of the large white butterfly, *Pieris brassicae*'. *Ecological Entomology* 37: 439–445.

Suckling, D. M., Conlong, D. E., Carpenter, J. E., Bloem, K. A., Rendon, P., and Vreysen, M. J. B. (2017). 'Global range expansion of pest Lepidoptera requires socially acceptable solutions'. *Biological Invasions* 19(4), 1107–1119.

Tobin, P.C., Kean, J.M., Suckling, D.M., McCullough, D.G., Herms, D.A. and Stringer, L.D. (2014). 'Determinants of successful arthropod eradication programs'. *Biological Invasions* 16: 401–414.

Toft, R., Brown, K., Courtney, S. and Green, C. (2012). *Proposed Eradication Strategy for Great White Cabbage Butterfly* (Pieris brassicae). Unpublished report Wellington: Department of Conservation.

Townsend, A.J., de Lange, P.J., Duffy, C.A.J., Miskelly, C.M., Molloy, J. and Norton, D.A. (2008). *New Zealand Threat Classification System Manual*. Wellington: Department of Conservation. <http://www.doc.govt.nz/Documents/science-and-technical/sap244.pdf>.

Effort required to confirm eradication of an Argentine ant invasion: Tiritiri Matangi Island, New Zealand

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Abstract Tiritiri Matangi Island in the Hauraki Gulf, Auckland, New Zealand is a 220 ha restoration island managed by the Department of Conservation as an open sanctuary. Following eradication of the only mammalian predator, the Pacific rat (*Rattus exulans*) in 1993, a variety of threatened birds, lizards and a giant invertebrate have been transferred to the island. In March 2000, Argentine ant (*Linepithema humile*) (Hymenoptera: Formicidae) was discovered and delimiting surveys revealed a 10 ha infestation. Managers were concerned that the ant could have significant negative impacts on invertebrates, birds and lizards. Early surveys confirmed a dramatic decline in all other ant species within the infested area. In February 2001, an eradication programme commenced with paste baits (a.i. 0.01% fipronil) spread manually in a 2 m × 3 m grid over the entire area. The second year employed a 1 m × 3 m spacing. A second incursion part way through the programme extended the area to 11 ha. The same toxic bait was used throughout the programme to kill residual colonies and a non-toxic version was used as a lure to intensively monitor progress. Eradication was declared in 2016. Critical parts of the programme included detection of post treatment survivors and the level of effort required to confirm successful eradication. New treatment techniques were developed to kill the last small nests by placing toxic baits inside vials on the ground to prolong bait life. Such nests exhibited non-invasive behaviour, short foraging distances, and were prone to disturbance leading to foraging cessation. Bait densities and field placement were critical to success. Sites with residual nests were deemed free of Argentine ant once there had been no detections over three consecutive years of ongoing monitoring. With few successful Argentine ant eradications in the world the techniques used here can inform and improve success rates for other ant eradication attempts.

Keywords: *Linepithema humile*, monitoring vials, paste bait, surveillance, toxic baiting

INTRODUCTION

Argentine ant (*Linepithema humile*) is one of the world's worst invasive ant species and an important conservation concern (Holway, et al., 2002) with considerable negative impacts to native biodiversity (Rowles & O'Dowd, 2007; Stringer, et al., 2009). Argentine ant infestations have proven difficult to eradicate with few reports of successful programmes (Silverman & Brightwell, 2008; Hoffmann, et al., 2011; Hoffman, et al., 2016). To date, only around 10% of ant eradications have been greater than 10 ha (Hoffmann, et al., 2016). Detectability of ants in low densities is one of the most critical factors to increase the likelihood of successful eradication (Hoffmann, 2011). Despite a long history of invasive ant management, utilising widely varying approaches, eradication failures are common (Hoffmann, et al., 2016).

A variety of techniques are used to sample Argentine ant such as visual searching, baits placed on the substrate, in vials or in pitfall traps (Stanley, et al., 2008; Casellas, et al., 2009). However, visual detection is less effective in more complex vegetated environments (Ward & Stanley 2013), such as on offshore islands that act as conservation sanctuaries, compared to urban areas. A study by Ward & Stanley (2103) of the detection probability of an Argentine ant population, using vials with honey and sausage meat, found that a site should be surveyed three times to be confident about the presence or absence of ants. Pest eradication programmes on islands are generally considered successful if no detections are found during two years of post-treatment monitoring (Howald, et al., 2007).

Argentine ant was first detected in New Zealand in 1990 (Green, 1990), and subsequently on Tiritiri Matangi Island in March 2000 (Harris, 2002). Following eradication of the only mammalian predator, the Pacific rat (*Rattus exulans*) from Tiritiri Matangi in 1993, a variety of threatened birds, lizards and a giant invertebrate were transferred to the island (Galbraith & Cooper, 2013). Managers were concerned that the ant could have considerable negative impacts on invertebrates (Sanders, et al., 2003), birds (Sockman, 1997; Suarez, et al., 2005) and lizards (Suarez & Case, 2002) through direct predation and competition for invertebrate food sources. Modelling has predicted that sites near Auckland, including Tiritiri Matangi, are hot spots for potential Argentine ant occupancy (Pitt, et al., 2009) with consequent implications for island biosecurity programmes. Two infestations were found on the island, one large area covering ca. 10 ha centred around the wharf and a second, smaller (<0.5 ha) area at Northeast Bay at the northern end of the island (Fig. 1). The latter arose

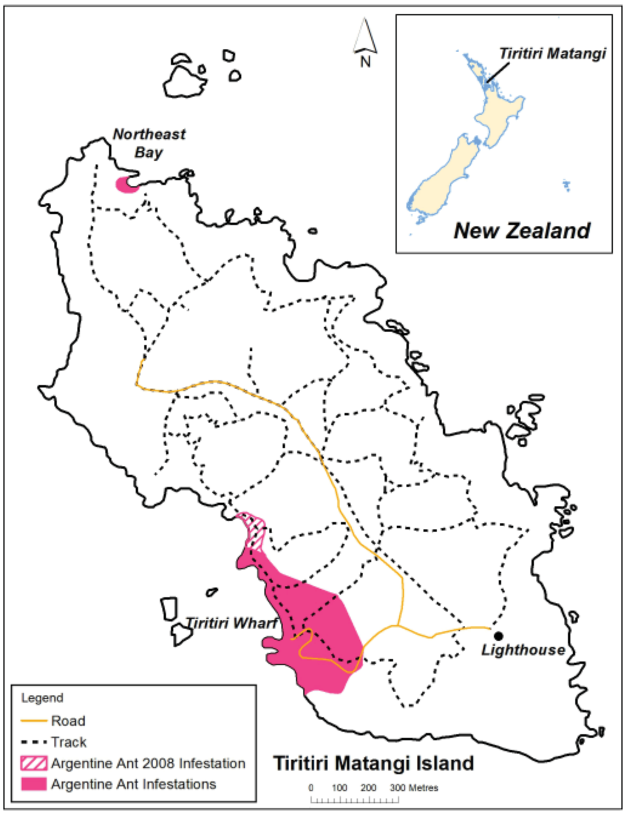

Fig. 1 Tiritiri Matangi Island Argentine ant infestations, tracks and roads.

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from the movement of an infested dingy from the wharf area (Harris, 2002). During the eradication programme a second incursion occurred at Hobbs Beach in 2008 at the northern end of the Tiritiri wharf infestation (Fig. 1). The shape of the newly infested ca. 0.5 ha area indicated its likely origin as the south end of Hobbs Beach. This new incursion extended the total infested area on Tiritiri to 11 ha.

Here I outline the programme against Argentine ant on Tiritiri Matangi Island and describe the effort required to confirm eradication.

METHODS

Study area

Tiritiri Matangi Island (Fig. 1) is a 220 ha Scientific Reserve 28 km north of Auckland City in the Hauraki Gulf, New Zealand, managed by the Department of Conservation as an open sanctuary. There are over 38,000 visitors to the island annually (DOC, 2014) with most arriving via commercial ferry to the wharf but the public are free to land on the beaches via private craft. The Supporters of Tiritiri Matangi and the ferry operator facilitate biosecurity measures relating to clean footwear and public awareness messaging on pests including Argentine ant. All freight and goods for island management arrive at the wharf and pest detection operates via inspections for all species, including Argentine ants, plus traps or indicator baits for rodents. The nearest land is the Whangaparaoa Peninsula 3.5 km to the west. The island is low lying and has been the subject of an extensive restoration programme involving the planting of over 280,000 trees over ten years from 1984–1994 (Galbraith & Cooper, 2013). The two areas infested with Argentine ants, Northeast Bay and around the Tiritiri wharf, host a range of plants characteristic of coastal habitats in the region including flax (*Phormium tenax*) (Fig. 2), karamu (*Coprosma robusta*), taupata (*Coprosma repens*), mahoe (*Melicytus ramiflorus*) and the coastal vine pohuehue (*Meulenbeckia complexa*). Typically, the canopy height was up to 6 m with the occasional pohutukawa tree (*Metrosideros excelsa*) exceeding 10 m.

Toxic baiting

Following the discovery of Argentine ant on the island in 2000, the infestation was delimited using visual assessment. A ca. 20 m buffer was added to the boundary of the entire infested area. Bait treatment during the first two years consisted of a single application of Xstinguish™ Argentine ant bait (a.i. 0.01% fipronil) over the 11 ha infestation. The paste baits (2–3 g) were hand laid using a caulking gun to extrude baits on the ground in a grid over the entire area with 2 m × 3 m and 1 m × 3 m spacing in February 2001 and the following season in December 2001, respectively. Where possible, baits were placed under vegetation to avoid exposure to the sun and reduce desiccation. From 2003, all remnant infestations were treated twice a year with toxic bait, four to eight weeks apart. The 2008 incursion (Fig. 1) was double-treated in 2009 with 1 m × 3 m spacing. From 2010, treatment consisted of toxic baits placed inside vials (25 mm × 50 mm) on the ground for five days, repeated two weeks later. Vials were spaced 1 m apart out to 5 m from the remnant colony then extruded baits on the ground out a further 5 m. Vials had netting covers to prevent lizards and larger invertebrates entering. Vials were placed in shade beneath vegetation to reduce desiccation.

Fig. 2 Flax (*Phormium tenax*) (foreground) on beach edge as a typical preferred habitat for Argentine ants on Tiritiri Matangi Island.

All baiting operations were carried out when the ground was dry and weather conditions were warm (air temperature 20–25 ºC) dry, and no rain forecast for at least 24 hours. These conditions were optimal for Argentine ant activity on Tiritiri and thus maximised the chances of bait detection.

Post-treatment monitoring

Intensive post-treatment monitoring commenced from 2003 using ca. 2 g of non-toxic Xstinguish™ Argentine ant monitoring paste lure in vials placed every 2–5 m in a grid over the target area. Although some visual detection was possible for larger remnant infestations during 2003, this was largely ineffective at detecting small infestations. Thus, from 2004 all monitoring used the lure in vials as above. During 2003, baits were left out for approximately four hours. From 2004, this was extended to 24 hours. During collection, the open vials were sealed with a lid and all trapped ants were identified and later verified using a microscope.

During the 16-year programme, the entire treated area was only intensively monitored on two occasions, in 2006 and 2008. Following the 2006 monitoring the whole previously infested 10 ha wharf area was assessed for sites that appeared to be preferred by Argentine ant. Due to limited resources, during years other than 2006 and 2008 varying levels of monitoring focused on these preferred Argentine ant sites. In addition, all detection sites from 2003 onwards were intensively monitored. Due to the initial very high densities of ants and the ongoing survival of a few nests, sites close to the wharf were monitored every year for all 16 years of the programme. Sites where nests remained undetected in alternate years were monitored two or three occasions per year to increase the likelihood of detection.

All lure operations were carried out under the same environmental conditions as described above for baiting, i.e. warm and dry.

RESULTS

Toxic baiting was extremely effective at reducing Argentine ant numbers to very low levels (Fig. 3). No Argentine ants were seen at Northeast Bay after 2001. However, remnant populations persisted at the larger wharf infestation after the initial single treatment per year. Thus, from 2003 toxic treatments were applied on two occasions each year, with a period between treatments sufficient to allow surviving ants to regroup into functioning nests, with foraging ants susceptible to being attracted to baits. This

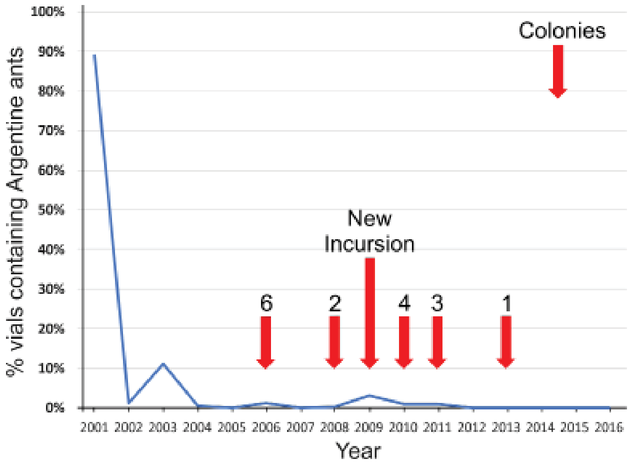

Fig. 3 Percentage of vials with Argentine ants during the 16-year programme. There were zero detections in 2007, 2012, 2014–16.

strategy effectively reduced the infestation to very small colonies, each consisting of a few nests, and sometimes just a single nest.

Argentine ants were detected visually for the first three years of the programme. However, for the remainder of the programme surviving nests could only be detected by using lured vials. This was largely due to the complex nature of the vegetated habitat and the small size of the remnant colonies. As the programme continued, fewer nests were detected (Fig. 3).

The entire treated area was intensively monitored in 2006 and revealed six infestations. Following treatment, no Argentine ants were detected in 2007. Intensive monitoring of the entire treated area again in 2008 detected two residual colonies. However, during these years in the middle of the programme, some small colonies remained undetected in some years. There were at least two sites within the wharf infestation where ants appeared to survive after the 2006 and 2008 treatments as they were re-detected one or two years later in very similar locations. One colony was not detected until 2011, despite the site being monitored annually since 2003. This site was within 5 m of the vehicle and trailer used to transport all arriving baggage and freight to buildings at the top of the island. All these surviving colonies vanished after toxic baited vials were introduced in 2010. Bait inside the vials generally remained moist and palatable to ants for the full five days. No Argentine ants were detected after 2014.

The 2008 incursion (Fig. 1) was discovered and double-treated in 2009. Three surviving colonies were detected in 2010 and two in 2011, with at least one of these being a survivor from 2010. No Argentine ants were detected following treatment in 2011.

The 2006 assessment for preferred Argentine ant sites revealed sites typically characterised by a warm northerly aspect where sun could reach the ground during much of the day. Vegetation was less than 3 m tall and usually had open areas within or adjoining, such as roads, tracks, the coast or exposed banks with just ground cover vegetation. Flax plants (Fig. 2) were often a feature of preferred sites although not a prerequisite.

When Argentine ant was first discovered on the island, the population density was very high close to the wharf, which was assumed to be the entry point. Some of the most problematic nests to destroy were located at sites near the wharf. Thus, these were monitored on two or three occasions per year from 2014. However, this repeated monitoring did not yield any additional detections.

Repeated intensive use of lured vials detected surviving nests. On some occasions, ants were detected in consecutive vials on adjacent lines indicating a larger population, likely to be more than one nest. However, detections were predominantly made in a single vial reflecting the presence of a single nest. Some of these remnant nests appeared to be very small as trails featured few ants and vials contained less than 10 Argentine ants when collected. Much of the lure was still present indicating a lack of substantial feeding activity over the 24 hours. In contrast, lure monitoring early in the programme when large colonies were detected yielded hundreds of ants with little lure remaining after four hours.

DISCUSSION

A single application of toxic bait was not sufficient to eradicate Argentine ant from Tiritiri Matangi Island (Harris, 2002). Although the bait was successful at quickly eradicating the small, recent infestation at Northeast Bay the larger wharf infestation required many years of intensive baiting of small remnant nests to achieve eradication. Increased levels of effort were required throughout the programme to improve both ant detection and treatment techniques to eliminate nests.

Early in the programme when Argentine ant first established on Tiritiri the species' behaviour fitted the usual pattern of being extremely competitive with other ant species for food sources (Human & Gordon, 1999). Foraging Argentine ants recruited to any new food source, including the toxic bait and non-toxic lure, in very large numbers, often within minutes. This behaviour contrasts with that of foraging ants from small, post-treatment, remnant nests that were not necessarily attracted to bait or the lure given the availability of other natural food sources. Detectability of ants in low densities is one of the most critical factors to increase the likelihood of successful eradication (Hoffmann, 2011).

In the latter stages of the programme there were occasions when small nests were detected but not seen again despite there being no toxic treatment in that area during that season. All single nests appeared to be lacking the "invasive" element in their behaviour and were observed foraging over short distances. It is possible that these ants lacked competitiveness to survive with other ant species (Rice & Silverman, 2013). All ants in monitoring vials were identified during the programme and some ant species were in high numbers. Several of those recorded, including *Monomorium antarcticum*, a New Zealand endemic, and *Ochetellus glaber*, a naturalised Australian species, have been shown to be competitive with Argentine ant (Westermann, et al., 2014).

The lack of competitiveness and aggressive behaviour normally seen in invasive species made detection of remnant Argentine ant nests more difficult. It is often true that the last remaining few in an eradication attempt require the greatest effort (Morrison, et al., 2007). As the programme continued, it was necessary to prolong the time that lured vials were available to foraging ants. While a four-hour monitoring period was adequate to measure the level of Argentine ant activity when ants exhibited invasive behaviour, it became clear later that even 24 hours was not adequate so needed to be repeated, as recommended by Ward & Stanley (2013). For the most preferred sites, particularly on coastal banks exposed to the sun most of the day, 24-hour monitoring was repeated three times per season for three seasons to verify eradication.

Ants on trails from small remnant nests often appeared uninterested in lures or baits even when placed next to the trail, despite their known palatability as seen early in the programme. Argentine ants prefer liquid or paste baits/lures (Nyamukondiwa & Addison, 2014), but the disadvantage of such baits/lures is that they have a very short field-life once applied in the environment. This is especially the case when used in warm/hot conditions which are optimal for ant activity. The life of the paste baits when placed on the ground was short (<12 hrs: Harris, 2002), which gave a limited time for ants to be attracted to them and commence feeding. Baits had to compete with other natural food sources for the attention of ants. In addition, on several occasions foraging ceased if the trail was disturbed while placing baits on the ground. This may have contributed to nest survival at some baited sites.

To increase the time of interaction between ants and toxin, baits can be delivered repeatedly through the season, as on Santa Cruz Island (Boser, et al., 2014, Boser, et al., 2017), or the life of each bait can be extended after application by slowing desiccation. Baits placed in vials and shaded under vegetation retained moisture and remained palatable for at least five days. Hoffman, et al. (2001; 2016) highlighted the need for new techniques to eradicate invasive ants. The innovation of placing toxic baits in vials reported here allowed the potential interaction between ants and toxic baits to occur over five days rather than 12 hours. Once toxic baited vials were deployed at detection sites, no further Argentine ants were seen at these sites and eradication was achieved.

During the programme there were at least two sites within the wharf infestation where ants apparently survived the 2006 and 2008 treatments as they were re-detected one or two years later in very similar locations. It is possible that the toxic baiting had sub-lethal effects on either Argentine ants and/or other ant species, such as *M. antarcticum* (Barbieri, et al., 2013), leading to changed interspecific dynamics and subsequent survival of Argentine ant nests. It is also possible that a surviving Argentine ant nest moved away from the monitored area and was not detected until it moved back in a subsequent season. All ant species readily move their nests if disturbed and this was observed with Argentine ants. Trails from surviving small nests were particularly prone to disturbance. The two sites in the wharf infestation were on the edge of Wharf Road in highly preferred locations. They could have moved away from the edge into less preferred locations beyond monitoring lines due to disturbance but returned to the edge and were detected in subsequent seasons.

Toxic vials were used only around the immediate vicinity of remnant nests to restrict the non-target impacts on other invertebrates. Relatively few vials had all the bait removed over the five days. In contrast, non-toxic lured vials used for monitoring often had much of the bait removed by non-target species over just 24 hours. Therefore, it was not worthwhile to leave the monitoring vials out longer than 24 hours.

Since the eradication of Argentine ants from Tiritiri Matangi, the island's biosecurity procedures have altered to include annual surveillance for any new incursions. This study has confirmed that ants from new, expanding populations are readily attracted to baits (Ward & Stanley, 2013), and the level of surveillance monitoring can be less intensive compared to that required to confirm post treatment eradication. Early detection of new incursions through surveillance programmes gives a greater chance of successful eradication (Clout & Williams, 2009; Ward, et al., 2010).

There are very few reported, successful Argentine ant eradications (Silverman & Brightwell, 2008, Hoffmann, et al., 2011, Hoffmann, et al., 2016). The successful Argentine ant eradication programme reported here required considerable effort and improved techniques to achieve eradication. It took 13 years to extirpate the last ants from the main infested area near the wharf, which had areas of very high population density. Problematic remnant nests were mostly found in these high-density areas. With the new monitoring and surveillance techniques developed here, there is confidence that if a new incursion is detected that eradication will be possible within a much shorter timeframe, as demonstrated by the 2008 Hobbs Beach incursion site which took only three years. These techniques would be readily applicable to discrete Argentine ant populations infesting 10 ha or less elsewhere in the world, thus achieving an increased success rate of eradication attempts.

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REFERENCES

Barbieri, R.F., Lester, P.J., Miller, A.S. and Ryan, K.G. (2013). 'A neurotoxic pesticide changes the outcome of aggressive interactions between native and invasive ants'. *Proceedings of the Royal Society B: Biological Sciences*, 280(1772): 20132157.

Boser, C.L., Hanna, C., Faulkner, K.R., Cory, C., Randall, J.M. and Morrison S.A. (2014). 'Argentine ant management in conservation areas: results of a pilot study'. *Monographs of the Western North American Naturalist* 7: 518–530.

Boser, C.L., Hanna, C., Holway, D.A., Faulkner, K.R., Naughton, I., Merrill, K., Randall, J.M., Cory, C., Choe, D.-H., and Morrison S.A. (2017). 'Protocols for Argentine ant eradication in conservation areas'. *Journal of Applied Entomology* 141: 540–550.

Casellas, D., Gomez, C. and Clavero, M. (2009). 'Comparing methods of evaluating the spread of Argentine ants in natural habitats: Pitfall traps vs. baiting'. *Sociobiology* 53(3): 927–938.

Clout, M.N. and Williams, P.A. (2009). *Invasive Species Management: A Handbook of Principles and Techniques*. Oxford: Oxford University Press.

DOC. (2014). CMS: *Conservation Management Strategy: Auckland 2014–2024*. Volume 1. Wellington, New Zealand: Department of Conservation.

Galbraith, M. and Cooper, H. (2013). 'Tiritiri Matangi – an overview of 25 years of ecological restoration'. *New Zealand Journal of Ecology* 37(3): 258–260.

Green, O.R. (1990). 'Entomologist sets new record at Mt. Smart, or *Iridomyrmex humilis* established in New Zealand'. *Weta* 13:14–16.

Harris, R.J. (2002). 'Potential Impact of the Argentine Ant (*Linepithema humile*) in New Zealand and Options for its Control'. *Science for Conservation* 196. Wellington, New Zealand: Department of Conservation.

Hoffmann, B., Davis, P., Gott, K., Jennings, C., Joe, S., Krushelnycky, P., Miller, R., Webb, G. and Widmer, M. (2011). 'Improving ant eradications: details of more successes, a global synthesis and recommendations'. *Aliens: The Invasive Species Bulletin* 31: 16–23.

Hoffmann, B.D. (2011). 'Eradication of populations of an invasive ant in northern Australia: successes, failures and lessons for management'. *Biodiversity and Conservation* 20(13): 3267–3278.

Hoffmann, B.D., Luque, G.M., Bellard, C., Holmes, N.D. and Donlan, C.J. (2016). 'Improving invasive ant eradication as a conservation tool: A review'. *Biological Conservation* 198: 37–49.

Holway, D.A., Lach, L., Suarez, A.V., Tsutsui, N.D. and Case, T.J. (2002). 'The causes and consequences of ant invasions'. *Annual Review of Ecology and Systematics* 33(1): 181–233.

Human, K.G. and Gordon, D.M. (1999). 'Behavioural interactions of the invasive Argentine ant with native ant species'. *Insectes Sociaux* 46: 159–163.

Howald, G., Donlan, C.J., Galvan, J.P., Russell, J.C., Parks, J., Samaniego, A., Wang, Y., Veitch, D. Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21(5): 1258–1268.

Morrison, S.A., Macdonald, N., Walker, K., Lozier, L. and Shaw, M.R. (2007). 'Facing the dilemma at eradication's end: uncertainty of absence and the Lazarus effect. *Frontiers in Ecology and the Environment* 5: 271–276.

Nyamukondiwa, C. and Addison, P. (2014). 'Food preference and foraging activity of ants : Recommendations for field applications of low-toxicity baits'. *Journal of Insect Science* 14: 14–48.

Pitt, J.P.W., Worner, S.P. and Suarez, A.V. (2009). 'Predicting Argentine ant spread over the heterogeneous landscape using a spatially explicit stochastic model'. *Ecological Applications* 19(5): 1176–1186.

Rice, E.S. and Silverman, J. (2013). 'Propagule pressure and climate contribute to the displacement of *Linepithema humile* by *Pachycondyla chinensis*'. *PLOS ONE* 8(2): e56281.

Rowles, A.D. and O'Dowd, D.J. (2007).'Interference competition by Argentine ants displace native ants: implications for biotic resistance to invasion'. *Biological Invasions* 9: 73–85.

Sanders, N.J., Gotelli, N.J., Heller, N.E. and Gordon, D.M. (2003). 'Community disassembly by an invasive ant species'. *Proceedings of the National Academy of Sciences* 100: 2474–2477.

Silverman, J. and Brightwell, R.J. (2008). 'The Argentine ant: Challenges in managing an invasive unicolonial pest'. *Annual Review of Entomology* 53: 231–252.

Sockman, K.W. (1997). 'Variation in life-history traits and nest-site selection affects risk of nest predation in the California gnatcatcher'. *The Auk* 114(3): 324–332.

Stanley, M., Ward, D., Harris, R., Arnold, G., Toft, R. and Rees, J. (2008). 'Optimizing pitfall sampling for the detection of Argentine ants, *Linepithema humile* (Hymenoptera: Formicidae)'. *Sociobiology* 51(2): 461–472.

Stringer, L.D., Stephens, A.E.A., Suckling, D.M. and Charles, J.G. (2009). 'Ant dominance in urban areas'. *Urban Ecosystems* 12(4): 503–514.

Suarez, A.V. and Case, T.J. (2002). 'Bottom-up effects on persistence of a specialist predator: ant invasions and horned lizards'. *Ecological Applications* 12(1): 291–298.

Suarez, A.V., Yeh, P. and Case, T.J. (2005). 'Impacts of Argentine ants on avian nesting success'. *Insectes Sociaux* 52: 378–382.

Ward, D.F., Green, C., Harris, R.J. Hartley, S., Lester, P.J., Stanley, M.C., Suckling, D.M. and Toft R.J. (2010). 'Twenty years of Argentine ants in New Zealand : past research and future priorities for applied management'. *New Zealand Entomologist* 33: 68–78.

Ward, D.F. and Stanley, M.C. (2013). 'Site occupancy and detection probability of Argentine ant populations'. *Journal of Applied Entomology* 137(3): 197–203.

Westermann, F.L., Suckling, D.M. and Lester, P.J. (2014). 'Disruption of foraging by a dominant invasive species to decrease its competitive ability'. *PLOS ONE* 9(3): e90173.

Analysis of the secondary nest of the yellow-legged hornet found in the Balearic Islands reveals its high adaptability to Mediterranean isolated ecosystems

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Abstract The yellow-legged hornet (*Vespa velutina*) was detected for the first time in the north of Spain in 2010, but was not detected in Majorca, Balearic Islands until 2015 and only one secondary nest, with 10 combs, was found in the northwest of the island. During 2016, nine more nests were found in the same region. To better understand the biology of *V. velutina* in isolated conditions, the following objectives were proposed: (I) describe the architecture and structure of nests; (II) analyse the shape of combs and develop a new method to confirm the circular pattern of breeding; (III) determine the colony size and (IV) determine the succession of workers and sexual individuals throughout the season. For these reasons, nests that were removed were frozen for at least 48 days until analysis. Our results show that this species has a high reproductive potential under isolated conditions. Results reveal that parameters such as weight, height, diameter, number of cells and total individual production are directly related. Moreover, each mature nest can produce up to 9,000 individuals and several hundred potential founder queens. All results inform formulation of an efficient control or eradication programme in the Balearic Islands, as we are in the early stages of invasion and intervention is essential to eradicate *V. velutina* on Majorca Island.

Keywords: architecture, breeding provision, caste differentiation, individual production, Latter's formula, Majorca, *Vespa velutina nigrithorax*

INTRODUCTION

The yellow-legged hornet (*Vespa velutina* Lepeletier 1836) is a social Hymenopteran of the family Vespidae. It is native to tropical and subtropical areas of Southeast Asia (Archer, 1994; Martin, 1995; Carpenter & Kojima, 1997). It was reported for the first time in south-west France in 2004 (Haxaire, et al., 2006; Rome, et al., 2009; Villemant, et al., 2011) and rapidly spread to nearby European countries: Spain (Castro & Pagola-Carte, 2010; López, et al., 2011), Portugal (Grosso-Silva & Maia, 2012), Italy (Demichelis, et al., 2014), Belgium (Bruneau, 2011; Rome, et al., 2013) and Germany (Witt, 2015). This species is also established in South Korea (Choi, et al., 2012; Choi, et al., 2013) and Japan (Ueno, 2014). The most recent incursion was in Great Britain in 2016, and Switzerland in 2017 (UK National Bee Unit, 2016; Budge, et al., 2017).

The introduction of *V. velutina* to Europe could lead to important economic and ecological impacts. The main impact of the yellow-legged hornet is the likely decrease in honeybee (*Apis mellifera*) populations (Tan, et al., 2007; Monceau, et al., 2013a; Monceau, et al., 2013b), as wasp larvae feed on the proteins of honeybees. Honeybees are considered one of the most important pollinators for agriculture, so the decrease of *A. mellifera* populations is anticipated to decrease the production of their crops resulting in economic losses for the farmers (Villemant, et al., 2011; Arca, et al., 2014). In addition, it is possible that the yellow-legged hornets attack humans when colonial nests are established in urban areas (Villemant, et al., 2006). In the particular case of Majorca, a yellow-legged hornet invasion could be devastating for the populations of honeybees, the fragility of the ecosystem (typical of the island ecosystems) and the impact on endemic insects.

The life cycle of *Vespa velutina* is annual. In optimal ambient conditions, when the temperature is high and the food resources are abundant, one founder queen will build an embryo nest (Edwards, 1980; Archer, 2010), after that the workers begin to emerge. In spring the workers build combs around the embryo nest; this is called the primary nest. The primary nest has an irregular structure with the embryo nest in the centre (Spradbery, 1973). During summer, the colony increases and the primary nest is left and another nest is built in the same location, if the conditions are favourable (food resources, temperature, humidity, etc.). If the ambient conditions are unfavourable (cold conditions and limited food resources), they build the secondary nest in a different location, normally in large trees. This new nest is named the secondary nest and is larger than the primary nest, with the objective that the colony increases. The nests of this invasive species are classified as a calyptodomus type (concealed nest) (Fig. 1), having an external spherical structure, but the combs are of a conical structure. The upper combs have large diameters and the lower ones smaller diameters, with a slight narrowing in the last comb (Jeanne, 1975). When the reproductive caste emerges in autumn, the nest is

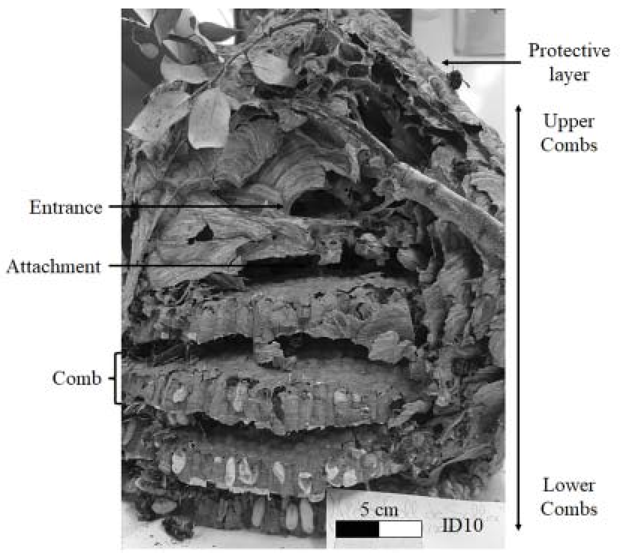

Fig. 1 Calyptodomus nest of *V. velutina*.

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called mature, because it is possible to separate males from females by their morphology (Choi, et al., 2012; Rome, et al., 2015). The male hornets fertilise new founder queens, after which the colony dies during the winter. Only new founder queens survive the winter and build new nests the following season and start the annual life cycle (Edwards, 1980; Matsuura & Yamane, 1990).

The yellow-legged hornet is established in the northern regions of Spain (Navarra, Basque Country, Galicia and Cantabria) (Castro & Pagola-Carte, 2010; López, et al., 2011), and in Catalonia (Pujade-Villar, et al., 2012). In 2015 it was reported in Soller (Majorca, Balearic Islands). The hornet was detected by a beekeeper and was identified by the laboratory of Zoology of the University of Balearic Islands. Together with the local authorities, an intensive survey was implemented to detect nests, as is described in Leza, et al. (2017). In 2015 only one nest of *V. velutina* was found in the north-west of the island. However, during 2016 nine more nests were found in the same region. At this moment, the invasion is in its early stages (Leza, et al., 2017), and is the first incursion on an island where eradication through locating and destroying nests can be used to control the spread; a scenario very different to mainland Europe. This immediate intervention plays an important role in the invasion or eradicating the species on the island.

Although the general structure and production of the nests of this species has been previously described in Asia (Spradbery & Kirk, 1978; Matsuura, 1991) and Europe (Rome, et al., 2011), it is important to study the nests in local conditions in order to find out if the adaptation of *V. velutina* is similar to other regions or if they would be unable to breed on the island. For this purpose, the detailed study of the nests found on Majorca (3,667 km^2, situated 176 km from the mainland) can be a useful tool to understand if this invasive species has the same biotic fitness or if they have some problems adapting in an island context. The results could help plan future surveys and possible dedicated control or eradication measures. Therefore, the study's goal was to better understand the biology of *V. velutina* in isolated conditions. For this reason, the following objectives were proposed: (I) describe the architecture and structure of nests; (II) analyse the shape of combs and develop a new method to confirm the circular pattern of breeding; (III) determine the colony size and (IV) determine the succession of workers and sexual individuals throughout the season.

MATERIAL AND METHODS

Nest collection

Nine nests were located from August to November 2016, after an active search for nests using the triangulation method (Leza, et al., 2017). All nests were entirely removed and frozenfor a minimum of 48 hours. Nests were kept frozen at -25°C until dissection.

All nests were located in the ''Serra de Tramuntana'', in the north-west of Majorca (the exact location is shown in Table 1). This region has a meso-Mediterranean climate (Emberger classification), where there is more precipitation than in other parts of the island (mean of 1,400–1,600 mm per annum) and cooler temperatures.

Architecture and structure of nests

External morphology of nests was analysed and described. Weight, height and maximum diameter of each nest was measured and the number of combs was recorded. Total weight was the result of the weight of the structure and its individuals and the total height corresponds to the height of the whole nest with the external envelope.

Shape of combs

In order to check the circular organisation of the combs described in other species of wasps (Spradbery & Kirk, 1978; Matsuura, 1991), a new method was proposed. It follows a similar methodology of comparison between two sequences of DNA (Brudno, et al., 2004). In our study the sequences were the diameters of combs and the nucleotides were the different brood stages, as follows: every developmental stage (empty cell, egg, larvae, prepupae, pupae and teneral adult) in cells across three diameters in each comb was analysed and compared with each other (the first with the second, the second with the third and the first with the third) (Figs. 2 & 3).

Each diameter comparison received an arbitrary categorisation: "2" was assigned if the stage was the same in cells at the same distance from the breeding centre, "1" if one of the stages was before or after the other stage (for example: in one diameter it is a larva and the other a

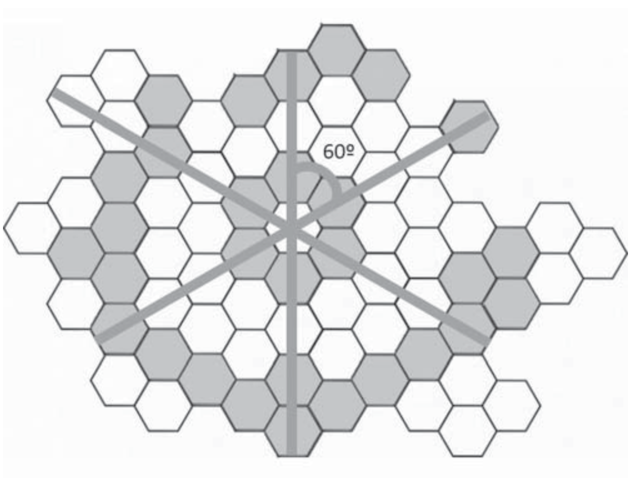

Fig. 2 Diagram of a nest comb. The grey cells represent operculated cells and the lines indicate the three diameters, which pass through the breeding centre and separated by 60º, where the stages of the individuals are determined.

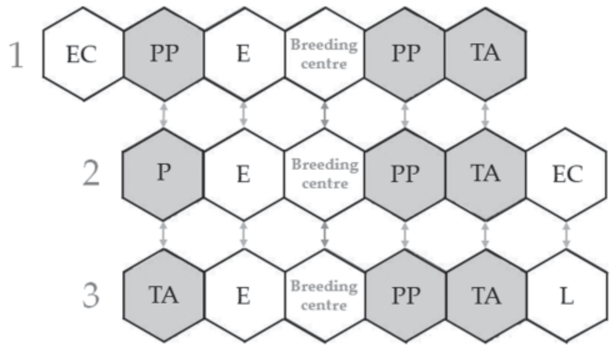

Fig. 3 Example of the three diameters, aligned in the breeding centre, and comparing the development stages at the same distance from the centre. EC = Empty cell, E = Egg, L = Larva, PP = Prepupa, P = Pupa and TA = Teneral adult.

Table 1 Characteristics of the nine nests. L = number obtained by Latter's formula. GP = individual production.

		ID1	ID2	ID3	ID4	ID5	ID6	ID7	ID8	ID9
UTM		39°45'26.33"N 2°42›37.04»E	39°46'56.30"N 2°45›40.72»E	39°46'0.80"N 2°41'4.30"E	39°46'5.80"N 2°44'25.70"E	39°44'17.29"N 2°38'7.82"E	39°45'36.7"N 2°39›14.3»E	39°47'36.5"N 2°46›02.6»E	39°47'28.0"N 2°46'03.5"E	39°44'14.3"N 2°41'29.5"E
Date removed		23/08/2016	31/08/2016	8/09/2016	13/09/2016	14/09/2016	21/09/2016	17/10/2016	2/11/2016	24/11/2016
No. of combs		6	5	5	5	2	6	7	5	9
Weight (g)		5,090.0	2,884.12	3,882.6	2,073.7	136.3	3,313.0	2,893.4	1,448.9	3,583.3
Height (cm)		49.3	30.6	41.5	14.4	7.0	18.3	32.6	18.6	47.6
Max diameter (cm)		33.0	25.5	24.6	24.6	12.0	36.0	23.0	21.2	38.0
No. of females	Workers	1338	789	285	615	127	684	610	351	151
	Error interval	4	0	1	0	0	2	40	15	316
	Founders	0	0	1	0	0	0	0	0	180
	Total	1350	810	293	624	127	701	653	367	656
Males		0	0	0	0	0	436	0	134	429
No. of adults		1,350	810	293	624	127	1,137	653	501	1,085
L. cells		6,197.75	2,712.0	3,077.25	2,778.75	200.0	7,008.25	2,916.25	2,412.75	9,355.0
L. eggs		131.25	196.5	16.5	203.0	1.25	155.5	1.25	74.25	40.25
L. larvae		1,055.0	774.25	947.25	1,066.75	176.25	563.75	809.0	36.2	440.5
L. meconium		2,074.0	796.25	1,202.0	1,186.75	0.0	3,370.75	1,239.25	1,629.75	7,750.25
GP		4,610.25	2,577.0	2,458.75	3,080.5	304.5	5,227.0	2,702.5	2,241.2	9,316.0
GP/cell		0.74	0.95	0.80	1.11	1.52	0.75	0.93	0.93	1.0

prepupa) and "0" if it does not coincide with any of the previous cases, as long as both cells have some stage or are empty (for example: in one diameter it is an egg and the other a pupa) If one diameter had more cells than the other, those cells in a diameter that did not have their partner in the other would not receive any value. For each diameter comparison the sum of each arbitrary punctuation was divided by the number of cells multiplied by “2”, the maximum arbitrary punctuation, obtaining a coincidence percentage with the circular organisation.

Colony size

The number of cells was estimated with Latter’s formula (Latter, 1935): $N = (3n/2 + 1) \cdot n/2$, where N is the total number of cells in one comb and n is the number of cells counted across its maximum diameter. This formula was extrapolated to estimate the number of eggs, immature stages (larvae and pupae) and meconium pellets (meconium is the gut content eliminated immediately by an individual when moulting from larval to pupal instars and was recorded only as presence or absence, indicating that at least one individual had bred). The number of adults was counted manually.

The estimated total individual production of a nest was defined as the sum of the estimated number of eggs, immature stages and meconium pellets, estimated with Latter’s formula, and adults.

Pearson’s and Spearman’s rank correlations were made between the estimated individual production and the following variables: number of combs, weight, height, diameter, cells, eggs, immature stages or meconium pellets.

Sexual and caste differentiation

Females and males were distinguished by morphological differences (apex of last sternite bilobate in male but sharp in female). For females, founders and workers were distinguished based on their wet weight. Below 593.09 mg individuals can be considered as workers and individuals weighing over 593.09 mg can be considered as potential future queens. The 5% level of uncertainty was reached beyond 525.44 mg for workers and below 664.84 mg for founders. Dry and wet weights are strongly correlated (rho = 0.88, $p < 2.2 \cdot 10^{-16}$) with the following linear regression formula: $y\ (Wwet) = 2.05 \cdot (Wdry) + 80.59$ and dry and wet weights and proved to be useful to discriminate workers and queens (Rome, et al., 2015). Every female was weighed with a precision balance (ADAM NBL 423i: 420 g capacity and precision of 0.001 g).

Statistical analysis

RStudio 3.3.2 software (R core team 2016) was used for analysis. It evaluated the correlation factor and its significance differences between the nest characteristics (Kruskal - Wallis and its Dunn *post-hoc*).

RESULTS

Architecture and structure of nests

Table 1 presents the characteristics of the nine *V. velutina* nests collected from August to November 2016 in Majorca. All the nests found in Majorca analysed in this work were secondary nests (no embryo nests were found inside), presented a calyptodomus typology and had ovoid morphology.

The number of combs within nests analysed ranged from five to nine (the last one, ID9, found in November), except for one nest (ID5, found in September) that had only 2 combs. The weight varied from 136 g (ID5) to 5,090 g (2811,7±482,4), the height from 7 cm (ID5) to 49.3 cm (28.9±5.1) and the maximum diameter from 12 cm (ID5) to 38 cm (26.4±2.7) (Table 1).

Shape of combs

The lower combs had a high coincidence percentage (88.6%) with a circular organisation. However, the coincidence percentage drops in the upper combs to 62.1%. A t-test was applied to observe if the mean of coincidence percentage with a circular organisation of the two upper combs was different from the two lower combs, which produced a $p = 0.0065$, so the circular organisation is lost ascending in the combs because the percentage in upper combs was lower than lower combs (Table 2).

Also, in upper combs there is reduced individual production, in the number of eggs and larvae; lower than in the first lower comb. In lower combs there are more immature stages so the individual production moves to the lower combs. There were significant differences relating to the number of cells between combs ($p = 0.0006$), and also in individual production ($p < 0.0001$). In both cases, cells and individual production, the significant differences were for the first and second lower combs.

Colony size

The evaluation of the total number of cells in the 50 combs of the nine mature nests, using Latter’s formula, revealed that the number of cells ranged from 200 (ID5) to 9,355 (4,073.1±947.5) and the general production (which is the sum of the estimated number of eggs, immature stages, meconium pellets (estimated with Latter’s formula) and adults) varied between 304.5 (ID5) and 9,316 individuals (3,613.08±853.67) (Table 1).

Spearman’s rank correlation test showed that diameter and the general production are highly correlated (rho = 0.895, $p = 0.001$) indicating that diameter is a good parameter for estimating the colony size. The general production follows the exponential function: $y = 0.1778x^{2.8995}$, where y is the

Table 2 Values obtained after applying the arbitrary categorisation. Total = sum of coincidence values between two diameters. Max = sum of the number of paired cells multiplied by the arbitrary categorisation “2”, it corresponds to a 100% coincidence percentage with a circular organisation. Total / Max = index of coincidence to a circular arrangement of breeding.

Combs	1			2			3			4			5		
Total	15	19	12	26	27	32	32	30	22	27	36	25	27	24	32
Maximum	16	22	14	40	40	44	52	46	46	50	58	52	40	42	52
Total/ Maximum	0.938	0.864	0.857	0.650	0.675	0.727	0.615	0.652	0.478	0.540	0.621	0.481	0.675	0.571	0.615
Mean	0.886			0.684			0.582			0.547			0.621		

general production of the nest and x is the largest diameter of the nest.

Sexual and caste differentiation

Males started to appear during autumn (ID6, ID8 and ID9), except for ID7, which was removed in October and had no males (Table 1). Season starts at 21 of September in north hemisphere and males of these three nests represented 57.9% of the adults found (999 males; 1,724 females in nests ID6, ID8 and ID9). All other nests (ID1-ID5) were found and removed before the first fortnight in September, and had not produced males.

Caste differentiation was determined by weighing all individuals and a weight increase was observed over time. Individuals in the last nests collected, weighed 63 mg more than those in the first nests removed in summer. During the dissection of the nine nests, a total of 5,581 females were weighed. The 97.4% of females found in the first eight nests (ID1-ID8) were workers and only the 1.3% of the females were in the uncertainty interval. However, in the last nest found (ID9) the percentage of females in the uncertainty interval was 48.2%.

DISCUSSION

All the nests found in Majorca during 2016 had a calyptodomus typology, and the number of combs within nests analysed ranged from 5 to 9. In comparison, the general production in mature nests found in France (4,797.75±606.40) revealed that the analysed nest, under isolated conditions, had similar production. So, the nests found in Majorca presented the same morphology as those nests found in other regions of Europe (Rome, et al., 2015).

Here, we suggest that the diameter of the nest is a good parameter to estimate the colony size, and the general production follows the exponential function: $y = 0.1778x^{2.8995}$ where y is the general production of the nest and x is the largest diameter of the nest. This is interesting in order to analyse the fitness of the species, and provide an easy way to analyse it as, by taking only one measurement (the largest diameter), the potential of each nest can be estimated.

Regarding the shape of combs, we provide a new method to check the circular organisation. The lower combs had a high coincidence percentage with a circular organisation and the coincidence percentage drops in the upper combs. The loss of the circular organisation when ascending in the combs and the higher number of immature individuals (such as eggs and larvae), in the lower combs, and pupae and meconium pellets in upper combs, is due to the fact that that this species of genus *Vespa* does not clean the cells after adult emergence (Janet, 1903), limiting each cell to produce between one and four individuals (Archer, 2008). This pattern is similar to the nest structure of *V. crabro* (Nadolski, 2012). Other species of *Vespa* have a higher number of meconium pellets per cell (Yamane & Makino, 1977; Yamane, 1992; Archer, 1993; Makino & Yamane, 1997), with four as the maximum (Archer, 2011) before the queen stops laying eggs inside the combs. Moreover, some authors suggest that un-cleaned combs are the reason the offspring are found in lower combs, which are cleaner than upper combs (Janet 1895). Moreover, the nest analysed presented the lower combs with more immature stages, so the individual production moves to the lower combs, which corresponded with Martin (1991, 1992).

Regarding the sexual and caste differentiation, it is important to note that the method of caste differentiation used in this work, based on the wet weight, was not very useful in our analysis as many individuals were in the error interval, and the increase in weight of 63 mg in workers was observed in autumn. Other authors proposed alternative methods for caste differentiation, such as Perrard, et al. (2012), who used size and nerve structure of the wings to distinguish individuals. Other possible methods are by genitalia differentiation or molecular methods used for other species of Hymenoptera (Barchuk, et al., 2007). So, for future research we will use these other methodologies. The presence of males during autumn is important information for a management plan on the island as it indicates the possibility that new founder queens can mate and create new nests the following season, signifying the beginning of the expansion of the invasive species.

In conclusion, the analysis of the secondary nests of the yellow-legged hornet found in the Balearic Islands reveals the high adaptability of this species to Mediterranean isolated ecosystems, which has important implications for the development of an effective eradication plan.

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REFERENCES

Arca, M., Papachristoforou, A., Mougel, F., Rortais, A., Monceau, K., Bonnard, O. and Arnold, G. (2014). ‘Defensive behaviour of *Apis mellifera* against *Vespa velutina* in France: Testing whether European honeybees can develop an effective collective defence against a new predator’. *Behavioural Processes* 106: 122–129.

Archer, M. (1993). ‘The life history and colonial characteristics of the hornet, *Vespa crabro* L. (Hym., Vespinae)’. *Entomologist's Monthly Magazine* 129: 151–163.

Archer, M. (1994). ‘Taxonomy, distribution and nesting biology of the *Vespa bicolor* group (Hym., Vespinae)’. *Entomologist's Monthly Magazine* 130: 149–158.

Archer, M. (2008). ‘Taxonomy, distribution and nesting biology of species of the genera *Provespa* Ashmead and *Vespa* Linneaus (Hymenoptera, Vespidae)’. *Entomologist's Monthly Magazine* 144: 69–101.

Archer, M. (2010). ‘The queen colony phase of vespine wasps (Hymenoptera, Vespidae)’. *Insectes Sociaux* 57: 133–145.

Archer, M. (2011). ‘Analysis of a nest of *Vespa multimaculata* Pérez, 1910’. *Malayan Nature Journal* 63: 561–567.

Barchuk, A.R., Cristino, A.S., Kucharski, R., Costa, L.F., Simões, Z.L. and Maleszka, R. (2007). ‘Molecular determinants of caste differentiation in the highly eusocial honeybee *Apis mellifera’. BMC developmental biology* 7: 70.

Brudno, M., Poliakov, A., Salamov, A., Cooper, G.M., Sidow, A., Rubin, E.M., Solovyev, V., Batzoglou, S. and Dubchak, I. (2004). ‘Automated whole-genome multiple alignment of rat, mouse, and human’. *Genome Research* 14: 685–692.

Bruneau, E. (2011). ‘Le frelon asiatique, déjà là’. *ActuApi* 55: 1–6.

Budge, G.E., Hodgetts, J., Jones, E.P., Ostoja-Starzewski, J.C., Hall, J., Tomkies, V., Semmence, N., Brown, M., Wakefield, M. and Stainton, K. (2017). ‘The invasion, provenance and diversity of *Vespa velutina* Lepeletier (Hymenoptera: Vespidae) in Great Britain’. *PLOS ONE* 12: 12.

Carpenter, J.M. and Kojima, J. (1997). ‘Checklist of the species in the subfamily Vespinae (Insectea: Hymenoptera: Vespida)’. *Natural History Bullety Ibaraki University* 1: 51–92.

Castro, L. and Pagola-Carte, S. (2010). ‘*Vespa velutina* Lepeletier, 1836 (Hymenoptera: Vespidae), recolectada en la Península Ibérica’. *Heteropterus Revista de Entomología* 10: 193–196.

Choi, M.B., Martin, S.J. and Lee, J.W. (2012). 'Distribution, spread, and impact of the invasive hornet *Vespa velutina* in South Korea'. *Journal of Asia-Pacific Entomology* 15(3): 473–477.

Choi, M.B., Lee, S-A., Suk, H.Y. and Lee, J.W. (2013). 'Microsatellite variation in colonizing populations of yellow-legged Asian hornet, *Vespa velutina nigrithorax*, in South Korea'. *Entomological Research* 43: 208–214.

Demichelis, S., Manino, A., Minuto, G., Mariotti, M. and Porporato, M. (2014). 'Social wasp trapping in north west Italy: comparison of different bait-traps and first detection of *Vespa velutina*'. *Bulletin of Insectology* 67: 307–317.

Edwards, R. (1980). *Social Wasps. Their Biology and Control*. Rentokil Ltd.

Grosso-Silva, J.M. and Maia, M. (2012). '*Vespa velutina* Lepeletier, 1836 (Hymenoptera, Vespidae), a new species for Portugal'. *Arquivos entomolóxicos* 6: 53–54.

Haxaire, J., Bouguet, J-P. and Tamisier, J-P. (2006) '*Vespa velutina* Lepeletier, 1836, une redoutable nouveauté pour la faune de France (Hym., Vespidae)'. *Bulletin de la Société entomologique de France* 111: 194.

Janet, C. (1895). 'Etudes sur les fourmis, les guêpes et les abeilles. 9th note. Sur *Vespa crabro* L. Histoire d'un nid depuis son origine (Studies on ants, wasps and bees. 9th note. About *Vespa crabro* L. History of a nest since its origin)'. *Mémoires de la Société Zoologique de France* 8: 1–140.

Janet, C. (1903). *Observations sur les Guêpes*. C. Naud, Paris.

Jeanne, R.L. (1975). 'The adaptiveness of social wasp nest architecture'. *The Quarterly Review of Biology* 50(3): 267–287.

Latter, O.H. (1935). 'A reason for the order in which the queen wasp constructs the cells of the comb, a method for computing the number of cells in a comb, and an instance of the limitation of the instincts of wasps'. *Proceedings of the Royal Entomological Society of London* 10: 74–78.

Leza, M., Miranda, M.Á. and Colomar, V. (2017). 'First detection of *Vespa velutina nigrithorax* (Hymenoptera: Vespidae) in the Balearic Islands (Western Mediterranean): a challenging study case'. *Biological Invasions*: 1-7.

López, S., González, M. and Goldarazena, A. (2011). '*Vespa velutina* Lepeletier, 1836 (Hymenoptera: Vespidae): first records in Iberian Peninsula'. *EPPO Bulletin* 41: 439–441.

Makino, S. and Yamane, S. (1997). 'Nest contents and colonial adult productivity in a common hornet, *Vespa simillima simillima* Smith, in northern Japan (Hymenoptera, Vespidae)'. *Japanese Journal of Entomology* 65: 47–54.

Martin, S.J. (1991) 'A simulation model for colony development of the hornet *Vespa simillima* (Hymenoptera, Vespidae).' *Japan Journal of Entomology* 59: 105–124.

Martin, S.J. (1992) 'Development of the embryo nest of *Vespa affinis* (Hymenoptera: Vespidae) in Southern Japan.' *Insectes Sociaux* 39: 45–57.

Martin, S.J. (1995). 'Hornets (Hymenoptera: Vespinae) of Malaysia'. *Malayan Nature Journal (Malaysia)* 49: 71–82.

Matsuura, M. (1991) '*Vespa* and *Provespa*.' In: K.G. Ross and R.W. Matthews (eds.) *The social biology of wasps,* pp. 232–262. Cornell University Press, New York,

Matsuura, M. and Yamane, S. (1990). *Biology of the Vespine Wasps*. Springer Verlag, Berlin.

Monceau, K., Maher, N., Bonnard, O. and Thiéry, D. (2013a). 'Predation dynamics study of the recently introduced honeybee killer *Vespa velutina*: learning from the enemy.' *Apidologie* 44: 209–221.

Monceau, K., Arca, M., Leprêtre, L., Mougel, F., Bonnard, O., Silvain, J.F., Maher, N., Arnold, G. and Thiéry, D. (2013b). 'Native prey and invasive predator patterns of foraging activity: the case of the yellow-legged hornet predation at European honeybee hives.' *PLOS One* 8: e66492.

Nadolski, J. (2012). 'Structure of nests and colony sizes of the European hornet (*Vespa crabro*) and Saxon wasp (*Dolichovespula saxonica*) (Hymenoptera: Vespinae) in urban conditions.' *Sociobiology* 59: 1075–1120.

Perrard, A., Villemant, C., Carpenter, J.M. and Baylac, M. (2012). 'Differences in caste dimorphism among three hornet species (Hymenoptera: Vespidae): forewing size, shape and allometry'. *Journal of Evolutionary Biology* 25: 1389–1398.

Pujade-Villar, J., Torrell, A. and Rojo, M. (2012). 'Confirmada la presència a Catalunya d'una vespa originària d'Àsia molt perillosa per als ruscs'. *Butlletí de la Institució Catalana d'Història Natural* 77: 173–176.

Rome, Q., Muller, F., Gargominy, O. and Villemant, C. (2009). 'Bilan 2008 de l'invasion de *Vespa velutina* Lepeletier en France (Hymenoptera: Vespidae)'. *Bulletin de la Société entomologique de France* 114: 297–302.

Rome, Q., Perrard, A., Muller, F., and Villemant, C. (2011). 'Monitoring and control modalities of a honeybee predator, the yellow-legged hornet *Vespa velutina nigrithorax* (Hymenoptera: Vespidae).' *Aliens* 31: 7–15.

Rome, Q., Dambrine, L., Onate, C., Muller, F., Villemant, C., García-Perez, L., Maia, M., Carvalho Esteves, P. and Bruneu, E. (2013). 'Spread of the invasive hornet *Vespa velutina* Lepeletier, 1836, in Europe in 2012 (Hym., Vespidae)'. *Bulletin de La Société Entomologique de France* 118: 15–21.

Rome, Q., Muller, F.J., Touret-Alby, A., Darrouzet, E., Perrard, A. and Villemant, C. (2015). 'Caste differentiation and seasonal changes in *Vespa velutina* (Hym.: Vespidae) colonies in its introduced range'. *Journal of Applied Entomology* 139: 771–782.

Spradbery, J.P. (1973). *Wasps: An Account of the Biology and Natural History of Solitary and Social Wasps with Particular Reference to Those of the British Isles*. Sidgwick and Jackson, London.

Spradbery, J.P., and Kirk, A.A. (1978). 'Aspects of the ecology of siricid woodwasps (Hymenoptera: Siricidae) in Europe, North Africa and Turkey with special reference to the biological control of *Sirex noctilio* F. in Australia.' *Bulletin of Entomological Research*, 68(3): 341–359.

Tan, K., Radloff, S.E., Li, J.J., Hepburn, H.R., Yang, M.X., Zhang, L.J. and Neumann, P. (2007). 'Beehawking by the wasp, *Vespa velutina*, on the honeybees *Apis cerana* and *A. mellifera*.' *Naturwissenschaften* 94: 469–472.

Ueno, T. (2014). 'Establishment of the invasive hornet *Vespa velutina* (Hymenoptera: Vespidae) in Japan'. *International Journal of Chemical, Environmental and Biological Sciences* 2: 220–222.

UK National Bee Unit (2016) <http://www.nationalbeeunit.com/index.cfm?pageId=208>. Accessed 29 May 2017.

Villemant, C., Haxaire, Je. and Streito, J.C. (2006). 'Premier bilan de l'invasion de *Vespa velutina* Lepeletier en France (Hymenoptera, Vespidae) (First assessment of the invasion of *Vespa velutina* Lepeletier in France (Hymenoptera, Vespidae)'. *Bulletin de La Société Entomologique de France* 111: 535–538.

Villemant, C., Barbet-Massin, M., Perrard, A., Muller, F., Gargominy, O., Jiguet, F. and Rome, Q. (2011). 'Predicting the invasion risk by the alien bee-hawking yellow-legged hornet *Vespa velutina nigrithorax* across Europe and other continents with niche models'. *Biological Conservation* 144: 2142–2150.

Witt, R. (2015). 'Erstfund eines nestes der Asiatischen nornisse *Vespa velutina* Lepeletier, 1838 in Deutschland und details zum nestbau (Hymenoptera, Vespinae)'. *Ampulex* 7: 42–53.

Yamane, S. (1992). 'A huge nest of *Vespa basalis* collected in Taiwan (Hymenoptera: Vespidae)'. *Chinese Journal of Entomology* 12: 1–11.

Yamane, S. and Makino, S.I. (1977). 'Bionomics of *Vespa analis insularis* and *V. mandarinia latilineata* in Hokkaido, northern Japan, with notes on vespine embryo nests (Hymenoptera: Vespidae)'. *Insecta Matsumurana*. *Series Entomology*. *New Series* 12: 1–33.

Methods for monitoring invertebrate response to vertebrate eradication

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Abstract Once an island vertebrate eradication is deemed successful, it is typically assumed that ecosystem recovery will follow. To date, most post-eradication monitoring focuses on the recovery of key threatened or charismatic species, such as seabirds. Little attention has been given to monitoring and quantifying the response of invertebrate communities. Rabbits (*Oryctolagus cuniculus*), house mice (*Mus musculus*), and ship rats (*Rattus rattus*) impacted sub-Antarctic Macquarie Island for over 140 years, with wide ranging ecosystem impacts. In 2014, the eradication of rabbits and rodents was officially declared successful. To determine whether management objectives are being met, we are investigating the response of invertebrate communities to pest eradication, using both historic data and contemporary surveys to track changes over space and time. To achieve this, we have developed a survey strategy that is effective and efficient. Here we report on the merits of utilising a variety of invertebrate trapping methodologies to establish current baselines for future invertebrate monitoring. We identify sampling techniques that are most effective for specific groups of taxa, particularly those of interest to post-eradication monitoring, and how the implementation of such methods can improve and facilitate effective post-eradication monitoring of invertebrates.

Keywords: baselines, conservation, insects, island, rabbits, restoration, rodents, sub-Antarctic

INTRODUCTION

Island invertebrates are impacted by invasive species, particularly on remote, unpopulated islands in the Southern Ocean (Chown, et al., 2008; Angel, et al., 2009; St Clair, 2011: Russell, 2012; Thoresen, et al., 2017). Non-native plants and invertebrates have been unintentionally introduced to Southern Ocean Islands (SOI) (Frenot, et al., 2005; Chown, et al., 2008; Convey & Lebouvier, 2009). Non-native vertebrates have also been introduced, both intentionally and inadvertently. For example, rabbits (*Oryctolagus cuniculus*), cats (*Felis catus*), dogs (*Canis lupus familiaris*), sheep (*Ovis aries*), goats (*Capra aegagrus hircus)*, weka (*Gallirallus australis*), pigs (*Sus scrofa domesticus*), brown trout (*Salmo trutta*) and reindeer (*Rangifer tarandus*) were all intentionally introduced to SOI, whereas ship rats (*Rattus rattus*), brown rats *(Rattus norvegicus)* and house mice (*Mus musculus*), were unintentional introductions (Copson & Whinam, 2001; Courchamp, et al., 2003; Convey and Lebouvier 2009; McGeoch, et al., 2015). Grazing by non-native vertebrates on SOI has led to invertebrate habitat modification (Vogel, et al., 1984; Chapuis, et al., 2004), and direct predation by rodents has severely modified and depleted invertebrate populations (Copson, 1986; Chown & Smith, 1993; Angel, et al., 2009; St Clair, et al., 2011; Russell, 2012; Treasure, et al., 2014).

Macquarie Island (54.6208° S, 158.8556° E) lies 1,500 km south-east of Tasmania, Australia. The island is a World Heritage area managed as a Nature Reserve by the Tasmanian Parks and Wildlife Service (Copson & Whinam, 2001). Discovered in 1810, the island's early human history involved seal harvesting (elephant seals, *Mirounga leonina;* fur seals *Arctocephalus pusillus, A. forsteri, A. tropicalis*) and penguin harvesting (king penguins, *Aptenodytes patagonicus* and royal penguins, *Eudyptes schlegeli*). Many non-native species of flora and fauna were introduced during this time, both intentionally and inadvertently. Ongoing control of cats and rabbits by various methods and at varying levels of effort led to fluctuating populations (Robinson & Copson, 2014; Terauds, et al. 2014). Consequently, native fauna and vegetation were impacted by varying levels of predation and grazing (Scott & Kirkpatrick, 2008; Scott & Kirkpatrick, 2013; Bergstrom, et al., 2009; Whinam, et al., 2014). Over time, island managers have removed almost all invasive vertebrate species from Macquarie Island (Copson & Whinam, 2001), the most recent being cats in 2000 (Robinson & Copson, 2014) and rabbits and rodents in 2014 (Springer, 2016). The latter were the target of the Macquarie Island Pest Eradication programme, which was the largest multi-species project of its kind at the time, costing AU$24.5 million. The project's overall objective was to '…restore Macquarie Island biodiversity and geodiversity to a natural balance – free of the impacts of introduced pest species… [with] …vegetation, seabird and invertebrate populations recovered to levels naturally supported by the environment' (Parks and Wildlife, 2009). We developed a study to assess the success of this project for invertebrates; specifically, to better understand if they have 'recovered' following removal of mammalian herbivores and predators, using both historic data and contemporary surveys.

Invertebrates play a key role in ecosystem function (Kremen, et al., 1993; Hutcheson, et al., 1999; Gerlach, et al. 2013). They drive nutrient-cycling and decomposition processes on SOI (Smith & Steenkamp, 1990; 1992; Smith, 2007; 2008). Thus, establishing a baseline and measuring their response to ecosystem change informs the state of the island ecosystem. Many types of invertebrates are useful proxies for assessing ecosystem change, reflected in their species richness, species turnover and community composition (Kremen, et al., 1993; Hutcheson, et al., 1999; Towns, et al., 2009). Indicator taxa are particularly useful in monitoring the effects of habitat management at the ground layer (e.g. ants, millipedes, snails, ground beetles, some spiders), in foliage (e.g. ants, leaf beetles, some spiders and moths), and in open habitats (e.g. ants, crickets, grasshoppers, and butterflies) (Gerlach, et al., 2013). Moreover, their high density, short life span, ubiquity and rapid response to changing environmental conditions, make invertebrates ideal for long-term monitoring (Samways, et al., 2010; McGeoch, et al., 2011).

Despite their suitability as indicators monitoring of invertebrates post-eradication is rarely undertaken and their response to eradication is infrequently determined.

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Developing appropriate survey methods and sampling strategies is crucial for a monitoring programme. Here we test a variety of invertebrate survey techniques and report on the merits of using specific invertebrate trapping methodologies to establish baselines for future invertebrate monitoring and to facilitate comparisons with previous surveys. Our recent surveys included most of the invertebrate trapping techniques previously employed on the island by historical surveys. Our survey design aimed to measure invertebrate response to vertebrate eradication and vegetation rehabilitation, track change in invertebrate community composition and numbers, and establish baselines for future monitoring. Specifically, our objectives in this paper, are to 1) compare the efficacy of using different invertebrate trap types in achieving monitoring objectives, 2) assess the effectiveness of historical trapping methods in informing contemporary survey design, and 3) investigate the benefits and limitations of using historical data for tracking changes over time. We also discuss how choosing appropriate methods is a key process for effective and efficient post-eradication monitoring of invertebrates.

METHODS

Survey design

Determining invertebrate community changes over time requires definitive and repeatable methods and detailed site information. Our experimental design (i.e. our choice of survey/trapping techniques and site selection) was informed by analysing invertebrate trapping experimental designs, methods and results from historical surveys on Macquarie Island. Following a thorough review of the scientific literature five key resources were selected to inform our experimental trapping design and methods: Watson (1967), Greenslade (1987), Anonymous (1993–94, reported in Stevens, et al., 2010), Davies & Melbourne (1999), and Stevens, et al., (2010). Each of these historical surveys utilised different combinations of methods (Table 1). Based on this information, the following survey methods were tested in our study: pitfall traps, sweep netting, litter extraction, and timed hand collecting (referred to as '20-minute counts').

Site selection

For this study, sampling was carried out at ten historic and ten new sites (Fig. 1). This provided 20 sites in total for the 2015/16 post-eradication survey. The ten new sites were selected to ensure broader island coverage and survey additional vegetation communities across the five dominant vegetation structures on Macquarie Island (based on Selkirk, et al., 1990) – feldmark (plateau), lower coastal slopes dominated by *Stilbocarpa polaris* (Macquarie Island cabbage), tall grassland (tussock) dominated by *Poa foliosa*, short grasslands (including *Deschampsia* spp., *Festuca contracta, Agrostis magellanica, Luzula crinita, Uncinia* spp.), and herbfield dominated by *Pleurophyllum hookerii*. Most sites were heavily impacted by rabbits in the past (Bergstrom, et al., 2009; Whinam, et al., 2014). In total, four *Stilbocarpa polaris* sites were surveyed in 2015/16, three short grassland sites, five tall grass sites, four herbfield sites, and four feldmark sites.

Sampling techniques

Five pitfall traps were established at each of the 20 sites, in a line transect along a recorded bearing, five metres apart. Expert advice from the Tasmanian Department of Primary Industries, Parks, Water and the Environment (M. Driessen, pers. comm.), informed the pitfall trap preparation, spacing, pattern of positioning, and preservative used. Pitfall traps were constructed of straight sided, plastic jars 7 cm in diameter, approximately 7 cm deep, with ca. 1 cm of 100% propylene glycol preservative added. Pitfall diameter was selected based on other studies

Table 1 Trapping methodology employed during invertebrate sampling studies on Macquarie Island – Watson in 1961 (reported in Watson, 1967), Greenslade in 1986–87 (reported in Greenslade, 1987), Anonymous in 1993–94 (reported in Stevens, et al., 2010), Davies and Melbourne in 1996 (reported in Davies & Melbourne, 1999), Stevens, et al., in 2009–10 (reported in Stevens, et al., 2010).

	Watson 1961	Greenslade 1986–87	Anonymous 1993–94	Davies & Melbourne 1996	Stevens, et al., 2009–10
Length of sampling	Year-long	December – January	Year-long	February –March	October –January
Extent of sampling	Island-wide	Northern sites	Northern sites	Island-wide	Northern sites
# Sites	Not specified	8	4	67	41
# Pitfalls/site	0	5	Not specified	3	3
# Pitfall trap days	-	5–20	30	ca. 42	7–21
Pitfall diameter (cm)	-	1.8	'Large' & 'small'	3	'Large' & 'small'
Pitfall medium	-	Ethanol	Not specified	Ethylene glycol/ detergent	Ethanol
# Yellow pan trap/site	Yes, # not specified	0	0	1	0
Vegetation Beating	No	Yes, # not specified	No	No	Yes, # not specified
Vegetation Sweeping	Yes, # not specified	Yes, # not specified	No	No	No
Litter volume (Lt)	Yes, # not specified	2–4	0	0	1 over 1 m²
Litter extraction method	Berlese funnels	Berlese funnels	n/a	n/a	Berlese funnels
# Soil Cores	Yes, # not specified	11–16	5	0	0
20-minute counts (hand collecting)	Yes (not timed)	Yes (not timed)	No	Yes	No

that have proven the effectiveness of larger trap sizes (Brennan, et al., 1999; Ward, et al., 2001; Work, et al., 2002; Woodcock, 2005). Propylene glycol was chosen a preservative due to it being environmentally benign, highly viscous and slow to evaporate. Pitfall trap holes were dug with a soil corer to ensure a snug fit and the pitfall rim was flush with the ground surface. Where necessary, a small amount of vegetation was cleared from the trap rim. At the 20 sites, pitfall traps were collected approximately every 10 days between October and December and reset upon collection for a total period of up to ca. 42 days. Further pitfall sampling was repeated monthly in January, February and March for approximately 5–10 days at eight sites.

For litter sampling, at each site, a 1m² quadrat was used to define the collection area, and three collections were made of one litre of litter at each site. Litter was transported back to the station laboratory for sorting and invertebrate extraction within a maximum of three days from collection. In feldmark sites where litter was scarce, litter collection was over 4 m² and up to 1 litre of material – the exact area and volume was recorded.

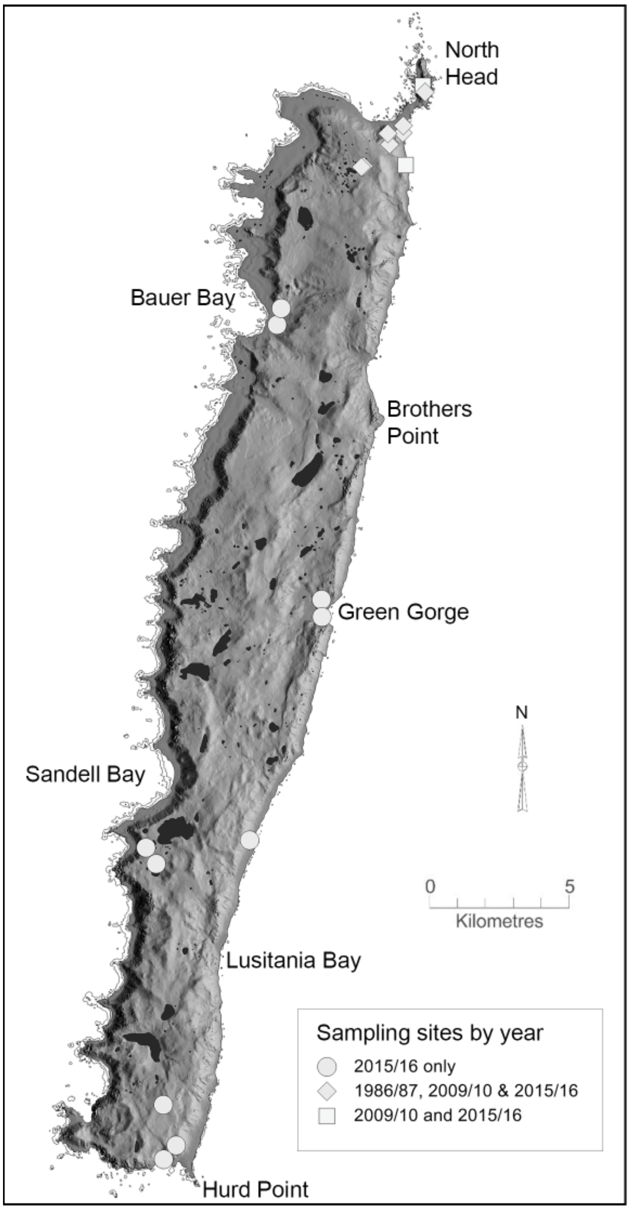

Fig. 1 Map of 20 invertebrate trapping sites surveyed at Macquarie Island in 2015/16. All historic sites sampled in 1986/87 (indicated by grey diamonds) were resampled in 2015/16.

Timed counts of 20 minutes were conducted at least twice over the study period at each site, involving focussed searching with an aspirator tube and tweezers, collecting all invertebrates encountered, particularly at the base of vegetation and under stones.

Sweeping of vegetation tops with nets required dry conditions with light winds. Hence, sweeping was conducted opportunistically, at a minimum of twice at each site over the study period, in all vegetation types regardless of the canopy (i.e. also in feldmark), by walking slowly and dragging the net across the vegetation tops 30 times, on three different trajectories in the site area per sampling event.

One temperature logger i-button was installed at each of the 20 sites to monitor microclimate, at the first pitfall trap of each transect. They were attached approximately 10 cm above the ground surface on a stake with a protective housing. At each site, site-specific features such as aspect, altitude, landscape features, vertebrate fauna presence and vegetation were noted.

Processing and identification

All samples were transferred promptly to 100% ethanol and transported back to the Australian Antarctic Division for identification and storage. Using a dissecting microscope, specimens were counted and identified to species where possible, except for Acarina, Annelida, and Nematoda, which were identified to Class or Phylum level.

Data analysis

We undertook preliminary comparisons of the 2015/16 survey data on Order richness and diversity with data from historical sites established in 1986/87 (Greenslade, 1987). We calculated taxonomic richness and diversity of invertebrates in different trap types, vegetation groups and in historical data. For these purposes we pooled data from different sites to obtain the total number of taxa trapped in different trap types and vegetation groups. Invertebrate richness was calculated by summing the total number of invertebrate Orders recorded in the trap type or vegetation group of interest. Simpson's Index of Diversity (SID) was selected to compare diversity, as it takes into account both abundance and richness in each habitat. We compared the Order richness and diversity of our pitfall traps to seven historical sites and also quantified changes in abundance for three target groups (beetles, spiders and moths), using a subset of the historic and contemporary pitfall data.

We analysed data at the level of Order/Class/Phylum (hereafter referred to as 'Order') to facilitate preliminary comparison with historic data sorted to Order resolution. For analysis, larval stages and adults were grouped together for Lepidoptera, Thysanoptera, Coleoptera and Diptera.

RESULTS

Contemporary survey

Our preliminary results from the 2015/16 survey demonstrated that pitfall traps collect the largest number of individuals – in particular, Collembola (Table 2). Even when Collembola were removed from the analysis, pitfalls still collected more individuals than other trapping methods. Despite the abundance of invertebrates in pitfalls, there was considerable variance in the nature and abundance of taxa caught by the different trapping methods, with some methods proving more effective for specific taxa than others (Table 2).

Richness (number of different orders caught) and diversity (SID – richness combined with the relative

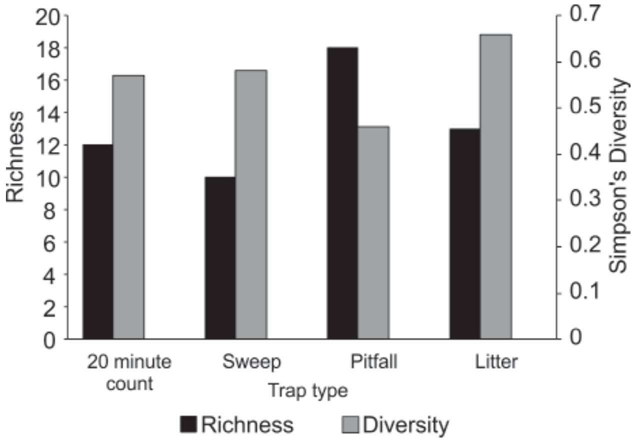

Fig. 2 Order richness (summed across 20 sites) and Simpson's diversity of four different trapping methods on Macquarie Island in 2015/16 following mammal eradication.

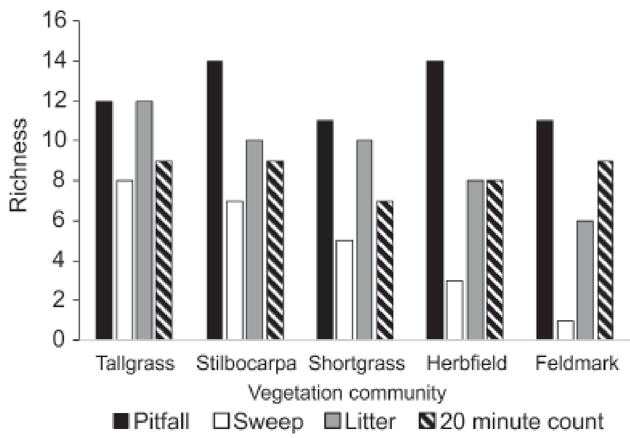

Fig. 3 Order richness (summed across 20 sites) of four trapping types in five vegetation communities on Macquarie Island in 2015/16 following mammal eradication.

abundance of the different orders caught) varied between trapping methods (Fig. 2). The SID demonstrated that although pitfalls traps yielded the greatest richness, they also had the lowest diversity, attributable largely to the dominance of Collembola. Conversely, sweeping had relatively low species richness but high SID, an indication of the greater relative abundance of different taxa trapped.

Pitfalls collected the most species regardless of habitat type (Table 2, Fig. 3). Effectiveness of the other trap types varied by vegetation community (Fig. 3). Sweeping vegetation was far more effective in tall grassland and *S. polaris*, which are often characterised by dense protective foliage, than for herbfield and feldmark vegetation, which typically have more prostrate, sparsely distributed plants. Litter collection also yielded high relative Order richness, particularly in tall grassland and short grassland vegetation communities. Twenty-minute counts were effective in feldmark, where richness was proportional to effort. The low number of taxa in this habitat were found more readily through this method of focused searching (disturbing stones and turf), than via passive pitfall trapping or surface litter collection.

The SID of pitfall trap samples across vegetation types was almost the inverse of their richness (Fig. 4). Across all vegetation types (except for feldmark), pitfall trapping diversity was much lower than for other trap types; a likely reflection of the dominance of the Collembola in pitfall traps except in feldmark. For short grassland, litter sampling proved to be exceptionally diverse. Interestingly, although taxonomic richness of sweeping in herbfield was relatively low, SID was high. Across all vegetation types, 20-minute counts were almost equal in diversity.

Table 2 The number of individuals from each Order of invertebrates collected via four different trapping methods on Macquarie Island following mammal eradication: pitfall traps, sweeping, 20-minute counts, and litter collection in the 2015/16 season following mammal eradication.

Order	Common Name	Pitfalls	Sweep	20 minute count	Litter
Gastropoda	Snails/slugs	935	2	1,294	1,019
Psocoptera	Booklouse	44	2	0	129
Hemiptera	Aphids/Bugs	3	0	0	0
Thysanoptera	Thrips	144	21	4	4
Coleoptera	Beetles	2,512	2	12	240
Diptera	Flies	945	61	51	61
Lepidoptera	Moths	3	0	4	8
Hymenoptera	Wasps	1	0	0	0
Isopoda	Crustacea	209	1	207	636
Araneae	Spiders	2,467	40	169	380
Platyhelminthes	Flatworms	20	0	1	1
Annelida	Worms	284	3	493	1,489
Copepoda	Copepods	3,615	0	2	8
Tardigrada	Tardigrades	69	0	0	0
Acarina	Mites	5,219	40	108	340
Siphonaptera	Fleas	1	0	0	1
Nematoda	Nematodes	19	0	0	0
Collembola	Springtails	43,641	277	3,609	5,040
	TOTAL	60,131	449	5,954	9,356

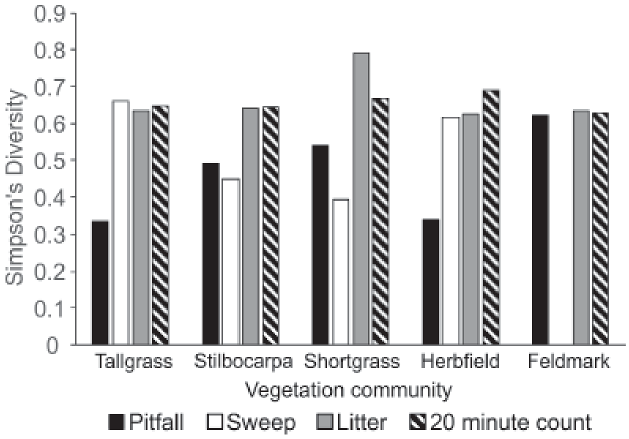

Fig. 4 Simpson's Index of Diversity (Order) of four trap types in five vegetation communities on Macquarie Island in 2015/16 following mammal eradication.

Comparisons with historical surveys

Preliminary comparisons of our data on Order richness and diversity data from the 1986/87 sites (Fig. 5) indicate considerable changes in invertebrate communities since the earlier surveys. Both Order richness and SID were generally lower during the earlier sampling period compared to 2015/16, with the exception of the feldmark F2 site, where 1986/87 samples were more speciose and more diverse. Diversity in the tall grassland site P2 and herbfield H1 were also lower in 2015/16 sampling, though richness was much higher.

Mouse prey species that were predicted to respond favourably to mouse removal, such as Coleoptera (beetles), Lepidoptera (moths) and spiders (Araneae), were trapped via pitfalls in 1986/87 and again in 2015/16 at seven sites across five vegetation types (Table 3). Coleoptera abundance was inconsistently higher in 1986/87, whereas Araneae were trapped in much higher numbers during the 2015/16 sampling. Lepidoptera were rarely trapped in both sampling events, present only in the feldmark F2 site.

DISCUSSION

When monitoring ecosystem responses following an eradication, it is critical to first identify the objectives of the management intervention. In this case, the facilitation of the "recovery" of macro-invertebrates on Macquarie Island was explicit. However, no mechanisms were put in place to assess the success (or otherwise) of this objective. Here, our preliminary study tackles the issue of how to effectively survey a suite of invertebrate species on a Southern Ocean island to detect recovery and response of invertebrates after an eradication event, and informs the selection of appropriate survey methods for specific species.

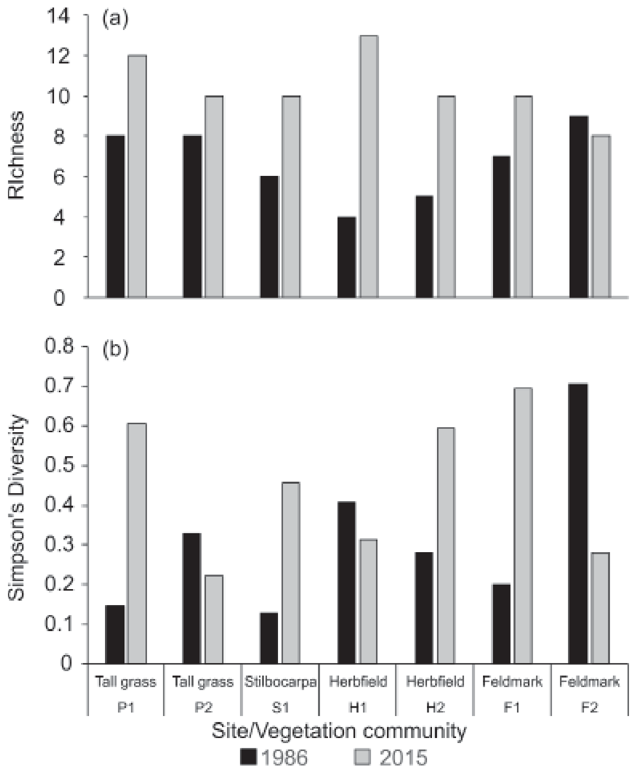

Fig. 5 (a) Order richness (summed across 20 sites) and, (b) Simpson's Index of Diversity of pitfall trapping (Order level) at seven invertebrate monitoring sites at Macquarie Island that were first sampled in 1986/87 (prior to mammal eradication) and repeat sampled in 2015/16 (post mammal eradication).

One of the clearest findings of our study was that pitfall traps collect the greatest abundance and richness of invertebrates, particularly Collembola, although they were the least diverse. Despite the difficulties in comparing abundance and sampling effort across different techniques (for example, the longer deployment time of pitfall traps compared to other trapping techniques), it is apparent that different trapping methods are more effective at capturing certain taxonomic groups. This is based on the functional traits, behaviours and preferred habitats of different taxa. For example, Psocoptera were primarily collected from litter samples, as they are detritivores with a preference for damp conditions under vegetation (Greenslade, 2006). However, some were also collected during vegetation sweeping, where they occur in smaller numbers under leaves (Greenslade, 2006). Tardigrades and Copepods

Table 3 Abundance of Coleoptera, Lepidoptera and Araneae in pitfall traps sampled at seven sites at Macquarie Island in 1986/87 (prior to mammal eradication) and 2015/16 (post- mammal eradication).

		Coleoptera		Lepidoptera		Araneae	
		1986/87	2015/16	1986/87	2015/16	1986/87	2015/16
P1	Tall grass	25	1,909	0	0	151	124
P2	Tall grass	19	7	0	0	105	280
S1	*Stilbocarpa*	2,426	277	0	0	84	175
H1	Herbfield	8	6	0	0	27	123
H2	Herbfield	2	1	0	0	42	191
F1	Feldmark	4	3	0	0	8	120
F2	Feldmark	4	0	1	4	28	127

were collected principally via pitfall traps, most likely due their existence in soil or at the soil surface, particularly in moist sites. Their small size makes them unlikely to be detected through other trapping means. Cosmopolitan groups like Coleoptera, Collembola and Acarina were detected by all trapping methods. For the Collembola, their ubiquity in many samples exemplified their abundance and diversity on the island, with 35 species recorded (Phillips, et al., 2017). They also occur in a variety of habitats, from soil-dwellers to canopy species (Greenslade, 2006; Terauds, et al., 2011). Similarly, the collection of predatory Staphylinidae coleopterans across all trapping methods suggests this group are wide-ranging across vegetation, possibly to maximise opportunities to encounter prey. Isopoda, Annelidae and Platyhelminthes were collected by all means except sweeping (with a few exceptions), in line with their cryptic habits under vegetation, close to the soil surface and in litter (Greenslade, 2006).

Knowledge of the target group is critical to inform the experimental design of trapping. For example, and perhaps counter-intuitively, sweeping did not yield high numbers of moths or flies. One reason may be that many resident flies on Macquarie Island are flightless and largely stay close to the ground (Greenslade 2006). Furthermore, the endemic moth *Eudonia mawsoni*, which is not nocturnal, is known to drop to the ground when dislodged from vegetation (i.e. by sweep nets) (Jackson, 1995), and often stays close to the ground, taking shelter in winds over 10 km/hr (Greenslade, 2006). Sweeping can only occur during low wind conditions, however winds are typically high on the island (Pendlebury & Barnes-Keoghan, 2007), dispersing many taxa (both flightless and flying) (Hawes & Greenslade, 2013). The moth's flight is stimulated by rain, however sweeping is not possible during rains as wet vegetation renders the sweep net ineffective. Such background understanding of target taxa and the environment informs the design and interpretation of trapping surveys.

If the monitoring or management objectives focus on a particular group or species it is important to consider the varying effectiveness of trapping methods (Zou, et al., 2012). For example, mice on SOI prey mainly on invertebrates, especially spiders, moths, beetles, aphids, Orthoptera (e.g. crickets), snails and earthworms (Copson, 1986; Crafford & Sholtz, 1987; Rowe-Rowe, et al., 1989; Le Roux, et al., 2002; Jones, et al., 2003; Angel, et al., 2009; St Clair 2011; Russell, 2012). Copson (1986) identified that mice on Macquarie Island had a particular preference for spiders (67% of 108 mouse stomach contents), caterpillars of the endemic moth *E. mawsoni* and, to a lesser extent, other invertebrates such as beetles and dipteran larvae. Therefore, increases in these taxa following mouse eradication and the removal of predation pressure could be anticipated. Our preliminary comparisons provided some support for this hypothesis (see below). To measure the response of invertebrates preyed upon by mice on Macquarie Island, our results indicate that pitfall trapping is effective for spiders and beetles and is therefore the most efficient mechanism for assessing their recovery. Monitoring of moths will require greater consideration and ongoing effort, as they were not detected in high numbers by any trapping method during the 2015/16 season.

Comparisons to historic datasets are vital to detect responses to eradication. It is important to consider there may be different responses and recovery times in different species. Again, although our comparative analyses are only preliminary, they do show a higher abundance of spiders in pitfalls in 2015/16 compared with 1986/87 pitfalls, across all sites. There is a high likelihood that this is related to the eradication of mice, given spiders were a major prey item (Copson, 1986). However, beetle abundance did not change consistently between the two trapping events, with numbers trapped varying across sites and between years (Table 3). One possible reason is that Staphylinidae beetles (which comprised all of the beetles caught) can occur in dense numbers where rich detritus or rotting material is present on coastal terraces in vegetation (Greenslade 2006). As a result, they can be very abundant in an individual sample from one event, and then absent in others at the same site. Vegetation recovery is slow, and therefore, if beetle abundance is driven by vegetation, there will be a delay in beetle response to rabbit eradication.

For the moth *E. mawsoni*, despite anecdotal reports of increased abundance across the island, our preliminary data do not show this. The moth pitfall counts were similar in 1986/87 and 2015/16, with adults rarely trapped, which is consistent with other studies on Macquarie Island (Jackson, 1995; Potts, 1997; Stevens, et al., 2010; Hawes & Greenslade, 2013). The low number of adults in our data could be due to the timing of our sampling regime, i.e. our trapping effort was low in late December and early January – the time when adults are most abundant and active (Watson, 1967). Davies & Melbourne (1999) captured many adults using yellow pan traps. With this knowledge, we have added this method to our trapping regime for future seasons of the invertebrate monitoring project to identify change. We also extended our future sampling to occur later in the summer, between January–March, to identify seasonal changes in species, improve likelihood of encountering different species, and improve chances of detecting different life stages in species (such as the moths and moth larvae). Species life history must be considered when designing trapping to inform responses to management.

Terrestrial invertebrate communities that are dependent on or restricted to specific vegetation or habitat types are hypothesised to be most likely to be impacted by rabbit grazing on Macquarie Island (Parks and Wildlife, 2009). Vegetation has undergone considerable changes between 1986/87 and 2015/16 (Copson & Whinam, 1998; Bergstrom, et al., 2009; Shaw, et al., 2011). Our preliminary results highlight the potential utility of historical data, when combined with targeted and appropriate sampling techniques, to explore the relationship between vegetation and invertebrates. Overall, there appears to be an increase in richness and diversity from 1986/87 to 2015/16. During the period of the initial sampling in 1986/87, rabbit numbers and the commensurate vegetation damage were relatively high (Terauds, et al., 2014), which may explain the low numbers of vegetation-dependant invertebrates. Herbfields were favoured by rabbits and heavily impacted by grazing (Scott, 1988; Selkirk, et al., 1990; Copson & Whinam, 1998). Herbfield invertebrate communities were particularly low in richness and diversity in 1986/87. Feldmark communities vary little between 1986/87 and 2015/16, most likely as rabbit impacts were low in this high-altitude vegetation group (Copson & Whinam, 1998).

Another important consideration of these comparisons is that the historical survey data we accessed were generally based on higher taxonomic groupings, which may impact our ability to detect subtle changes in invertebrate communities over time (e.g. Grimbacher, et al., 2008). Higher taxonomic groupings, unsurprisingly, do not necessarily reflect the finer details of invertebrate community assemblages, nor nuanced changes in their structure and responses (Grimbacher, et al., 2008; Driessen & Kirkpatrick, 2017) and may aggregate species that have different ecological or functional traits and responses to disturbance (Lenat & Resh, 2001; Heino & Soininen, 2007; Schipper, et al., 2010). A limitation of using historical datasets is that often clarification and further detail is simply not available.

Identification of specimens to fine taxonomic resolution, such as to species level, takes considerable time. Although we focussed on higher-level taxa here, our forthcoming analyses at finer taxonomic resolution, (such as to family, genus and for some groups, species) will provide further insights on trapping efficacy, survey design and most importantly, invertebrate community changes over time. However, there is also good evidence to suggest that higher-level identification can be appropriate surrogates for species and effective for detecting major disturbance events on invertebrate community structure, particularly where there are significant environmental perturbations or gradients (Driessen & Włodarska-Kowalczuk & Kędra, 2007; Bevilacqua, et al., 2012; Kirkpatrick, 2017). Sorting samples to higher taxonomic resolution is often more practical and cost-effective as it requires less training and maximises the potential for swift sample processing (Driessen & Kirkpatrick, 2017). The selected taxonomic resolution must be balanced between available resources and the value of results – the decision ultimately depending on the study objectives (Driessen & Kirkpatrick, 2017). It is fundamentally important to decide at the outset which invertebrate taxa, (if not all of them), are the focus of the monitoring, and what taxonomic resolution will best meet the objectives of the monitoring. For example, the varying ranges and habitat associations of the five Staphylinidae beetle species on Macquarie Island are not represented when grouped to the Coleoptera order in our data. A second example is the ubiquitous and numerous Collembola (springtails), that when aggregated to Order, fail to highlight the very different species trapped by each medium, such as those that were trapped via sweeping which inhabit the canopy, foliage and flowers in vegetation, and those edaphic groups trapped via pitfalls that either inhabit the soil or live close to the surface. Such details are important to our Macquarie Island monitoring objectives as we assess invertebrate communities in recovering vegetation.

The availability of historical data greatly enhances the power of long-term effective monitoring. In this instance, considerable time was invested in tracking down historical datasets and their metadata. Our future work will include in-depth analysis of contemporary survey results in relation to a broader suite of historical data. Whether historical data are available at the outset or not, establishing a baseline from which to measure changes into the future is critical for long-term monitoring, for making informed management decisions, and assessing management success. Our preliminary results demonstrate that invertebrate monitoring in a post-vertebrate eradication ecosystem can yield important and promising results. Effective monitoring for invertebrates also leads to improved surveillance for non-native species arrivals and potential non-native species impacts. Our future work includes the collection of two additional years of invertebrate surveys (2016/17 and 2018) across Macquarie Island and the establishment of four additional invertebrate monitoring sites to improve island and vegetation community coverage. We will also employ additional trapping methods (vegetation beating and yellow-pan trapping), and use Berlese funnels in the 2016/17 and 2018 surveys for more efficient litter processing. These improvements combined, will further develop baseline knowledge of invertebrate communities on Macquarie Island and inform future monitoring. This work will provide a comprehensive snapshot of ecosystem function and recovery following vertebrate eradication. We will use these results to develop and propose an efficient means of invertebrate monitoring using specific taxa or groups as biological indicators of broader ecosystem changes, to enable robust and efficient monitoring into the future.

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REFERENCES

Angel, A., Wanless, R.M. and Cooper, J. (2009). 'Review of impacts of the introduced house mouse on islands in the Southern Ocean: Are mice equivalent to rats?' *Biological Invasions* 11(7): 1743–1754.

Bergstrom, D.M., Lucieer, A., Kiefer, K., Wasley, J., Belbin, L., Pedersen, T.K. and Chown, S.L. (2009). 'Indirect effects of invasive species removal devastate world heritage island'. *Journal of Applied Ecology* 46(1): 73–81.

Bevilacqua, S., Terlizzi, A., Claudet, J., Fraschetti, S. and Boero, F. (2012). 'Taxonomic relatedness does not matter for species surrogacy in the assessment of community responses to environmental drivers'. *Journal of Applied Ecology* 49(2): 357–366.

Brennan, K.E., Majer, J.D. and Reygaert, N. (1999). 'Determination of an optimal pitfall trap size for sampling spiders in a Western Australian jarrah forest'. *Journal of Insect Conservation* 3(4): 297–307.

Chapuis, J.-L., Frenot, Y. and Lebouvier, M. (2004). 'Recovery of native plant communities after eradication of rabbits from the subantarctic Kerguelen Islands, and influence of climate change'. *Biological Conservation* 117(2): 167–179.

Chown, S.L. and Smith, V.R. (1993). 'Climate change and the short-term impact of feral house mice at the sub-Antarctic Prince Edward Islands'. *Oecologia* 96(4): 508–516.

Chown, S.L., Lee, J.E. and Shaw, J.D. (2008). 'Conservation of Southern Ocean islands: Invertebrates as exemplars'. *Journal of Insect Conservation* 12(3): 277–291.

Copson, G. R. (1986). 'The diet of the introduced rodents *Mus musculus* L. and *Rattus rattus* L on sub-Antarctic Macquarie Island'. *Wildlife Research* 13(3): 441–445.

Copson, G. and Whinam, J. (1998). 'Response of vegetation on subantarctic Macquarie Island to reduced rabbit grazing'. *Australian Journal of Botany* 46(1): 15–24.

Copson, G. and Whinam, J. (2001). 'Review of ecological restoration programme on subantarctic Macquarie Island: Pest management progress and future directions'. *Ecological Management and Restoration* 2(2): 129–138.

Courchamp, F., Chapuis, J.-L. and Pascal, M. (2003). 'Mammal invaders on islands: Impact, control and control impact'. *Biological Reviews* 78(03): 347–383.

Crafford, J.E. and Scholtz, C.H. (1987). 'Quantitative differences between the insect faunas of sub-antarctic Marion and Prince Edward islands: A result of human intervention?'. *Biological Conservation* 40(4): 255–262.

Davies, K.F. and Melbourne, B.A. (1999). 'Statistical models of invertebrate distribution on Macquarie Island: A tool to assess climate change and local human impacts'. *Polar Biology* 21: 240–250.

Driessen, M.M. and Kirkpatrick, J.B. (2017). 'Higher taxa can be effective surrogates for species-level data in detecting changes in invertebrate assemblage structure due to disturbance: a case study using a broad range of orders'. *Austral Entomology*: doi.org/10.1111/aen.12315.

Frenot, Y., Chown, S.L., Whinam, J., Selkirk, P.M., Convey, P., Skotnicki, M. and Bergstrom, D.M. (2005). 'Biological invasions in the Antarctic: Extent, impacts and implications'. *Biological Reviews* 80(1): 45–72.

Gerlach, J., Samways, M. and Pryke, J. (2013). 'Terrestrial invertebrates as bioindicators: An overview of available taxonomic groups'. *Journal of Insect Conservation* 17: 831–850.

Greenslade, P. (1987). Report on invertebrate fieldwork Macquarie Island, December 1986 – January 1987. Unpublished report. Tasmania, Australia: Department of Primary Industry, Parks, Water and the Environment.

Greenslade, P. (2006). *The Invertebrates of Macquarie Island.* Kingston: Australian Antarctic Division.

Grimbacher, P.S., Catterall, C.P. and Kitching, R.L. (2008). 'Detecting the effects of environmental change above the species level with beetles in a fragmented tropical rainforest landscape'. *Ecological Entomology* 33(1): 66–79.

Hawes, T.C. and Greenslade, P. (2013). 'The aerial invertebrate fauna of subantarctic Macquarie Island'. *Journal of Biogeography* 40(8): 1501–1511.

Heino, J. and Soininen, J. (2007). 'Are higher taxa adequate surrogates for species-level assemblage patterns and species richness in stream organisms?'. *Biological Conservation* 137(1): 78–89.

Hutcheson, J., Walsh, P. and Given, D.R. (1999). 'Potential Value of Indicator Species for Conservation and Management of New Zealand Terrestrial Communities'. *Science for Conservation* 109*.* Wellington, New Zealand: Department of Conservation.

Jackson, J. (1995). 'Notes on the biology of *Eudonia mawsoni* (Lepidoptera: Pyralidae) on subantarctic Macquarie Island'. *Polar Biology* 15(4): 289–294.

Jones, A.G., Chown, S.L., Webb, T.J. and Gaston, K.J. (2003). 'The free-living pterygote insects of Gough Island, South Atlantic Ocean'. *Systematics and Biodiversity* 1(2): 213–273.

Kremen, C., Colwell, R.K., Erwin, T.L., Murphy, D.D., Noss, R.F. and Sanjayan, M.A. (1993). 'Terrestrial arthropod assemblages: Their use in conservation planning'. *Conservation biology* 7(4): 796–808.

Le Roux, V., Chapuis, J.-L., Frenot, Y. and Vernon, P. (2002). 'Diet of the house mouse (*Mus musculus*) on Guillou Island, Kerguelen archipelago, Subantarctic'. *Polar Biology* 25(1):49–57.

Lenat, D.R. and Resh, V.H. (2001). 'Taxonomy and stream ecology—the benefits of genus-and species-level identifications'. *Journal of the North American Benthological Society* 20(2): 287–298.

McGeoch, M.A., Sithole, H., Samways, M.J., Simaika, J.P., Pryke, J.S., Picker, M., Uys, C., Armstrong, A.J., Dippenaar-Schoeman, A.S. and Engelbrecht, I.A. (2011). 'Conservation and monitoring of invertebrates in terrestrial protected areas'. *Koedoe* 53(2): 131–143.

McGeoch, M.A., Shaw, J.D., Terauds, A., Lee, J.E. and Chown, S.L. (2015). 'Monitoring biological invasion across the broader Antarctic: A baseline and indicator framework'. *Global Environmental Change* 32: 108–125.

Parks and Wildlife. (2009). *Macquarie Island Pest Eradication Plan. Part C: Environmental Impact Statement*. Hobart, Tasmania: Department of Primary Industries, Parks, Water and the Environment.

Pendlebury, S. and Barnes-Keoghan, I. (2007) 'Climate and climate change in the sub-Antarctic'. *Papers and Proceedings of the Royal Society of Tasmania* 141(1): 67–81.

Phillips, L., Janion-Scheepers, C., Houghton, M., Terauds, A., Potapov, M. and Chown, S.L. (2017). 'Range expansion of two invasive springtails on sub-Antarctic Macquarie Island'. *Polar Biology*: 1–6.

Potts, T. (1997). 'Monitoring Invertebrate Migration and Dispersal on Subantarctic Macquarie Island'. BSc Honours Thesis. Hobart: University of Tasmania, Institute of Antarctic and Southern Ocean Studies (IASOS).

Robinson, S.A. and Copson, G.R. (2014). 'Eradication of cats (*Felis catus*) from subantarctic Macquarie Island'. *Ecological Management and Restoration* 15(1): 34–40.

Rowe-Rowe, D., Green, B. and Crafford, J. (1989). 'Estimated impact of feral house mice on sub-Antarctic invertebrates at Marion Island'. *Polar Biology* 9(7): 457–460.

Russell, J.C. (2012). 'Spatio-temporal patterns of introduced mice and invertebrates on Antipodes Island'. *Polar Biology* 35(8):1187–1195.

Samways, M.J., McGeoch, M.A. and New, T.R. (2010). *Insect Conservation: A Handbook of Approaches and Methods*. Oxford: Oxford University Press.

Schipper, A.M., Lotterman, K., Geertsma, M., Leuven, R.S. and Hendriks, A.J. (2010). 'Using datasets of different taxonomic detail to assess the influence of floodplain characteristics on terrestrial arthropod assemblages'. *Biodiversity and Conservation* 19(7): 2087–2110.

Scott, J. (1988) 'Rabbit distribution history and related land disturbance, Macquarie Island'. *Papers and Proceedings of the Royal Society of Tasmania* 122(1): 255–266.

Scott, J. and Kirkpatrick, J. (2008). 'Rabbits, landslips and vegetation change on the coastal slopes of subantarctic Macquarie Island, 1980–2007: implications for management'. *Polar Biology* 31(4): 409–419.

Scott, J.J. and Kirkpatrick, J.B. (2013). 'Changes in the cover of plant species associated with climate change and grazing pressure on the Macquarie Island coastal slopes, 1980–2009'. *Polar Biology* 36(1): 127–136.

Selkirk, P., Seppelt, R. and Selkirk, D. (1990). *Subantarctic Macquarie Island: Environment and Biology*. Cambridge: Cambridge University Press.

Shaw, J., Terauds, A. and Bergstrom, D. (2011). 'Rapid commencement of ecosystem recovery following aerial baiting on sub-Antarctic Macquarie Island'. *Ecological Management and Restoration* 12(3): 241–244.

Smith, V.R. and Steenkamp, M. (1990). 'Climatic change and its ecological implications at a subantarctic island'. *Oecologia* 85: 14–24.

Smith, V.R. and Steenkamp, M. (1992). 'Soil macrofauna and nitrogen on a sub-Antarctic island'. *Oecologia* 92(2): 201–206.

Smith, V.R. (2007). 'Introduced slugs and indigenous caterpillars as facilitators of carbon and nutrient mineralisation on a sub-Antarctic island'. *Soil Biology and Biochemistry* 39: 709–713.

Smith, V.R. (2008). 'Energy flow and nutrient cycling in the Marion Island terrestrial ecosystem: 30 years on'. *Polar Record* 44(3): 211–226.

Springer, K. (2016). 'Methodology and challenges of a complex multi-species eradication in the sub-Antarctic and immediate effects of invasive species removal'. *New Zealand Journal of Ecology* 40(2): 273–278.

St Clair, J.J.H. (2011). 'The impacts of invasive rodents on island invertebrates'. *Biological Conservation* 144(1): 68–81.

St Clair, J.J.H., Poncet, S., Sheehan, D.K., Székely, T. and Hilton, G.M. (2011). 'Responses of an island endemic invertebrate to rodent invasion and eradication'. *Animal Conservation* 14(1): 66–73.

Stevens, M., Hudson, P., Greenslade, P. and Potter, M. (2010). *Report on invertebrate monitoring of long term field sites on Macquarie Island 2009/2010*. Unpublished report. South Australia: South Australia Museum.

Terauds, A., Chown, S.L. and Bergstrom, D.M. (2011). 'Spatial scale and species identity influence the indigenous–alien diversity relationship in springtails'. *Ecology* 92(7): 1436–1447.

Terauds, A., Doube, J., McKinlay, J. and Springer, K. (2014). 'Using long-term population trends of an invasive herbivore to quantify the impact of management actions in the sub-Antarctic'. *Polar Biology* 37(6): 833–843.

Thoresen, J.J., Towns, D., Leuzinger, S., Durrett, M., Mulder, C.P. and Wardle, D.A. (2017). 'Invasive rodents have multiple indirect effects on seabird island invertebrate food web structure'. *Ecological Applications* 27(4): 1190–1198.

Towns, D.R., Wardle, D.A., Mulder, C.P., Yeates, G.W., Fitzgerald, B.M., Parrish, R.G., Bellingham, P.J. and Bonner, K.I. (2009). 'Predation of seabirds by invasive rats: Multiple indirect consequences for invertebrate communities'. *Oikos* 118(3): 420–430.

Treasure, A.M., Chown, S.L. and Diez, J. (2014). 'Antagonistic effects of biological invasion and temperature change on body size of island ectotherms'. *Diversity and Distributions* 20(2): 202–213.

Vogel, M., Remmert, H. and Lewis-Smith, R.I. (1984). 'Introduced reindeer and their effects on the vegetation and the epigeic invertebrate fauna of South Georgia (subantarctic)'. *Oecologia* 62(1): 102–109.

Ward, D.F., New, T.R. and Yen, A.L. (2001). 'Effects of pitfall trap spacing on the abundance, richness and composition of invertebrate catches'. *Journal of Insect Conservation* 5(1): 47–53.

Watson, K.C. (1967). 'The terrestrial arthropoda of Macquarie Island'. *ANARE Scientific Reports, Series B (1) Zoology* 99: 1–90.

Whinam, J., Fitzgerald, N., Visoiu, M. and Copson, G. (2014). 'Thirty years of vegetation dynamics in response to a fluctuating rabbit population on sub-Antarctic Macquarie Island'. *Ecological Management and Restoration* 15(1): 41–51.

Włodarska-Kowalczuk, M. and Kędra, M. (2007). 'Surrogacy in natural patterns of benthic distribution and diversity: selected taxa versus lower taxonomic resolution'. *Marine Ecology Progress Series* 351: 53–63.

Woodcock, B.A. (2005). 'Pitfall Trapping in Ecological Studies'. In: S. Leather (ed.) *Insect Sampling in Forest Ecosystems*, pp. 37–57. Oxford: Blackwell.

Work, T.T., Buddle, C.M., Korinus, L.M. and Spence, J.R. (2002). 'Pitfall trap size and capture of three taxa of litter-dwelling arthropods: Implications for biodiversity studies'. *Environmental Entomology* 31(3): 438–448.

Zou, Y., Jinchao, F., Dayuan, X., Weiguo, S. and Axmacher, J.C. (2012). 'A comparison of terrestrial arthropod sampling methods'. *Journal of Resources and Ecology* 3(2): 174–182.

Introduction of biological control agents against the European earwig (*Forficula auricularia*) on the Falkland Islands

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Abstract The Falkland Islands (FI), as with many island ecosystems, is vulnerable to invasive species, which can cause wide ranging social and environmental consequences. Control of invasive species is widely recognised as a priority, but there have never been attempts to use classical biological control (CBC) for this purpose in FI. The European earwig was recently introduced to the FI and has since become abundant in the Stanley area and some other settlements on the islands. Earwigs now cause considerable damage to garden crops and also pose a number of health hazards. There are also concerns that earwigs have started to spread into grasslands and irreversibly alter this important native ecosystem. After extensive stakeholder consultations it was decided to use the invasive earwigs as a case study for the introduction of CBC to the FI. Based on previous work on earwig control, supplemented by additional host range testing, two tachinid flies, *Triarthria setipennis* and *Ocytata pallipes*, were selected as the most suitable control agents for the Falkland Islands. Extensive awareness raising activities, focusing on the threat of invasive species, benefits and risks of CBC, secured the support of the wider public to go ahead with the release of both control agents during 2015 and 2016. Major challenges encountered during the release process were the need to install makeshift quarantine facilities and the switchover of the life-cycle of both control agents to southern hemisphere seasons.

Keywords: awareness raising, invasive species, *Ocytata pallipes*, tachinid flies, *Triarthria setipennis,* UK Overseas Territories

INTRODUCTION

The European earwig, (*Forficula auricularia*, Order Dermaptera) is widely regarded as a beneficial predator of insect pests in fruit orchards within its native range of Europe and West Asia (Nicholas, et al., 2005; Dib, et al., 2010), however outside this range there are reports that this species can cause significant agricultural problems (Kuhlmann, et al., 2001). In 1997/1998 the European earwig was reliably recorded for the first time in the Falkland Islands (FI), an archipelago in the South Atlantic Ocean, around 500 km off the southern Patagonian coast of South America. Since then the earwig has become a significant pest on the islands causing damage to garden and greenhouse plants and leading to a halt in the production of a number of commercial crops (Maczey, et al., 2012). The earwigs are also posing a number of health hazards, particularly in autumn (March/April) when they invade buildings in large numbers. They have been found in asthma inhalers and beneath the seals of oxygen masks causing the local hospital to introduce additional safety procedures checking equipment for the presence of earwigs prior to use. Many households currently spend substantial amounts of money to control earwigs primarily by having the foundations of their houses sprayed with pesticides once or twice a year. Since its introduction the earwig has become common in the Stanley area and a number of settlements in the wider countryside. There is also concern they may spread into native grasslands, with a risk of irreversibly altering the indigenous ecosystem posing a particular threat to a high number of endemic arthropods (Maczey, et al., 2012).

Classical biological control (CBC) has the potential to offer effective, economic and sustainable control of this invasive species. This method involves the deliberate release of specialist natural enemies – mainly insects and fungi – from the invasive's native range. The aim is to reduce the abundance of problem species in its introduced range below an ecological or economic threshold. The European earwig is a promising target species for CBC on the FI, particularly as chemical sprays are ineffective, and because of its great mobility (Santini & Caroli, 1992). Off the shelf solutions using parasitoid tachinid flies from Europe, including Great Britain, are feasible. One such species *Triarthria setipennis* has established successfully in British Columbia and Newfoundland where studies have indicated a considerable reduction in earwig numbers, most probably due to high levels of parasitism in the mid-1970s (Morris, 1984). However, since 1978, no further evaluation of parasitoid impact has been undertaken. A second species of parasitoid, *Ocytata pallipes*, was introduced into Canada to control the European earwig during the 1990s but no monitoring took place and establishment is unknown (Kuhlman, et al., 2001). *Ocytata pallipes* and *T. setipennis* have also been introduced into the USA as early as the 1920s (Oregon) and also into New Zealand (Kuhlman, et al., 2001). Again, little is known about the success of these releases.

During a workshop in Stanley in March 2012 there was consensus on the feasibility of biological control of invasive non-native species on the South Atlantic UK Overseas Territories and that the European earwig would be a target well suited for CBC in the FI. Experts working on invasive species on the FI and also members of the general public saw an urgent need for sustainable control of this species. Equally, the Government of South Georgia saw this as an opportunity to reduce the risk of future introductions of earwigs to South Georgia. The Falkland Island Government (FIG) therefore decided to commission a host range testing programme to assess the safety and suitability of two parasitoid flies, believed to be host specific to the European earwig, for introduction into the FI (Maczey, et al., 2016).

No native earwig species inhabit the FI, therefore host range tests were conducted on insect species (crickets and cockroaches) representing insect orders which are closely related to earwigs. The Falklands have one native species of cricket, the camel cricket (*Parudenus falklandicus*). The results showed that there was no indication that either of the two assessed fly species (*O. pallipes* and *T. setipennis*) can develop or otherwise impact on the viability of any of the test species, even when artificially forced to ingest parasitoid eggs or inoculated with fly larvae, which would rarely happen under natural conditions (Maczey, et al., 2016). The tests confirmed our opinion that there would be

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no risks to non-target species if one or both of these highly earwig-specific tachinid fly species were released on the FI (Maczey, et al., 2016).

Based on the results from the host range testing, the Environmental Committee and Executive Council of the FIG decided to go ahead with the release of both agents, provided there was sufficient support from the wider public. Up until this point stakeholder acceptance for the introduction of a new species into the FI had not been assessed and the Environmental Committee decided to conduct a range of awareness raising activities to encourage residents to voice their concerns and engage in open discussion on the safety and scope of CBC of earwigs. This paper covers both the outcomes of the awareness raising activities and the results of the subsequent release of the control agents conducted between 2015 and 2017.

MATERIALS AND METHODS

Stakeholder consultation

We wanted to engage with the residents of the FI to understand whether biological control, in general, and the release of two parasitoid fly species, in particular, was of any concern or would be largely welcomed by the general public and/or experts and scientists working in conservation or agriculture on the FI.

At the core of all consultations with stakeholders we communicated four major premises:

- The release of the control agents is safe and does not pose any risks for native species, human health or food production, in contrast to the current use of large quantities of a highly toxic pesticide (Demand® CS).
- Costs for release would largely be covered through secured funding from Defra (Darwin Initiative).
- Although we saw no major hurdles for a successful establishment of both fly species, establishment can never be guaranteed, and this could be a reason for failure.
- Equally, if successful establishment has taken place, the amount of control exerted by the released agents is difficult to predict. Although in the absence of hyperparasitoids (in this case parasitic wasps known to develop inside the pupal stage of the tachinid flies) the likelihood for a good control is high, this is something which cannot be predicted with absolute certainty.

Stakeholder consultation regarding the biological control of earwigs focused on three main steps:

Providing initial background information

Information explaining the principles of biological control and the safety testing of the proposed control agents, including a 'frequently answered questions' (FAQ) section, was made available on the FIG website. In addition, a two-page flyer providing information on our work was distributed throughout Stanley prior to any public events.

Advertising opportunities to get more detailed information and voice any concerns

Website and flyer announcements were made of the dates for presentations and opportunities for open discussions. The documents also gave contact e-mails to arrange meetings or discussions outside these dates or to voice any concerns via e-mail. Times and locations for all events were also broadcasted by radio and announced in the local newspaper. An invitation to add to the FAQ was also given on the website. Presentations given – one broadcasted by local TV – also included invitations to forward any questions or concerns to the project team.

To present CABI's work on earwig control and engage with the public

Aside from the widely advertised events, discussions with residents took place on many other occasions. These included meetings with pest controllers, members of the legislative assembly (MLAs), scientists from government departments and NGOs, teachers, commercial growers and farmers. Discussions continued after FIG was confident enough that it would have the backing of the public for the release of the control agents throughout the length of the project and also included direct demonstrations of the activities at the release facilities.

Release programme

In the native range, rates of parasitism by *T. setipennis* and *O. pallipes* vary considerably between meta-populations of earwigs, and large numbers of the host need to be collected to obtain sufficient parasitoids for release and establishment. There are no estimates of how many individuals need to be released to achieve the formation of a parasitoid population in a new environment, but as a general rule the more individuals are released the better the chances are for establishment.

Trapping of earwigs took place in orchards in England during 2015 and 2016. Sites selected for collecting focused on locations combining ease of access with high numbers of specimens, both of earwigs and parasitoids likely to be obtained. Trapping involved installing flowerpots, 10 x 10 x 17 cm, filled with egg cartons, into trees 1 to 2 m above ground. Distribution of traps and the collecting regime are given in Table 1. Earwigs were collected at roughly monthly intervals three times per year.

Collected earwigs were kept in 40l plastic containers, housing no more than an estimated 2,500 earwigs per container. Egg cartons were used to provide hiding places and lids were fitted with netted openings to give sufficient aeration. The edges of containers were covered with

Table 1 Earwig/parasitoid collecting regime.

Site	No. of traps 2015	No. of traps 2016	Setup date 2015	Setup date 2016	Collecting period 2015	Collecting period 2016
Silwood Park, Berkshire	43	31	29/6/15	27/5/16	29/7/15 – 29/9/15	26/7/16 – 21/9/16
South Darenth, Kent	230	200	03/7/15	25/5/16	28/7/15 – 10/9/15	15/7/16 – 20/9/16
East Malling Research, Kent (EMR)	300	325	13/7/15	01/6/16	5/8/15 – 25/9/15	11/7/16 – 16/9/16
Target Farm, Marden, Kent	-	160	-	23/5/16	-	13/7/16 – 20/9/16

Fluon® to prevent earwigs escaping. Food consisted of a mixture of vegetables (lettuce and carrots) and dry dog food applied three times a week. The earwigs were kept for a period of six to eight weeks and, afterwards, when the majority of parasitoid larvae had left their hosts, were released back at the trapping sites.

Earwig cultures in the lab were checked for parasitoid pupae three times a week. Pupae were separated to species and stored in glass tubes sealed with a mesh cover to allow aeration whilst preventing any potential hyperparasitoids from escaping (Fig. 1). The tubes were then placed inside a larger plastic container with a meshed opening to allow for aeration. Inside this plastic container moistened tissue facilitated high humidity to prevent desiccation of the pupae.

In 2015 all pupae were stored at 16˚C until mid-September, afterwards *O. pallipes* at 12˚C and *T. setipennis* at 8–10˚C until their shipment to Stanley in November 2015. In 2016, pupae of *O. pallipes* were stored at 10–12˚C. At this temperature hatching was delayed long enough to allow two separate shipments to Stanley in August and September. Pupae of *T. setipennis*, which hibernates in this stage, were stored at room temperature (18–20˚C) to mimic natural conditions. From October onwards, the pupae of *T. setipennis* were kept at 10˚C to simulate more natural overwintering conditions.

On arrival at Stanley, sealed storage boxes containing vials with pupae were stored in a specifically developed quarantine shed (details provided at: <http://www.darwininitiative.org.uk/project/DPLUS033/>) and kept at 20˚C to trigger hatching. Quarantine facilities were used as a safety precaution in case hyperparasitoids had contaminated the fly cultures. Hatched flies were transferred into rearing tents located in a polytunnel on a daily basis. *O. pallipes* were kept there for mating and depositing of micro-eggs on carrot pieces previously exposed to earwigs, so that they had obtained the scent of the host species. The carrot pieces contaminated with fly eggs were then fed to locally collected earwigs. After inspection confirmed that most fly eggs had been ingested by earwigs, these were released at sheltered locations in Stanley with high densities of earwigs. Adult flies of *T. setipennis* were kept only 4–5 days in rearing tents, to allow mating, after which they were released at sheltered locations with high host densities.

Fig. 1 Pupae of *T. setipennis* inside their storage containers.

RESULTS

Stakeholder consultation

Attendance of public events varied from only three visitors on one occasion to up to 30 visitors during the demonstration of the release facilities. Feedback after presentations centred on the safety of CBC. Most frequent questions were: whether the release control agents could replace one nuisance species with a second one; or what the flies would feed on once earwigs went down in numbers. Our impression was that within the attending audience it was relatively straightforward to dispel such concerns by explaining in more detail the host specificity and dependence of the control agents on host density levels and that CBC will not lead to complete eradication of the target species.

People were relieved when seeing the small size of pinned specimens of the agents passed around, having expected something much larger. Worries about flies invading buildings could be dispelled by pointing out that these species, in contrast to house flies and other species, will not actively be attracted to houses. Some gardeners worried that eggs or larvae of the biological control flies would end up on vegetables; although not being a health hazard in any way this was seen as unpleasant. The answer to this was that the flies will deposit eggs and larvae only on items already smelling strongly of earwigs and in the case of food items these would be already heavily damaged crops beyond consideration for human consumption.

Repeatedly, residents raised general concerns about the continued use of pesticides. Worries about the build-up of resistance, has already led to changed usage of different products. There were also concerns that spraying may temporarily reduce earwig densities to a satisfactory level which in turn could result in diminished support for CBC. However, most residents seemed to be aware of natural fluctuations and also that earwig numbers would be likely to increase when the use of pesticides is reduced. Several times the decline in native ‘black beetles’ (a species of rove beetle, Staphylinidae) was pointed out, which was also attributed to the use of pesticides. The loss of native ‘black beetles’ was mostly regretted but on occasion the intrusion of insects of any kind into buildings was seen as undesirable. On occasion it was suspected that the decline of native species was caused by the earwigs themselves and related to a scarceness of such species in areas with high earwig densities.

Frequently, people had questions about possible obstacles to the establishment of the control agents. Comments were made on the possible impact the current use of pesticides may have on the establishment and efficacy of the control agents. Pesticides are mostly applied in autumn when the *T. setipennis* will only be present as dormant pupae. However, spraying may still have some impact on the *O. pallipes*, which overwinters as larvae inside living earwigs. Given the climatic conditions on the FI the majority of earwigs will still overwinter outside and therefore escape pesticides. We expect that as the flies begin to establish and gradually start to control the population of earwigs in Stanley the need for spraying will reduce so impacting less on both earwigs and flies. There was also concern about the availability of flowers providing pollen and nectar for adult flies, something we believe is not a problem during the time period when adult flies occur during late spring and summer. Generally, the audience was also keen to reconstruct the history of earwig introduction with various speculations on time and entry points being discussed. There was general agreement that the biological control will support a reduction in demand for chemical treatment both reducing costs and risks for human health and the environment.

Release programme

During 2015, an estimated 50,000 earwigs were collected in the UK. In 2016 numbers dropped to 18,500 earwigs despite an increase of traps from 573 to 716. Earwig densities peaked in mid-August with the majority collected up to this time still being larvae. In each year, numbers dropped considerably until the end of the collecting period at the end of September.

A total of 147 pupae of *T. setipennis* and 237 of *O. pallipes* were obtained from the earwigs up to 28 October 2015. Discounting prematurely hatched flies, altogether 145 pupae of *T. setipennis* and 212 of *O. pallipes* were shipped to Stanley for a first release trial on the Falkland Islands in November 2015. In 2016, 358 pupae of *T. setipennis* and 284 of *O. pallipes* were collected until 21 December. Discounting prematurely hatched flies, a total of 256 pupae of *T. setipennis* were shipped to Stanley in January 2017, and 225 of *O. pallipes* in August and September 2016. A breakdown of collected parasitoids per site and estimated parasitism rates is given in Table 2.

In November 2015, hatching rates of *O. pallipes* at quarantine facilities in the FI were poor, with all flies dying shortly after emergence. The most likely cause for this was the prolonged storage of fly pupae under cold conditions prior to the release, which aimed to synchronise hatching with the onset of summer in the southern hemisphere. At the same time *T. setipennis* did not hatch at all and emergence only started in January/February 2016. Only a few flies hatched over several weeks, which were kept in the mating tents (Fig. 2) and, after six days, altogether only 15 flies were released into an open polytunnel containing high densities of earwigs.

Shortened storage periods for *O. pallipes* and prolonged hibernation of *T. setipennis* allowed a significantly improved hatching rate in 2016. More than 200 *O. pallipes* hatched in August and September 2016. They mated and subsequently deposited a large number of micro-eggs on carrot pieces which had previously been exposed to earwigs. 1,800 earwigs collected locally were then fed with pieces of carrots contaminated with fly eggs and released in Stanley in October. From 256 pupae of *T. setipennis* transported to Stanley in January 2017, 185 flies hatched during February. Some flies died within a short period after hatching, but a large proportion were released into sheltered places in Stanley.

Fig. 2 Dave Moore demonstrating the fly rearing tents during open day at Government House gardens, Stanley in Nov. 2015 (photo: Sharon Jaffray, Penguin News).

DISCUSSION

Stakeholder consultation

This was the first introduction of a non-native species for the control of an invasive species on the FI, and a certain level of concern from expert stakeholders and the general public was anticipated. Therefore, we tried to encourage residents to voice their concerns and engage in open discussion on the safety and scope of CBC. At the core of all consultations were these premises:

The release of the control agents is safe and does not pose any risks for native species, human health or food production

Both successful establishment of CBC agents and the amount of control they can exert can never be guaranteed and these can be a reason for failure.

The general feedback most people gave was that of cautious optimism and being in favour for biological control provided it is safe. It was important for most people to have the assurance that biological control does not lead to the introduction of a species which could become problematic. We believed that through in-depth discussions worries and concerns could largely be dispelled. People became willing to trial a release hoping that it would provide the anticipated long-term solution to the earwig problem, whilst being fully aware that there remains a certain risk of failure. However, this was only partly driven by direct support of CBC versus an equal measure of concern about risks and side-effects associated with the current use of toxic pesticides.

Compared to the amount of advertising preceding public events, the overall turnout was ~1% of the population of

Table 2 Earwigs, parasitoids and % parasitism recorded in 2015 and 2016.

Site/orchard	Year	Earwigs collected	*T. setipennis*	*O. pallipes*	% parasitism *T. setipennis*	% parasitism *O. pallipes*
Darenth	2015	3,000	16	3	0.5	0.1
Darenth	2016	2,800	49	6	1.8	0.2
Silwood	2015	1,000	6	0	0.6	0.0
Silwood	2016	950	52	8	5.5	0.8
EMR total	2015	46,000	125	234	0.3	0.5
EMR total	2016	10,300	149	234	1.4	2.3
Target farm	2016	4,250	108	36	2.5	0.8

the Falklands and thus relatively low (although one might consider drawing in 1,000 attendees in four events in a large town of 100,000 a very good turnout). Attendance during the first open day at the release facilities was (~30 visitors) comparably high and attracted the attention of local radio and television. However, this dropped significantly in the second release year, going down to just a handful of visitors. The same was true for other types of engagement towards the end of the project. Once initial concerns were dispelled there was increasingly less new information between individual events, both from the side of release activities and any residual concerns to be shared. This may have resulted in a declining interest or possible increasing acceptance by the public over time compared to the start of the project.

Release programme

Despite intensified efforts, earwig trapping in England during 2016 yielded less than half the numbers of earwigs obtained in the previous year, which was mainly due to a drastic population crash in a single cherry orchard at East Malling Research. In addition, earwig densities at Target farm varied strongly throughout the year with few earwigs being collected in September. Low earwig numbers in 2016 were offset by a recovery of parasitoid populations, which had been very low in 2015. In both years the quantity of *T. setipennis* pupae collected was substantially lower compared to 1,000+ pupae collected from 20,000 earwigs in 2013 when the host range testing took place (Maczey, et al., 2016). It remains unclear whether *T. setipennis* suffered a population crash in 2015 or if the collecting sites chosen in 2015/2016 were more generally characterised by low rates of parasitism. Studies in continental Europe recorded, on average, higher rates of parasitism for this species (Kuhlmann, 1995). In both years, although a few individuals emerged very early in the season, most *T. setipennis* pupae were found from the beginning of September onwards. This coincides with field observations of some pupae very early in the season in England indicating a more pronounced second generation compared to its phenology on the continent, where occurrence of pupae peaks in August (Kuhlmann, 1991). Collecting earlier would not have yielded more pupae for release though, as these mostly emerged early without a hibernation period, far too early for a release in the FI.

The low number of collected parasitoids and a low hatching rate in Stanley in 2015 was not sufficient to enable establishment of either of the two species. One major problem was switching the lifecycle from a northern hemisphere rhythm to the seasons in the FI. The lack of synchronisation of life cycles between the northern and southern hemisphere is a well-documented problem in biological control (Waterhouse & Sands, 2001; De Clerck-Floate et al., 2008). *Ocytata pallipes* normally remains in the pupal stage only for a short period and initially we tried to delay hatching until the Falkland summer through storage at lower temperatures, hoping to slow down development. However, the species does not tolerate being stored for long periods at low temperatures resulting in poor survival rates. In 2016, this was addressed by shipping pupae of *O. pallipes* to Stanley several times between August and October. Release in Stanley during late winter relied on creating a suitable local environment to allow it to parasitise earwigs soon after arrival. Therefore, flies were kept in an artificially heated polytunnel warm enough to allow both earwigs and flies to be active during the winter months.

The first release trial for *T. setipennis* also failed but for a different reason. November was too early to break the dormancy of this species, which hibernates in the pupal stage, and early exposure to elevated temperatures (20°C) only led to unsynchronised emergence in January/February. For the second release, pupae were kept at low temperatures until mid-January. This resulted in a much better synchronised hatching whilst still allowing a sufficiently long period during the summer in the FI for the completion of a full life-cycle.

The adapted methodology led to much improved results and both fly species were successfully released, albeit with lower numbers than initially hoped for. For *O. pallipes*, this was mitigated in 2016 by keeping hatched flies initially in cages up to the point of eggs being deposited and releasing larger numbers of earwigs fed with contaminated pieces of carrots. The ecology of *T. setipennis* does not allow a similar approach, but for this species hatching rates had strongly improved compared to the previous year and the chances for mating were increased by keeping this species caged for six days before the release.

At this stage of the release programme we do not know whether either or both fly species have established. If establishment has been successful, it is still far too early to observe an impact on earwig numbers and this will only become apparent during future years.

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REFERENCES

De Clerck-Floate, R., Hinz, H.L., Heard, T., Julien, M. Wardill, T. and Cook, C. (2008). 'Preliminary Results of a Survey on the Role of Arthropod Rearing in Classical Weed Biological Control'. In: M.H. Julien, R. Sforza, M.C. Bon, H.C. Evans, P.E. Hatcher, H.L. Hinz and B.G. Rector (eds.) *Proceedings of the XII International Symposium on Biological Control of Weeds*, pp. 528–534. Wallingford, UK: CABI Publishing.

Dib, H., Simon, S., Sauphanor, B. and Capowiez, Y. (2010). 'The role of natural enemies on the population dynamics of the rosy apple aphid, *Dysaphis plantaginea* Passerini (Hemiptera: Aphididae) in organic apple orchards in south-eastern France'. *Biological Control* 55: 97–109.

Kuhlmann, U. (1991). 'Zur Ökologie und Biologie zweier Raupenfliegen (Diptera: Tachinidae) als Parasitoide des Gemeinen Ohrwurms (*Forficula auricularia*, Dermaptera)'. *Diploma thesis*. Kiel, Germany: Christian Albrechts Universität.

Kuhlmann, U. (1995). 'Biology of *Triarthria setipennis* (Fallén) (Diptera: Tachinidae), a native parasitoid of the European earwig, *Forficula auricularia* L. (Dermaptera: Forficulidae), in Europe'. *Canadian Entomologist* 127: 507–517.

Kuhlmann, U., Sarazin, M.J., O'Hara, J.E., Mason, P.G. and Huber, J.T. (2001). '*Forficula auricularia* L., European Earwig (Dermaptera: Forficulidae)'. In: P.G. Mason and J.T. Huber (eds.) *Biological Control Programmes in Canada, 1981–2000*, pp. 127–131. Wallingford, UK: CABI Publishing.

Maczey, N., Tanner, R., Cheesman, O. and Shaw, R. (2012). *Understanding and Addressing the Impact of Invasive Non-native Species in the UK Overseas Territories in the South Atlantic: A Review of the Potential for Biocontrol*. Project report, DEFRA CR 0492. <http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&ProjectID=17564&FromSearch=Y&Publisher=1&SearchText=wc1001&SortString=ProjectCode&SortOrder=Asc&Paging=10#Description>

Maczey, N., Edgington, S., Moore, D. and Haye, T. (2016). 'Biology and host range testing of *Triarthria setipennis* and *Ocytata pallipes* (Diptera: Tachinidae) for the control of the European earwig (*Forficula auricularia*)'. *Biocontrol Science and Technology* 26: 447–461.

Morris, R.F. (1984). '*Forficula auricularia*, European Earwig (Dermaptera: Forficulidae)'. In: J.S. Kelleher and M.A. Hulme (eds.) *Biological Control Programmes Against Insects and Weeds in Canada 1969–1980*, pp. 39–40. Slough, UK: Commonwealth Agriculture Bureaux.

Nicholas, A.H., Spooner-Hart, R.N. and Vickers, R.A. (2005). 'Abundance and natural control of the woolly aphid *Eriosoma lanigerum* in an Australian apple orchard IPM program'. *Biological Control* 50: 271–291.

Santini, L. and Caroli, L. (1992). 'Damage to fruit crops by European earwig (*Forficula auricularia* L.)'. *Informatore-Fitopatologico* 42: 35–38.

Waterhouse, D.F. and Sands, D.P.A. (2001). *Classical Biological Control of Arthropods in Australia*. Monograph No. 77. Canberra, Australia: ACIAR.

Assessment of snail exposure to the anticoagulant rodenticide brodifacoum in the Galapagos Islands

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Abstract Eradication of invasive rodents has become a powerful tool to protect native island biota. Use of brodifacoum, an anticoagulant rodenticide, has contributed to hundreds of successful invasive rodent eradication efforts on islands. Application of bait containing brodifacoum for this purpose requires appropriate consideration of adverse effects on non-target wildlife. Thus, a priori identification of non-target risks and, where needed, approaches to mitigate these to acceptable levels, is now an essential component of eradication planning and implementation. As part of the plan for eradicating invasive rats and mice from Floreana Island in the Galapagos, we experimentally tested the effect of brodifacoum on the Galapagos endemic land snail species *Naesiotus unifasciatus*. Importantly, the trials were designed to evaluate effects of particular components of the bait pellets, namely the active brodifacoum, the pyranine biomarker, and a blue dye. We found no evidence for increased snail mortality following exposure to any of these bait components. We review results of past toxicity studies on terrestrial molluscs and find that, as for our own study, there is likely to be little impact of anticoagulant rodenticide on terrestrial mollusc survival as the result of application of brodifacoum bait. However, given the limited taxonomic representation in the toxicity tests performed on terrestrial molluscs so far, we recommend the continued used of captive toxicity trials to assess potential effect of any rodenticide applications on native malacological fauna on a case-by-case basis where large-scale eradication programmes are planned and undertaken.

Keywords: anticoagulant, brodifacoum, Bulimulidae, islands, restoration, rodent eradication

INTRODUCTION

Invasive mammal eradications are powerful conservation tools to protect biodiversity and prevent extinctions on islands (Lorvelec & Pascal, 2005; Bellingham, et al., 2010; Nogales, et al., 2013). Three rat species (*Rattus rattus*, *R. norvegicus*, *R. exulans*) and house mice (*Mus musculus*) are the most common rodents introduced to islands worldwide (Atkinson, 1985). These species are responsible for population declines and extinctions of insular flora and fauna, and they are known to interrupt ecosystem processes with negative cascading effects (Fukami, et al., 2006; Steadman, 2006; Towns, et al., 2006; Jones, et al., 2008; Kurle, et al., 2008; Varnham, 2010; Dunlevy, et al., 2011; St Clair, 2011). To recover endangered species and restore ecosystem processes, invasive rodents on islands are increasingly targeted for eradication, with at least 637 successful rodent eradications to date (based on DIISE island data ranked as good or satisfactory; DIISE, 2015). Ninety-seven percent of successful rodent eradications have involved the use of rodenticide, with brodifacoum having been used in 76% of them.

The common mode of toxicity of anticoagulant rodenticides in mammals and birds is to inhibit Vitamin K metabolism in liver, which in turns prevents the formation of chemical factors essential to blood coagulation (e.g., Rattner, et al., 2014). In mammals and birds a lethal exposure will cause these clotting factors to deplete to a level so that blood can no longer coagulate, resulting in death through internal haemorrhage (MacNicoll, 1993). ‘First-generation’ anticoagulants, such as warfarin, are most effective against rodents in multiple feeds but their intensive use as rodenticides resulted in the development of heritable resistance in some rodent populations (Rattner, et al., 2014). This prompted the development of the more potent ‘second generation’ anticoagulants, such as brodifacoum, which are effective against target rodents in a single feed (Rattner, et al., 2014). Sublethal or chronic effects of anticoagulants are not well described in wildlife (Rattner, et al., 2014), but sublethal exposure may result in the retention of residual anticoagulant concentrations in liver tissue. In this regard the ‘second generation’ anticoagulant rodenticides are more persistent in animal tissues, especially liver, than ‘first-generation’ anticoagulants (Fisher, et al., 2003). The second generation anticoagulant brodifacoum has been the most commonly used rodenticide for eradicating invasive rodents from islands, with a high success rate (Howald, et al., 2007; Parkes, et al., 2011; DIISE, 2015). Brodifacoum, incorporated at 20–50 ppm (0.002–0.005%) into cereal or wax baits, is applied to every rodent territory via bait stations, or broadcasted by hand or from a modified agricultural spreader bucket suspended from a helicopter. Large-scale broadcast of bait has facilitated increasingly large and complex island restoration projects involving the eradication of invasive rodents (e.g., Towns & Broome, 2003), but it also raises concerns about environmental contamination and adverse effects on non-target wildlife (Pain, et al., 2000; Eason, et al., 2002). Thus, *a priori* identification of non-target risks and the potential mitigation of these to acceptable levels is now an essential step to inform feasibility of large-scale eradication projects.

Invasive rats are known to prey upon terrestrial invertebrates (e.g., St Clair, 2011). On Galapagos, endemic land snails are particularly vulnerable (Clark, 1981), and recent field collections of land snail shells suggest that rats are particularly voracious snail predators on Floreana (Parent, unpublished data). Although the eradication of invasive rodents would likely benefit terrestrial molluscs, the potential impact of bait on non-target species should be evaluated. Indeed, a range of terrestrial invertebrate species, including snails and slugs, have been found to feed on cereal-based baits used for rodent control (e.g., Spurr & Drew, 1999; Johnston, et al., 2005). Reports that bait containing brodifacoum caused mortality in captive introduced and endemic snails (*Achatina fulica* and *Pachnodus silhouettanus*) from the Seychelles Islands that fed on the baits, and suspected field mortality of *Pachystyla bicolor* snails following operational baiting

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(anecdotally reported in Gerlach & Florens (2000a; 2000b) and Gerlach (2005)) raised concerns for other native and endemic snail species on islands where rodent eradication using anticoagulants was proposed. Limited information suggests that invertebrates are generally less susceptible to brodifacoum toxicity than mammals and birds (Booth, et al., 2001; Eason & Spurr, 1995), but current knowledge of snail physiology is insufficient to predict with confidence its effect on snails. To assess the feasibility of using brodifacoum to eradicate rats and mice from Floreana Island in the Galapagos (Island Conservation, 2013), a need to investigate risk to endemic land snails was therefore identified.

Land snails are known for their remarkable diversity in island systems (Cameron, et al., 2013). On Galapagos Islands, the land snail fauna comprises 103 endemic species distributed in 13 genera. Approximately 80 species and subspecies belong to the genus *Naesiotus* (Family Bulimulidae) and form the most species-rich adaptive radiation of these islands (Parent, et al., 2008). Recent field and genetic work suggests that most (if not all) Galapagos bulimulid species are single-island endemics (Parent & Crespi, 2006; Parent, unpublished data). Twenty species (and eight subspecies) of endemic land snails are known from Floreana Island. Eight of these species are critically endangered, and three are endangered (IUCN, 2015), whereas others remain to be evaluated. Thus, given the conservation status of Floreana endemic snails, we identified the need to evaluate whether exposure of these endemic snails to the brodifacoum bait type proposed for rodent eradication was likely to cause mortality.

Four previous experimental studies, together assessing twelve species of terrestrial molluscs, have failed to find significant effect of brodifacoum exposure on individual short-term mortality. The only exception to this trend is the study reported by Gerlach & Florens (2000a; 2000b) mentioned above. Therefore, a precautionary approach demands that the effects of exposure to brodifacoum bait should be tested on island endemic snails prior to large-scale rodent eradication measures on islands. Importantly, the rodenticide baits are composed of more than 99% inert ingredients, most of which is compacted cereal grains but may include other inert ingredients such as dye or biomarkers. Past studies failed to explicitly test the effect of inert ingredients on land snails. Thus, the main objectives of the study are to: (1) test various bait formulations to identify which component(s) of the baits are responsible for any mortality that might be observed in land snails, and (2) review and synthesize the literature on experimental toxicity tests of bait-based rodenticides on terrestrial molluscs.

MATERIALS AND METHODS

Site description

Floreana Island is part of the Galapagos archipelago, which straddles the equator approximately 1,000 km off the western coast of Ecuador. The islands are oceanic and have never been connected to any continent. Floreana is volcanic in origin, and at 17,253 ha is the sixth largest island in the archipelago. The maximum elevation of the island is 640 m, and its generally conical shape results in two distinct habitat types: dry lowlands and lush central highlands, which meet and overlap to some extent into what is referred to as a transition zone (McMullen, 1999). Over 98% of the island land area is Galapagos National Park, with the remaining 2% divided between a small town in the lowlands and agricultural and pastoral areas in the highlands (DPNG, 2014). The island is home to an estimated 140 residents as of 2014.

Snail population

Snails were collected at Cerro Pajas on Floreana (Latitude: 01.2968°S, Longitude: 90.4559° W) in November 2012. The Cerro Pajas site was selected based on the relatively high density of snails found there (Parent, unpublished data). We collected snails opportunistically from leaf litter on the ground and from low (< 0.5 m) vegetation. We chose to collect snails near the ground because those snails would be more likely to encounter bait pellets on or near the ground. There are at least three Galapagos endemic species of land snails occurring at that particular site (*Naesiotus nux*, *N. unifasciatus*, and *Succinea brevior*; endangered, critically endangered, and unknown status, respectively), and these species are expected to be either detritivores or to consume algae and lichens scraped from the substrate. For the present study, we used adult individuals of *N. unifasciatus* since the population density of this species was the highest and we felt confident that our sampling would not impact the survival of the population at that particular location (100 adult individuals were collected, less than 5% of the individuals encountered over a period of approximately one hour). We did not collect specimens from any other snail species because their population density was such that we would have had to collect more than 5% of the adult individuals encountered for our experiments. It is important to note that the information gathered from our toxicity experiments will by far outweigh the potential detrimental effects of our population sampling of *N. unifasciatus*. Bulimulid snails are hermaphrodites and therefore sexing them was not applicable.

Experimental design

We housed snails in small cylindrical plastic containers (11 cm high, 17 cm and 14 cm in diameter at the top and base of the container, respectively) replacing lids with tightly covering mesh secured with rubber bands to prevent snail escapes. Two sheets of task wipe (Kimwipe ®) paper were placed on the bottom of each container and kept moist for the duration of the experiment. A small amount of litter from which the snails were collected was sifted and visually inspected to remove any other small invertebrates. Approximately 10 grams of sifted litter was added to each container as a source of natural food and shelter. Each container held five snails at the beginning of the experiment.

We used three types of pelleted bait as experimental treatments: (1) non-toxic baits containing a blue dye (well less than 1% of pellet content) that is a standard proprietary component of the bait formulation, (2) non-toxic baits containing pyranine (a fluorescent marker dye allowing easy detection of metabolized bait in snails' bodies, faeces and slime trails when exposed to ultraviolet (UV) light, also representing well less than 1% of pellet content); and (3) bait containing blue dye and 50 ppm brodifacoum. For treatment groups, one moistened bait pellet was placed in each container at the start of the trial. Control group containers had no pellets, but were otherwise the same as treatment containers. We prepared five containers per treatment and for the control group (total n = 100 snails). All containers were kept on Floreana, at sea level in the shade at ambient temperature (25-28°C). In parallel, we kept five pellets in a container under the same conditions but without snails to evaluate the effect of the containers on the pellets themselves. All containers were opened twice daily to increase circulation of fresh air and to remoisten the tissue paper and bait pellet by spraying water, as necessary.

The experiment was conducted in two parts; the first over 10 days during which all containers with snails were monitored daily for mortality. A 10-day period was selected as slightly longer than the four to eight days that bait pellets are likely to be available to snail populations in the Floreana highlands (Island Conservation, unpubl. data) and substantially longer than the maximum of 72 hours over which snail mortality occurred in the study reported by Gerlach & Florens (2000a and 2000b).

Snail activity was noted twice a day by recording whether the snails were immobile and firmly attached to substrate (i.e., estivating) or moving in the containers. Snails found estivating were moved onto the vegetation and sprayed with water; this reliably caused the snails to become active once again. Snails found dead were frozen immediately in individual vials to preserve tissue and be dissected later if any statistically significant mortality effects were to be detected in our study. In addition, containers were visually inspected daily by illumination with a UV light to detect any fluorescent traces of pyranine in the slime trails and faeces of the snails in containers of treatment 2 which would have indicated ingestion of this bait by the snails. Control containers were inspected in the same manner. Finally, we also monitored bait consumption by noting any changes to the surfaces of the bait pellets. At the end of the 10-day period, living snails from the two treatment groups using non-toxic bait types were returned to the location where they were collected.

In the second part of the experiment, the remaining snails in the control group and the treatment group with brodifacoum baits were monitored for an additional 11 days (for a total of 21 days, well in excess of the period over which signs of poisoning and mortality would have occurred in mammals) and any snails alive were euthanized by freezing at the end of this second part of the experiment to use in subsequent residue content analysis if any significant mortality was observed.

Statistical analyses

We used a logistic regression approach with a Bernoulli (binomial) distribution to evaluate the effect of each bait component on the survival of the snails. We used post-hoc tests to determine whether any of the bait components had a significant effect on snail survival. We implemented all statistical analyses in R version 3.1.0 (R Core Team, 2014).

RESULTS

Snails remained active (i.e., not estivating) throughout the experiment. Individuals were confirmed to consume baits, either by direct observation (snails on bait pellet) or evidence of blue dye or pyranine fluorescence in the snails' bodies, faeces or slime trails. We did not track each individual snail's consumption of bait, but the observed evidence suggested that most snails consumed or were in direct physical contact with bait when available.

The survival of snails over the course of the bait treatments did not differ significantly from the survival of snails in the control groups without baits (Fig. 1). Because our treatment groups did not represent all possible combinations of bait type (with/without brodifacoum, presence/absence of blue dye, and presence/absence of pyranine), we could not directly compare the individual effect of each of these bait components on the snails. However, we found that when all treatments were analysed simultaneously, none of the components had a significant effect on snail survival (Fig. 1). In contrast, in a post-hoc test for individual effects of each component, we found that the survival of snails exposed to bait containing pyranine was greater than the survival of snails exposed to bait without pyranine (Fig. 2; Welch two-tailed t-test for unequal sample size, $t = 3.056$, d.f. = 14, $P < 0.01$). The survival of the snails over 21 days in the containers with bait containing brodifacoum did not significantly differ from the control (Welch two-sample two-tailed *t*-test, $t = 1.497$, d.f. = 5.611, $P > 0.05$).

DISCUSSION

The goal of our study is to quantify the short-term impacts of anticoagulant rodenticide bait on Galapagos endemic land snails. Importantly, any potential short-term impact the bait might have has to be considered against the long-term benefits that rodenticide bait application can bring to terrestrial malacofauna. These potential benefits include, for example, release from invasive rodent predation and general habitat improvement.

Our results suggest that none of the baits tested were toxic to the snails over the 10-day exposure period (i.e.

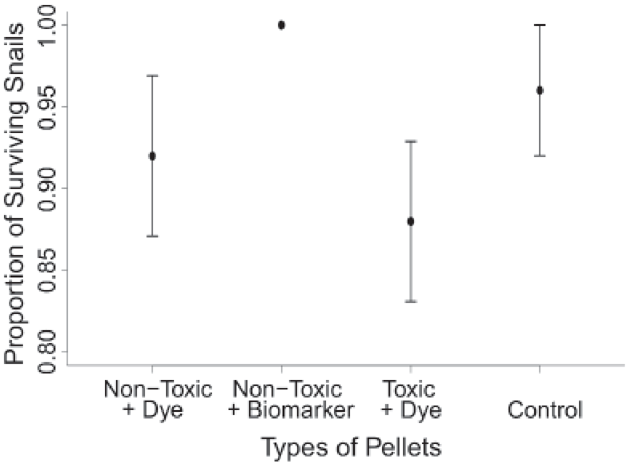

Fig. 1 Survival of snails exposed to different types of pellets: non-toxic pellets with blue dye, non-toxic pellets with biomarker, pellets with 50 ppm brodifacoum and blue dye, and control populations without pellets. Survival for all treatments was not significantly different than the survival of the snails in the control populations ($P > 0.05$). Error bars represent standard errors.

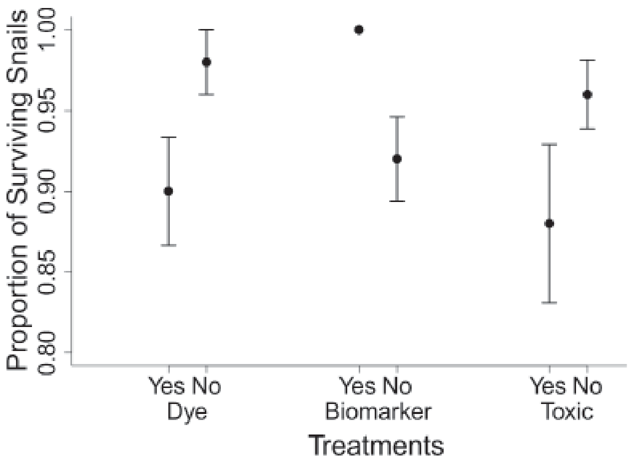

Fig. 2 Survival of snails as a function of presence/absence of pellet components: blue dye, biomarker, and brodifacoum. A logistic regression approach with a Bernoulli (binomial) distribution including all samples at once reveals no significant effect for any of the components ($P > 0.05$). However, a post-hoc t-test indicates that snails exposed to pellets with biomarkers have significantly higher survival than snails exposed to pellets without biomarker ($P < 0.01$). Error bars represent standard errors.

Table 1 Terrestrial mollusc species experimentally tested for the effects of anticoagulant rodenticides.

Species	Family	Locality	Use of control(s)?	Number of individuals tested	Anticoagulant rodenticide	Brodifacoum exposure time (max no of days)	Effect on organism survival	Reference
Oxychilus oglasicola	Zonitidae	Montecristo Island, Italy	Information not available	12	Brodifacoum 0.005%, bromadiolone	Information not available	Not significant	Sposimo et al., 2011
Ciliellopsis oglasae	Hygromiidae	Montecristo Island, Italy	Information not available	4	Brodifacoum 0.005%, bromadiolone	Information not available	Not significant	Sposimo et al., 2011
Deroceras laeve	Agriolimacidae	Hawaii, USA	yes	15	HACCO Ramik ® Green (0.005% diphacinone)	7	Not significant	Johnston et al., 2005
Oxychilus spp.	Zonitidae	Hawaii, USA	yes	15	HACCO Ramik ® Green (0.005% diphacinone)	7	Not significant	Johnston et al., 2005
Limax maximus	Limacidae	Hawaii, USA	yes	15	HACCO Ramik ® Green (0.005% diphacinone)	7	Not significant	Johnston et al., 2005
Orobophana solidula	Helicinidae	Henderson Is, South Pacific	yes	28	Pestoff 20R (0.002% brodifacoum)	10	Not significant	Brooke et al., 2011
Pacifella sp., Tornatellides sp., Lamellidae sp., Tubuaia hendersoni	Achatinellidae	Henderson Island, South Pacific	yes	43	Pestoff 20R (0.002% brodifacoum)	10	Not significant	Brooke et al., 2011
Pupisoma orcula	Pupilidae	Henderson Is, South Pacific	yes	1	Pestoff 20R (0.002% brodifacoum)	10	Not significant	Brooke et al., 2011
Helix aspersa	Helicidae	New Zealand	yes	24	Talon 20P (0.002% brodifacoum)	14	Not significant	Booth et al., 2003
Pachnodus silhouettanus, Achatina fulica	Cerastidae Achatinidae	Fregate Island, Seychelles	Information not available	Information not available	0.01 - 0.2mg brodifacoum	4	Mortality was observed but not statistically tested	Gerlach and Florens, 2000a & 2000b
Naesiotus unifasciatus	Bulimulidae	Floreana Island, Ecuador	yes	25	Bell Laboratories Brodifacoum 50D Conservation Blue FP2015 (0.005% brodifacoum)	21	Not significant	This study

survival of snails exposed to any bait treatment was not significantly different than 1.0). For the purposes of a conservative risk assessment, our experiment simulated a 'worst case' exposure to snails through use of 50 ppm brodifacoum in bait, compared to the lower concentrations proposed for the rodent eradication on Floreana (25 ppm) and used in previous similar experiments with snails (e.g., Booth, et al., 2003; Brooke, et al., 2011; Table 1). In confining snails under conditions favourable for foraging and in close proximity to bait, we also simulated a worst-case exposure potential, in comparison to the expected availability of bait to snails following an operational aerial application. Application rates for rodent eradication on Floreana Island remain to be determined, nonetheless relatively few snails are expected to encounter and consume bait before it is removed by other animals or breaks down naturally. Additionally, the operation will occur during the driest time of the year, corresponding to the time when snails are more likely to be estivating (Parent, unpublished data).

An apparent absence of toxic effects of brodifacoum bait on snails was further supported by survival being significantly higher in a post-hoc test in snails exposed to non-toxic baits that contained pyranine compared to snails exposed to baits without the biomarker during the first 10 days of the experiment. It is possible that one or some of inert components of the bait types used in our experiment provided a nutritional supplement benefiting the snails. Any such benefits from a boost in diet would become more evident over time. However, this pattern of increased survival did not carry over in snails that were kept for the full-length (21 days) of the experiment.

Our results add to a growing body of research suggesting that exposure to rodenticide bait formulations containing brodifacoum does not cause significant mortality in snails (Table 1). Reports by Gerlach & Florens (2000a; 2000b) and Gerlach (2005) appear to be exceptions, but are also brief and lacking in detail that would allow statistical evaluation. While the absolute toxicity of brodifacoum and its mechanism in snails remain to be established, in the context of potential exposure to rodenticide baits it is important to also consider the possible effects of other, nominally inert, ingredients of specific bait formulations (e.g., binders, preservatives, emulsifiers, pH regulating, flavouring or colouring agents). In designing our study we sought to account for some of these factors and recommend that future studies of the effects of rodenticides on invertebrates contain a mechanism to ensure any observed mortality is in fact due to the active anticoagulant agent and not other bait components or experimental conditions.

We caution that our tests were performed on a single species of land snail, and are therefore not extensive enough to confirm that exposure to brodifacoum bait would not have adverse effects in other terrestrial malacofauna on Galapagos or elsewhere. However, given that most Galapagos endemic snail species are of the same genus as the species tested here, we feel confident that at least this important group will not be affected by exposure to brodifacoum bait if it was applied for eradication of invasive rodents on Floreana and other Galapagos Islands. Most importantly, these snails are known to be consumed by introduced rats (Clark, 1981; Parent, unpublished data), and therefore eradication of invasive rodents is more likely to result in positive effects on Galapagos endemic snail populations.

Secondary exposure pathways must be considered when assessing non-target risk and when developing measures to prevent non-target mortality (Eason, et al., 1999). We did not test for residual brodifacoum concentrations in the bodies of exposed snails in our study, but Booth et al. (2003) measured brodifacoum residues in the bodies of some snails that had consumed bait. We expect that any snails that consume bait on Floreana Island could constitute a secondary exposure pathway for their predators such as some of the larger land birds. The only Galapagos birds that have been verified to be preying on endemic snails are the Galapagos mocking birds which have been extirpated from Floreana Island. There are no other known potential secondary exposure pathways for non-target species on Floreana Island involving the endemic snails as intermediate.

Evidence to date and our results indicate that rodent bait containing brodifacoum does not present a high risk of non-target mortality to terrestrial snails. However, our study is limited to the detection of mortality (i.e. we did not monitor for other potentially negative effects) and was over a short period of time. Given the general trend across terrestrial molluscs of the effect of brodifacoum on snail mortality, we recommend a re-evaluation of this effect for the species included in the study reported by Gerlach & Florens (2000a; 2000b) that would incorporate a more complete set of treatments and controls. More specifically, we recommend more toxicity tests on the invasive giant African snails (*Achatina fulica*) given its broad distribution (tests could be conducted in a range of localities on continents and islands) and the negligible impact these tests would have on this highly invasive species (Lowe, et al., 2004). We conclude that it is prudent to continue to assess toxicity risk on a species by species basis, where rodent eradication using brodifacoum or other rodenticides is planned. Trials to determine whether captive snails would eat baits and whether exposure to baits results in measurable mortality are a relatively straightforward and low-cost means to test theoretical assessments of non-target risk.

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REFERENCES

Atkinson, I.A.E. (1985). 'The Spread of Commensal Species of *Rattus* to Oceanic Islands and their Effects on Island Avifaunas'. In: P.J. Moors (ed.) *Conservation of Island Birds*, pp. 35–81. Cambridge: International Council for Bird Preservation.

Bellingham, P.J., Towns, D.R., Cameron, E.K., Davis, J.J., Wardle, D.A., Wilmshurst, J.M. and Mulder, C.P.H. (2010). 'New Zealand island restoration: Seabirds, predators, and the importance of history'. *New Zealand Journal of Ecology* 34: 115–136.

Booth, L.H., Eason, C.T. and Spurr, E.B. (2001). 'Literature review of the acute toxicity and-persistence of brodifacoum to invertebrates'. *Science for Conservation* 177: 1–9.

Booth, L. H., Fisher, P., Heppelthwaite, V. and Eason, C. T. (2003). 'Toxicity and residues of brodifacoum in snails and earthworms'. *Department of Conservation Science Internal Series* 143: 5–14.

Brooke, M. de L., Cuthbert, R.J., Mateo, R. and Taggart, M.A. (2011). 'An experimental test of the toxicity of cereal pellets containing brodifacoum to the snails of Henderson Island, South Pacific'. *Wildlife Research* 38: 34–38.

Cameron, R.A.D., Kostas, A.T., Parent, C.E., Guilhaumon, F., Alonso, M.R., Ibánez, M., Frias Martins, A.M., Ladle, R.L. and Whittaker, R.J. (2013). 'Snails on oceanic islands: Testing the general dynamic model of oceanic island biogeography using linear mixed effect models'. *Journal of Biogeography* 40(1): 117–130.

Clark, D.A. (1981). 'Foraging patterns of black rats across a desert-montane forest gradient in the Galapagos Islands'. *Biotropica* 13: 182–194.

DIISE (2015). *The Database of Island Invasive Species Eradications, developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand*. <http://diise.islandconservation.org>. Accessed 11 December 2015.

DPNG (2014). 'Plan de Manejo de las Áreas Protegidas de Galapagos para el Buen Vivir'. Puerto Ayora, Galapagos: Dirección Parque Nacional Galapagos, Ministerio de Ambiente.

Dunlevy, P., Ebbert, S.E., Russell, J.C. and Towns, D.R. (2011). 'Eradication of Invasive Predators on Seabird Islands'. In: Mulder, C.P.H. Anderson, W.B., Towns, D.R. and Bellingham, P.J. (eds.) *Seabird Islands Ecology, Invasion and Restoration*, pp. 283–316. New York: Oxford University Press.

Eason, C.T., and Spurr, E.B. (1995). 'A review of the toxicity and impacts of brodifacoum on non-target wildlife in New Zealand'. *New Zealand Journal of Zoology* 22(4): 371–379.

Eason, C.T., Milne, L., Potts, M., Morris, S.G., Wright, R.G. and Sutherland, O.R.W. (1999). 'Secondary and tertiary poisoning risks associated with brodifacoum'. *New Zealand Journal of Ecology* 23(2): 219–224.

Eason, C.T., Murphy, E.C., Wright, G.R.G. and Spurr, E.B. (2002). 'Assessment of risks of brodifacoum to non-target birds and mammals in New Zealand'. *Ecotoxicology* 11: 35–48.

Fisher, P., O'Connor, C., Wright, G. and Eason, C. (2003). 'Persistence of four anticoagulant rodenticides in the liver of laboratory rats'. *Department of Conservation Science Internal Series* 139: 1–19.

Fukami, T., Wardle, D.A., Bellingham, P.J., Mulder, C.P.H., Towns, D.R., Yeates, G.W., Bonner, K.I., Durrett, M.S., Grant-Hoffman, M.N. and Williamson, W.M. (2006). 'Above- and below-ground impacts of introduced predators in seabird-dominated island ecosystems'. *Ecology Letters* 9: 1299–1307.

Gerlach, J. and Florens, V. (2000a). 'Considering molluscs in rodent eradication projects'. *Tentacle – The Newsletter of the IUCN/SSC Mollusc Specialist Group* 9: 7–8.

Gerlach, J. and Florens, V. (2000b). 'Toxicity of 'specific' rodenticides and the risk to non-target taxa'. *Phelsuma* 8: 75–76.

Gerlach, J. (2005). 'The impact of rodent eradication on the larger invertebrates of Fregate Island, Seychelles'. *Phelsuma* 13: 44–54.

Howald, G., Donlan, C.J., Galván, J.P., Russell, J., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Island Conservation (2013). 'Floreana ecological restoration: Rodent and cat eradication feasibility analysis version 6.0'. Santa Cruz, CA: Island Conservation.

IUCN (2015). *The IUCN Red List of Threatened Species. Version 2014.2*. Gland, Switzerland and Cambridge, UK: IUCN. <http://www.iucnredlist.org>. Accessed 24 July 2015.

Johnston, J.J., Pitt, W.C., Sugihara, R.T., Eisemann, J.D., Primus, T.M., Holmes, M.J., Crocker, J. and Hart, A. (2005). 'Probabilistic risk assessment for snails, slugs and endangered honeycreepers in diphacinone rodenticide baited areas on Hawaii, USA'. *Environmental Toxicology and Chemistry* 24: 1557–1567.

Jones, H.P., Tershy, B.R., Zavaleta, E.S., Croll, D.A., Keitt, B.S., Finklestein, M.E. and Howald, G.R. (2008). 'Severity of the effects of invasive rats on seabirds: A global review'. *Conservation Biology* 22: 16–26.

Kurle, C.M., Croll, D.A. and Tershy, B.R. (2008). 'Introduced rats indirectly change marine rocky intertidal communities from algae- to invertebrate-dominated'. *Proceedings of the National Academy of Sciences* 105: 3800–3804.

Lorvelec, O. and Pascal, M. (2005). 'French attempts to eradicate non-indigenous mammals and their consequences for native biota'. *Biological Invasions* 7: 135–140.

Lowe, S., Browne, M., Boudjelas, S. and De Poorter, M. (2004). '100 of the world's worst invasive alien species: A selection from the Global Invasive Species Database'. Auckland: The Invasive Species Specialist Group (ISSG) a specialist group of the Species Survival Commission (SSC) of the World Conservation Union (IUCN).

MacNicoll, A.D. (1993). 'Anticoagulant rodenticides - tolerance and resistance'. *Phytoparasitica* 21(3): 185–188.

McMullen, C. K. (1999). *Flowering Plants of the Galápagos*. Ithaca: Cornell University Press.

Nogales, M., Vidal, E., Medina, F.M., Bonnaud, E., Tershy, B.R., Campbell, K.J. and Zavaleta, E.S. (2013). 'Feral cats and biodiversity conservation: The urgent prioritization of island management'. *Bioscience* 63: 804–810.

Pain, D.J., Brooke, M. de L., Finnie, J.K. and Jackson, A. (2000). 'Effects of brodifacoum on the land crab, *Gecarcinus lagostoma* on Ascension Island: acute effects, persistence and distribution in tissues'. *The Journal of Wildlife Management* 64: 380–387.

Parent, C.E., Caccone, A. and Petren, K. (2008). 'Colonization and diversification of Galapagos terrestrial fauna: a phylogenetic and biogeographical synthesis'. *Philosophical Transactions of the Royal Society of London B: Biological Sciences* 363: 3347–3361.

Parent, C.E. and Crespi, B.J. (2006). 'Sequential colonization and rapid diversification of Galapagos endemic land snail genus *Bulimulus* (Gastropoda, Stylommatophora)'. *Evolution* 60: 2311–2328.

Parkes, J., Fisher, P. and Forrester, G. (2011). 'Diagnosing the cause of failure to eradicate introduced rodents on islands: Brodifacoum versus diphacinone and method of bait delivery'. *Conservation Evidence* 8: 100–106.

R Core Team (2014). *R: A language and environment for statistical computing*. Vienna: R Foundation for Statistical Computing.

Rattner, B., Lazarus, R.S., Elliott, J.E., Shore, R.F. and Van Den Brink, N.W. (2014). 'Adverse outcome pathway and risks of anticoagulant rodenticides to predatory wildlife'. *Environmental Science & Technology* 48: 8433–8445.

Sposimo, P., Baccetti, N., Pellicciони, E.R., Guberti, V., Giannini, F. and Capizzi, D. (2011). 'Piano per l'eradicazione del ratto nero *Rattus rattus* nell'Isola di Montecristo, Archipelago Toscano'. <http://www.montecristo2010.it/stealthV3_pubblica/08203500TA4264078046.pdf>. Accessed 7 October 2014.

Spurr, E.B. and Drew, K.W. (1999). 'Invertebrates feeding on baits used for vertebrate pest control in New Zealand'. *New Zealand Journal of Ecology* 23: 167–173.

St Clair, J.J.H. (2011). 'The impacts of invasive rodents on island invertebrates'. *Biological Conservation* 144: 68–81.

Steadman, D.W. (2006). *Extinction and Biogeography of Tropical Pacific Birds*. Chicago: University of Chicago Press.

Towns, D.R. and Broome, K.G. (2003). 'From small Maria to massive Campbell: Forty years of rat eradications from New Zealand islands'. *New Zealand Journal of Zoology* 30: 377–398.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?' *Biological Invasions* 8: 863–891.

Varnham, K. (2010). *Invasive Rats on Tropical Islands: Their History, Ecology, Impacts and Eradication*. Report No. 41. UK: Royal Society for the Protection of Birds Research.

Criteria to help evaluate and guide attempts to eradicate terrestrial arthropod pests

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Abstract Attempts to eradicate invasive terrestrial arthropods are often regarded as gambles. They offer the possibility of long term freedom from a pest but are usually confronted with substantial uncertainty and come with a range of technical, economic, environmental, social and political risks. Few guidelines are available for evaluating eradication attempts against terrestrial arthropods. Here, we build on scientific literature, including six criteria previously developed to evaluate the feasibility of vertebrate eradications, and our own experiences to define nine criteria that are intended to both assist experts to evaluate proposed arthropod eradication attempts and guide attempts that are underway. The criteria are straight forward and easily interpreted, though evaluating them for a particular programme relies on expert group assessment that will often benefit from rigorous supporting statistical and/or modelling analyses.

Keywords: area wide management, decision support, eradication attempt, eradication campaign, eradication feasibility, invasive species, pest control

INTRODUCTION

Invasive species eradications make substantial contributions to conservation (Keitt, et al., 2015; Hoffmann, et al., 2016), agriculture (Vreysen, et al., 2007b; Suckling, et al., 2014a) and human health (Kay & Russell, 2013; Monteiro, et al., 2014). They offer perpetual benefits over long-term pest damage and associated control costs, but are usually expensive, can be disruptive both socially and ecologically, require whole-hearted long-term commitment from those involved, and success is far from guaranteed (Myers, et al., 2000; Myers, 2003; Tobin, et al., 2014; Liebhold, et al., 2016). Thus, eradication attempts invoke a range of technical, economic, environmental, social and political risks. Weighing up information about the potential benefits, costs, risks and probabilities of success of eradication attempts can be fraught with uncertainty (Brown, et al., 2019; Cannon, et al., 1999) and demands that diverse technical issues and societal perspectives be considered (Simberloff, et al., 2013).

Bomford & O'Brien (1995) defined six criteria to help evaluate the feasibility of eradicating vertebrate pests. They drew from lessons learnt in eradicating feral goats from islands, coypus in England, and infectious human diseases in various countries. These criteria have become widely adopted (Brown & Sherley, 2002; Burbidge & Morris, 2002; Clout & Veitch, 2002; Simberloff, 2003a), and have been used in New Zealand by the Department of Conservation (DOC) and Ministry for Primary Industries to help assess the feasibility of eradicating various vertebrate and arthropod pests (Cromarty, et al., 2002; Ashcroft, et al., 2010). We were members of a Technical Advisory Group convened by DOC to assist it with its eventually successful attempt to eradicate a non-native butterfly, *Pieris brassicae* (Lepidoptera: Pieridae), from New Zealand (Phillips, et al., 2016; Brown, et al., 2019). This Palearctic butterfly was regarded as a major risk to New Zealand's 79 native (mostly endemic) Brassicaceae species, many of which were already at risk (Hasenbank, et al., 2011; de Lange, et al., 2013), and also to cultivated exotic brassicas. We began using Bomford & O'Brien's (1995) criteria while evaluating the *P. brassicae* eradication programme and found them useful, though not entirely appropriate for arthropods. Moreover, we were cognisant of valuable insights about arthropod eradications that had been described in the literature since Bomford & O'Brien (1995). Thus, we refined and added to their criteria near the outset of the *P. brassicae* eradication attempt, then used these modified criteria throughout the programme to help both evaluate if the campaign should continue and identify the improvements required to maximise its chance of succeeding.

We present our refined set of criteria here with the aim of assisting others with expertise in pest eradication and arthropod ecology to evaluate and guide further arthropod eradication attempts. Other authors have summarised the elements and processes needed to mount an effective eradication campaign (Cromarty, et al., 2002; Hosking, 2002a; Hosking, 2002b; Vreysen, et al., 2007b; Pacific Invasives Initiative, 2013). However, to our knowledge, criteria developed specifically to help evaluate attempts to eradicate terrestrial arthropods have not previously been documented. Information about previous eradication programmes has recently been compiled in an on-line database (Kean, et al., 2018), yet much valuable information about eradication attempts remains either as grey literature or unrecorded, which impedes improvements in eradication methods (Myers, 2003; Hoffmann, et al., 2011). Thus, we endeavour to document some of our own lessons here.

We do not: review any eradication programmes (Vreysen, et al., 2007b; Hoffmann, et al., 2011; Suckling, et al., 2014a); discuss the growing ecological knowledge and developing technologies that are steadily increasing the potential for eradication attempts to succeed (Vreysen, et al., 2007b; Liebhold, et al., 2016; Alphey & Bonsall, 2017; Scott, et al., 2017); or discuss the enormous benefit of protecting countries or regions from invasions by new pests and diseases (Leung, et al., 2002; Hoffmann, et al., 2011; Lovett, et al., 2016). Nor do we attempt to provide a list of criteria that must be irrefutably met before choosing to initiate – or persevere with – an eradication attempt. Rather, we aim to list some readily interpretable, easily used criteria to assist constructive discussion, decision making and planning within the broader context of what is at stake if the pest is allowed to persist and spread, the pest's priority relative to other problems, and the availability of the resources, expertise and personnel required to mount an effective campaign. Evaluating the criteria will often benefit from expert group assessment and rigorous supporting statistical and/or modelling analyses. Eradication attempts usually involve many uncertainties,

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and the criteria should help to identify those that are most important to resolve as programmes proceed. Certainly, we will have missed our goal if the criteria impede prompt, effective action (Simberloff, 2003a; Simberloff, 2003b; Martin, et al., 2012; Sims & Finnoff, 2013).

METHODS

We adapted the six criteria of Bomford & O'Brien (1995) to make them clearer and more directly applicable to arthropod eradications. We reviewed science literature about factors that influence the success of eradication attempts, and used the insights gained from the science publications to further refine the six criteria. Based both on the literature and our own experiences of eradication attempts, we also developed three additional criteria.

RESULTS

The nine criteria are listed below, each with clarifying comments and, where available, supporting evidence from the literature. The list begins with criteria that deal mainly with details of the species being considered, the tools available to suppress it, and the physical environment in which it occurs, and ends with those that relate more to the societal and organisational context of the eradication attempt. Criteria 1–3, 5, 7 and 8 are based on the six criteria of Bomford & O'Brien (1995), though we modified them to make them clearer and more directly applicable to terrestrial arthropod eradications. Criterion 6 is from Pacific Invasives Initiative (2013), and we added criteria 4 and 9 based both on recent research (Pluess, et al., 2012a; Pluess, et al., 2012b; Tobin, et al., 2014; Buddenhagen & Tye, 2015) and our own experiences (Cromarty, et al., 2002; Brown & Brown, 2015; Keitt, et al., 2015; Phillips, et al., 2016; Brown, et al., 2019).

1. The pest population can be forced to decline from one generation to the next, irrespective of its density.

This is a re-wording of criterion 1 of Bomford & O'Brien (1995): *Rate of removal exceeds rate of increase at all population densities.* Myers, et al. (2000), in a review that covered eradications of species from several phyla including arthropods, also emphasised that the pest must be susceptible to control. Our changes recognise that terrestrial arthropods typically have several life stages per generation, which will likely have differing susceptibilities to control. Thus, it may be acceptable for particular life stages to numerically increase (e.g., egg-stage offspring may outnumber their adult parents) provided the overall effect of control measures is to cause inter-generational declines. Moreover, the availability of tools to suppress the pest population both at very high and at very low population densities should be considered. The eradication attempt must be capable of driving the population to extinction once it has been suppressed to very low densities and becomes more difficult to detect. Some commonly used pest control methods, such as host plant removal, biological control, and insecticide applications, do not require direct detection of pest individuals, and have potential to be effective across a range of population densities. Some species suffer Allee effects at low population densities, which can drive them to extinction once they have been suppressed to below critical density thresholds (Blackwood, et al., 2012; Liebhold, et al., 2016).

2. Every pest individual must be at risk of control at some stage of its development.

This is a re-wording of criterion 3 of Bomford & O'Brien (1995): *All reproductive animals must be at risk.* Our changes recognise that terrestrial arthropods typically have several life stages per generation, and these will likely have differing susceptibilities to control. A combination of control techniques targetting different life stages could increase the likelihood the pest will be successfully eradicated (Blackwood, et al., 2012; Suckling, et al., 2014b; Hoffmann, et al., 2016). Control methods might include augmenting natural enemies that already occur in or near the treated area to increase predation or parasitism of the pest (Montoya, et al., 2007; Hogg, et al., 2013; Richards, et al., 2016).

3. Pest individuals can be detected at low population densities.

This is a minor re-wording of criterion 4 of Bomford & O'Brien (1995): *Animals can be detected at low densities.* It is supported by many studies (Myers, et al., 2000; Simberloff, 2003a; Tobin, et al., 2014). The latter study analysed factors that influenced the outcomes of 672 arthropod eradication programs and found that high detectability contributed to success rates. Population declines must be measurable and, assuming the pest population is eventually suppressed, management tools should be adequate to confirm it has been eradicated. Are effective lures, attractants or traps available, or can they be developed in a timely fashion? When such tools are unavailable and eradication attempts depend on visual searches, they are more successful when targeting easily-observed foliage-feeding species, rather than species that occupy more cryptic niches such as roots, fruit or stems (Tobin et al., 2014). Moreover, programmes without sensitive detection tools that capitalise on citizen reports of sightings have higher probabilities of success than those that do not (Tobin et al., 2014). Thus, detection will be more likely when: effective attractants are available; the pest and its feeding damage are conspicuous and easily recognised; the pest's host plants are low growing, easily searched and of interest to gardeners, commercial growers and/or citizen ecologists; and citizen surveillance is supported by effective outreach programmes. Some control methods may impede detection. For example, using pheromones to disrupt mating will reduce their efficacy as lures for detection (Suckling, et al., 2014b).

4. Success is favoured by small spatial extent of the population.

Bomford & O'Brien (1995) acknowledged pest population spatial extent as important under 'Other factors'. The meta-analysis of Tobin, et al. (2014) considered 672 arthropod eradication programs that involved pest infestations ranging in area from about 0.1 km^2 to about 100,000 km^2. Overall, there was a base rate of 59% success, and the spatial extent of the targeted population was the most important factor explaining variation around this rate (Tobin, et al., 2014). Population spatial extent was also recognised as a critically important factor in the outcomes of 136 eradication programs against invertebrates, plants and plant pathogens (Pluess, et al., 2012a). When infested areas are small, eradication attempts are less expensive and more likely to be successful (Myers, et al., 2000; Simberloff, 2003a; Brockerhoff, et al., 2010; Pluess, et al., 2012a; Tobin, et al., 2014). This is why "wait and see" responses to detections of new pests are seldom justifiable even when uncertainty is high (Sims & Finnoff, 2013).

5. Immigration and emigration can be prevented.

Here, we added 'and emigration' to criterion 2 of Bomford & O'Brien (1995) (*Immigration prevented*) because an attempt to eradicate a localised arthropod population will fail if individuals disperse from the eradication zone and establish new undetected populations nearby. Myers, et al. (2000) and Hoffmann, et al. (2016) emphasised that immigration (reinvasion) must be prevented, and Bomford & O'Brien (1995) acknowledged pest dispersal rates as important under 'Other factors'. Attempts to eradicate isolated localised populations (e.g. on islands or other geographically isolated areas) might benefit from low likelihoods of natural pest dispersal in or out of the eradication zone (Myers, et al., 2000), though an analysis of 173 eradication programmes found no evidence that eradication attempts were more successful on islands (Pluess, et al., 2012b). The likelihood the pest will be transported in or out of the infested area in association with humans must also be considered, as should the extent to which this risk can be mitigated (e.g. by implementing regulatory controls on host plant movements). Pluess, et al. (2012b) found that implementing sanitary measures to restrict pest emigration made an important contribution to eradication success rates. A further consideration is the capability of the programme to identify when immigration or emigration is occurring, and to respond effectively to such processes. Recognising that immigration is occurring may be challenging, though genetically characterising the population within the eradication zone could help to identify new immigrants if they differ genetically from the initially targeted population (Barr, et al., 2014; Hiszczynska-Sawicka & Phillips, 2014; Piertney, et al., 2016). Detecting emigrants will depend on the extent, intensity and efficacy of active and passive surveillance outside the known infested area.

6. Environmental impacts of the programme are acceptable.

Most methods used to manage pests will have non-target impacts (Bomford & O'Brien, 1995; Pacific Invasives Initiative, 2013). These include host plant removal, biological control, synthetic pesticides, biopesticides and traps (e.g. due to by-catch). Eradicating a pest from an ecosystem could release other non-native species from competition, predation or parasitism, thus solving one problem while exacerbating another (Myers, et al., 2000). Decision makers must consider if such impacts will be reversible and/or socially and environmentally acceptable, and if they will be substantially less than those likely to be sustained if the pest became permanently established and more widely distributed. If the infested area being treated is small and the expected term of the programme is short, then environmental impacts might be ephemeral because those non-target species negatively impacted by the eradication programme may be able to recover once the programme ends.

7. Benefit-cost analysis favours eradication over control.

This is a minor re-wording of criterion 5 of Bomford & O'Brien (1995): *Discounted benefit-cost analysis favours eradication over control*. It was also listed by Pacific Invasives Initiative (2013). We omitted the word 'discounted' from Bomford & O'Brien's (1995) criterion because discounting in benefit-cost analysis remains controversial (Gollier & Hammitt, 2014; Hockley, 2014). Myers, et al. (2000) acknowledged that evaluating the benefits and costs of eradication is difficult, and contended that the benefits of eradication are often over estimated and the costs of eradication under estimated. Nevertheless, many successful eradication programmes have been regarded as highly cost-effective (Brockerhoff, et al., 2010; Buddenhagen & Tye, 2015; Scott, et al., 2017). Benefit-cost analyses provide useful frameworks for aggregating information about an eradication attempt to support decision making. However, they must often include educated guesses about parameter values, struggle to quantify the value of biodiversity and ecosystem services, and seldom address uncertainty (Born, et al., 2005; Epanchin-Niell & Hastings, 2010; Simberloff, et al., 2013; Hockley, 2014; Brown, et al., 2019).

8. Suitable social, political, legal and institutional environment.

This is a minor re-wording of criterion 6 of Bomford & O'Brien (1995): *Suitable socio-political environment*. It was emphasised by Buddenhagen & Tye (2015) and similar criteria were listed by Pacific Invasives Initiative (2013) and Simberloff (2003a). Myers, et al. (2000) also stressed that funding must be sufficient and lines of authority clear. Our changes more clearly specify the need for eradication programmes to be supported by every facet of society that has an important role or stake in the programme. Those evaluating eradication attempts must ask questions like: Will property owners allow or support eradication activities on their land? Would those implementing the eradication programme have legal authority to implement control actions on private and public land? Will the programme be supported by stakeholders such as local and regional authorities, farmer organisations and environmental advocacy groups? Will all management levels of the institution(s) attempting the eradication remain fully committed – especially financially – to the programme for the long haul?

9. Programme is effectively managed, and its status is reliably monitored and accurately recorded.

Efficient, meticulous and effective planning and management are critical to eradication success (Cromarty, et al., 2002), as are clear lines of authority (Myers, et al., 2000; Simberloff, 2003a). These programme attributes must be supported by efficient and robust data collection and analysis to enable progress to be monitored, assumptions tested, weaknesses identified, and improvements devised and implemented (Vreysen, et al., 2007a). Brown & Brown (2015) suggested that systematic and persistent effort by individuals with a "completer-finisher" personality type (Belbin, 2010) or an "eradication attitude" can increase the likelihood of success. For arthropod eradications, it is clearly important to involve people with expertise in arthropod ecology and management.

DISCUSSION

We propose that the nine criteria can help to focus discussion and evaluate and guide attempts to eradicate terrestrial arthropods. We repeatedly scored the criteria throughout the *P. brassicae* eradication programme (Phillips, et al., 2016) and, although our individual assessments often differed, they always provided a valuable basis for discussion and planning. Moreover, our individual assessments all became progressively more optimistic as the programme proceeded and uncertainty declined (Phillips, et al., 2015). In fact, optimism grew even as the pest's known geographical distribution increased because we also gained confidence that the pest was detectable and controllable.

We found it useful to classify each criterion as being either 'not met', 'marginally met' or 'substantially met'. These qualitative terms recognised that criteria can be met to varying degrees and using just three classes simplified the assessment process and eased interpretation. Criteria were classified as 'not met' if the eradication attempt was likely to fail unless improvements to that aspect of the programme were urgently made. Criteria were considered 'marginally met' if there was some evidence the criterion could be (or was being) met, but knowledge gaps caused uncertainty to be high and made assessing the criterion difficult. This classification also signalled a need for action because important knowledge gaps had to be addressed to ensure eradication feasibility. Criteria were scored as 'substantially met' when these elements of the eradication attempt appeared (likely to be) effective. Improvements to aspects of the eradication classified as 'not met' or 'marginally met' were regarded as critical and urgent, and improvements to those classified as 'substantially met' were regarded as desirable.

In the context of vertebrate eradications, Bomford & O'Brien (1995) considered criteria 1, 2 and 5 (numbers as used in the main text of the Results in this paper) as essential to success, and criteria 3, 7 and 8 as desirable, though they emphasised that negatives in the latter three criteria "will greatly reduce the feasibility and desirability of eradication". With our criteria for terrestrial arthropod eradications, we suggest that all of the criteria except numbers 4 (small spatial extent) and 7 (benefit-cost analysis) will need to be substantially met before eradication is eventually achieved. However, it may be reasonable to initiate an eradication attempt before many of the critical criteria are substantially met. This is because, with thoughtful management, new knowledge and tools will often be developed during the course of a programme (Vreysen, et al., 2007b; Scott, et al., 2017) that will rectify some or all of its deficiencies and/or enable the criteria to be scored with more confidence. Indeed, the criteria aim to highlight those aspects of programmes that most need improvement. In cases where few critical criteria are substantially met, it will be particularly important to specify conditions under which the attempt will cease (e.g. when a key programme deficiency is not rectified by a specified date) in order to minimise expenditure on programmes that are doomed to failure.

The capability to robustly evaluate programme progress and confidently reclassify criteria is especially dependent on criterion 9 (excellent management). It is also highly desirable that a (proposed) programme substantially meets criteria 4 (small spatial extent) and 7 (benefit-cost analysis). Yet, with criterion 4, there are examples of arthropod populations with very large spatial extents that have been successfully eradicated (Vreysen, et al., 2007b; Monteiro, et al., 2014; Scott, et al., 2017), thus scores of 'not met' or 'marginally met' may be acceptable in cases where the other criteria for achieving eradication can be substantially met and resources are available to work effectively across large geographical areas. For criterion 7, perceptions of the potential economic and/or environmental benefits of an eradication attempt will strongly influence the level of risk that is deemed acceptable when deciding whether to initiate or persist with the attempt. However, the previously mentioned limitations of benefit-cost analyses (Born, et al., 2005; Epanchin-Niell & Hastings, 2010; Simberloff, et al., 2013; Hockley, 2014; Brown, et al., 2019) combined with overwhelming evidence of negative impacts of many invaders suggests that the precautionary principle (Simberloff, et al., 2013) should be applied particularly to criterion 7, and scores of 'marginally met' may be adequate to justify action.

We intend the criteria to be used by people with expertise in pest eradication and arthropod ecology and management. During the *P. brassicae* eradication attempt, we found it productive to discuss programme performance against each criterion as a group because our individual perspectives often initially differed. Our evaluations of one or more criteria were frequently supported by statistical and/or modelling analyses of data being collected by the programme. Eventually we would reach a consensus that enabled us to provide better advice to the programme than any one of us could have alone. Thus, we advocate using the criteria in fora similar to the 'Technical Advisory Groups' that are often applied in New Zealand to support management decision making. We believe that using the criteria to help evaluate the feasibility of an eradication attempt and its progress towards success will help to improve decision making and increase programme success rates. However, the quality of decision making will of course also depend on the values and motivations of decision makers, the experience and problem-solving abilities of the expert group, the quality of data analysis, and the preparedness of all involved to fill knowledge gaps and take timely action.

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REFERENCES

Alphey, N. and Bonsall, M.B. (2017). 'Genetics-based methods for agricultural insect pest management'. *Agricultural and Forest Entomology* 20: 131–140.

Ashcroft, T., Richardson, E., Voice, D. and Dustow, K. (2010). *Strategies and Feasibility to Undertake Eradication of* Pieris brassicae *(GWCB) in New Zealand (No. version 3 (final)).* Wellington, New Zealand: Ministry of Agriculture and Forestry.

Barr, N., Ruiz-Arce, R. and Armstrong, K. (2014). 'Using Molecules to Identify the Source of Fruit Fly Invasions'. In: T. Shelly, N. Epsky, E.B. Jang, J. Reyes-Flores and R. Vargas (eds.) *Trapping and the Detection, Control, and Regulation of Tephritid Fruit Flies*, pp. 321–378. Dordrecht, The Netherlands: Springer.

Belbin, R. (2010). '*Management Teams: Why they Succeed or Fail*'. Oxford, UK: Elsevier.

Blackwood, J.C., Berec, L., Yamanaka, T., Epanchin-Niell, R.S., Hastings, A. and Liebhold, A.M. (2012). 'Bioeconomic synergy between tactics for insect eradication in the presence of Allee effects'. *Proceedings of the Royal Society of London B: Biological Sciences* 279: 2807–2815.

Bomford, M. and O'Brien, P. (1995). 'Eradication or control for vertebrate pests?'. *Wildlife Society Bulletin* 23: 249–255.

Born, W., Rauschmayer, F. and Brauer, I. (2005). 'Economic evaluation of biological invasions—a survey'. *Ecological Economics* 55: 321–336.

Brockerhoff, E.G., Liebhold, A.M., Richardson, B. and Suckling, D.M. (2010). 'Eradication of invasive forest insects: concepts, methods, costs and benefits'. *New Zealand Journal of Forestry Science* 40: 117–135.

Brown, K. and Brown, D. (2015). 'Control to eradication of *Tradescantia fluminensis* on Stephens Island (Takapourewa): the importance of systematic and persistent effort'. *DOC Research & Development Series* 15. Wellington, NZ: Department of Conservation.

Brown, K., Phillips, CB, Broome, K., Green, C., Toft, R, and Walker, G. (2019). 'Feasibility of eradicating the large white butterfly (*Pieris brassicae*) from New Zealand: Data Gathering to Inform Decisions about the Feasibility of Eradication.' In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 364–369. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Brown, K.P. and Sherley, G.H. (2002). 'The eradication of possums from Kapiti Island, New Zealand'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 46–52. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Buddenhagen, C.E. and Tye, A. (2015). 'Lessons from successful plant eradications in Galapagos: commitment is crucial'. *Biological Invasions* 17: 2893–2912.

Burbidge, A.A. and Morris, K.D. (2002). 'Introduced mammal eradications for nature conservation on Western Australian Islands: A Review'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 64–70. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Cannon, R., Pemberton, A. and Bartlett, P. (1999). 'Appropriate measures for the eradication of unlisted pests'. *EPPO Bulletin* 29: 29–36.

Clout, M.N. and Veitch, C.R. (2002). 'Turning the tide of biological invasion: the potential for eradicating invasive species.' In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 1–3. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Cromarty, P.L., Broome, K.G., Cox, A., Empson, R.A., Hutchinson, W.M. and McFadden, I. (2002). 'Eradication planning for invasive alien animal species on islands – the approach developed by the New Zealand Department of Conservation'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 85–91. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

de Lange, P.J., Rolfe, J.R., Champion, P.D., Courtney, S., Heenan, P.B., Barkla, J.W., Cameron, E.K., Norton, D.A. and Hitchmough, R. (2013). *Conservation Status of New Zealand Indigenous Vascular Plants, 2012*. Wellington, NZ: Department of Conservation.

Epanchin-Niell, R.S. and Hastings, A. (2010). 'Controlling established invaders: integrating economics and spread dynamics to determine optimal management'. *Ecology Letters* 13: 528–541.

Gollier, C. and Hammitt, J.K. (2014). 'The long-run discount rate controversy'. *Annual Review of Resource Economics* 6: 273–295.

Hasenbank, M., Brandon, A. and Hartley, S. (2011). 'White butterfly (*Pieris rapae*) and the white rust *Albugo candida* on Cook's scurvy grass (*Lepidium oleraceum*)'. *New Zealand Journal of Ecology* 35: 69–75.

Hiszczynska-Sawicka, E. and Phillips, C.B. (2014). 'Mitochondrial cytochrome c oxidase subunit 1 sequence variation in New Zealand and overseas specimens of *Pieris brassicae* (Lepidoptera: Pieridae)'. *New Zealand Plant Protection* 67: 8–12.

Hockley, N. (2014). 'Cost-benefit analysis: a decision-support tool or a venue for contesting ecosystem knowledge?' *Environment and Planning C: Government & Policy* 32: 283–300.

Hoffmann, B., Davis, P., Gott, K., Jennings, C., Joe, S., Krushelnycky, P., Miller, R., Webb, G. and Widmer, M. (2011). 'Improving ant eradications: details of more successes, a global synthesis and recommendations'. *Aliens* 31: 16–23.

Hoffmann, B.D., Luque, G.M., Bellard, C., Holmes, N.D. and Donlan, C.J. (2016). 'Improving invasive ant eradication as a conservation tool: a review'. *Biological Conservation* 198: 37–49.

Hogg, B.N., Wang, X.-G., Levy, K., Mills, N.J and Daane, K.M. (2013). 'Complementary effects of resident natural enemies on the suppression of the introduced moth *Epiphyas postvittana*'. *Biological Control* 64: 125–131.

Hosking, G. (2002a). *An Emergency Response Guide for Forestry Incursions*. Rotorua, NZ: Forest Research.

Hosking, G. (2002b). 'Incursion response – back to basics'. *New Zealand Journal of Forestry* 47: 24–27.

Kay, B. and Russell, R. (2013). '*Mosquito Eradication: The Story of Killing Campto*'. Collingwood, Victoria, Australia: CSIRO Publishing.

Kean, J.M., Suckling D.M., Sullivan N.J., Tobin P.C., Stringer L.D., Smith G.R., Kimber B, Lee D.C., Flores Vargas R., Fletcher J., Macbeth F., McCullough D.G., Herms D.A. et al. (2018). 'Global Eradication and Response Database'. <http://b3.net.nz/gerda>. Accessed 20 April 2018.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

Leung, B., Lodge, D.M., Finnoff, D., Shogren, J.F., Lewis, M.A. and Lamberti, G. (2002). 'An ounce of prevention or a pound of cure: bioeconomic risk analysis of invasive species'. *Proceedings of the Royal Society of London (B) Biological Sciences* 269: 2407–2413.

Liebhold, A.M., Berec, L., Brockerhoff, E.G., Epanchin-Niell, R.S., Hastings, A., Herms, D.A., Kean, J.M., McCullough, D.G., Suckling, D.M. and Tobin, P.C. (2016). 'Eradication of invading insect populations: from concepts to applications'. *Annual Review of Entomology* 61: 335–352.

Lovett, G.M., Weiss, M., Liebhold, A.M., Holmes, T.P., Leung, B., Lambert, K.F., Orwig, D.A., Campbell, F.T., Rosenthal, J. and McCullough, D.G. (2016). 'Nonnative forest insects and pathogens in the United States: impacts and policy options'. *Ecological Applications* 26: 1437–1455.

Martin, T.G., Nally, S., Burbidge, A.A., Arnall, S., Garnett, S.T., Hayward, M.W., Lumsden, L.F., Menkhorst, P., McDonald-Madden, E. and Possingham, H.P. (2012). 'Acting fast helps avoid extinction'. *Conservation Letters* 5: 274–280.

Monteiro, F.A., Shama, R., Martins, A.J., Gloria-Soria, A., Brown, J.E. and Powell, J.R. (2014). 'Genetic diversity of Brazilian *Aedes aegypti*: patterns following an eradication program'. *PLOS Neglected Tropical Diseases* 8: e3167.

Montoya, P., Cancino, J., Zenil, M., Santiago, G. and Gutierrez, J. (2007). 'The Augmentative Biological Control Component in the Mexican National Campaign against *Anastrepha* spp. Fruit Flies'. In: M.J. Vreysen, A.S. Robinson and J. Hendrichs, (eds.) *Area-wide Control of Insect Pests: From Research to Field Implementation*, pp. 661–670. Dordrecht, The Netherlands: Springer Science & Business Media.

Myers, J.H. (2003). 'Eradication: Is it Ecologically, Financially, Environmentally, and Realistically Possible'. In: D.J. Rapport, B.L. Lasley, D.E. Rolston, N.O. Nielsen, C.O. Qualset and A.B. Damania (eds.) *Managing for Healthy Ecosystems*, pp. 533–539. Boca Raton: Lewis Publishers.

Myers, J.H., Simberloff, D., Kuris, A.M. and Carey, J.R. (2000). 'Eradication revisited: dealing with exotic species'. *Trends in Ecology and Evolution* 15: 316–320.

Pacific Invasives Initiative (2013). Resource kit for rodent and cat eradication. <http://www.pacificinvasivesinitiative.org/rk/project/2_Feasibilty_Study.html>. Accessed 9 May 2013.

Phillips, C.B., Brown, K., Green, C., Broome, K., Toft, R., Shepherd, M., Bayley, S. and Rees, J. (2015). Pieris brassicae *(Great White Butterfly) Eradication Annual Report 2014/15* (Department of Conservation Annual Report No. R77017), *Pieris brassicae* Eradication. Nelson, New Zealand: Department of Conservation.

Phillips, C.B., Brown, K., Green, C., Broome, K., Toft, R., Shepherd, M., Bayley, S. and Rees, J. (2016). Pieris brassicae *(Great White Butterfly) Eradication Annual Report 2015/16* (Department of Conservation Annual Report), *Pieris brassicae* Eradication. Nelson, New Zealand: Department of Conservation.

Piertney, S.B., Black, A., Watt, L., Christie, D., Poncet, S. and Collins, M.A. (2016). 'Resolving patterns of population genetic and phylogeographic structure to inform control and eradication initiatives for brown rats *Rattus norvegicus* on South Georgia'. *Journal of Applied Ecology* 53: 332–339.

Pluess, T., Cannon, R., Jarošík, V., Pergl, J., Pyšek, P. and Bacher, S. (2012a). 'When are eradication campaigns successful? A test of common assumptions'. *Biological Invasions* 14: 1365–1378.

Pluess, T., Jarošík, V., Pyšek, P., Cannon, R., Pergl, J., Breukers, A. and Bacher, S. (2012b). 'Which factors affect the success or failure of eradication campaigns against alien species?'. *PLOS ONE* 7: e48157.

Richards, N., Hardwick, S., Toft, R. and Phillips, C. (2016). 'Enhancing parasitism of *Pieris brassicae* pupae by *Pteromalus puparum*'. *New Zealand Plant Protection* 69: 126–132.

Scott, M.J., Concha, C., Welch, J.B., Phillips, P.L. and Skoda, S.R. (2017). 'Review of research advances in the screwworm eradication program over the past 25 years'. *Entomologia Experimentalis et Applicata* 164: 226–236.

Simberloff, D. (2003a). 'Eradication—preventing invasions at the outset'. *Weed Science* 51: 247–253.

Simberloff, D. (2003b). 'How much information on population biology is needed to manage introduced species?'. *Conservation Biology* 17: 83–92.

Simberloff, D., Martin, J.-L., Genovesi, P., Maris, V., Wardle, D.A., Aronson, J., Courchamp, F., Galil, B., García-Berthou, E. and Pascal, M. (2013). 'Impacts of biological invasions: what's what and the way forward'. *Trends in Ecology and Evolution* 28: 58–66.

Sims, C. and Finnoff, D. (2013). 'When is a "wait and see" approach to invasive species justified?'. *Resource and Energy Economics* 35: 235–255.

Suckling, D., Kean, J.M., Stringer, L.D., Cáceres-Barrios, C., Hendrichs, J., Reyes-Flores, J. and Dominiak, B.C. (2014a). 'Eradication of tephritid fruit fly pest populations: outcomes and prospects'. *Pest management science* 72: 456–465.

Suckling, D., Stringer, L.D., Stephens, A.E., Woods, B., Williams, D.G., Baker, G. and El-Sayed, A.M. (2014b). 'From integrated pest management to integrated pest eradication: technologies and future needs'. *Pest Management Science* 70: 179–189.

Tobin, P., Kean, J., Suckling, D., McCullough, D., Herms, D. and Stringer, LD (2014). 'Determinants of successful arthropod eradication programs'. *Biological Invasions* 16: 401–414.

Vreysen, M., Gerardo-Abaya, J. and Cayol, J. (2007a). 'Lessons from Area-wide Integrated Pest Management (AW-IPM) Programmes with an SIT Component: An FAO//IAEA Perspective'. In M. Vreysen, A. Robinson and J. Hendrichs (eds.) *Area-wide Control of Insect Pests: From Research to Field Implementation*, pp. 723–744. Dordrecht, The Netherlands: Springer Science & Business Media.

Vreysen, M., Robinson, A. and Hendrichs, J. (eds.) (2007b). *Area-wide Control of Insect Pests: From Research to Field Implementation*'. Dordrecht, The Netherlands: Springer Science & Business Media.

Eradication of invasive alien crayfish: past experiences and further possibilities

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Abstract The EU regulation 1143/2014 "On the prevention and management of the introduction and spread of invasive alien species" entered into force on 1 January 2015. On 13 July 2016, the EU list of invasive alien species that require action was adopted. The list includes five different crayfish species. Member states will be required to take measures for early detection and rapid eradication of these species. Except for some eradications performed in the United Kingdom and Norway, there has not been much effort put into eradication of invasive crayfish species throughout Europe. The reasons for this are probably complex and differ between member states. Are the main reasons legislative constraints, ability to eradicate or lack of knowledge and experience? Is eradication of alien crayfish possible and desirable, and what is left to save in Europe? Focus could be put into identifying or creating island populations of special concern and preserve them for the future survival of European native crayfish populations. Eradication measures should be considered as an option in this work. What are the experiences from completed eradication efforts in Europe? Two crayfish eradications have been performed in Norway, and both have been successful. The eradications were performed in locations with several ponds and small streams and performed using the synthetic pyrethroid-based pharmaceutical BETAMAX VET®. Both legislative and funding constraints seem less prominent as successful eradications have been confirmed. Time will show if this trend will spread throughout Europe.

Keywords: crayfish management, invasive alien species, IAS, pyrethroids, signal crayfish

INTRODUCTION

The EU regulation 1143/2014 "On the prevention and management of the introduction and spread of invasive alien species" (<http://data.europa.eu/eli/reg/2014/1143/oj>) entered into force on 1 January 2015. On 13 July 2016 the EU list of invasive alien species, IAS that requires action was adopted. The list includes five different crayfish species, spinycheek crayfish (*Orconectes limosus*), virile crayfish (*Orconectes virilis*), signal crayfish (*Pacifastacus leniusculus*), red swamp crayfish (*Procambarus clarkii*), and marbled crayfish (*Procambarus fallax*) (<http://data.europa.eu/eli/reg_impl/2016/1141/oj>). Crayfish are one of the most successful and widely distributed invasive species in the world (Holdich, et al., 2014). Twenty eight different crayfish species have been translocated from their native range, and seven of them have been identified with invasive potential (Gherardi, 2010). At least ten non-native species of crayfish have been introduced to Europe (Souty-Grosset, et al., 2006). The five indigenous European freshwater crayfish species are all threatened by different factors, but the most detrimental is probably the North American signal crayfish *Pacifastacus leniusculus* and the crayfish plague caused by the oomycete parasite *Aphanomyces astaci* (Holdich & Sibley, 2009). Signal crayfish are natural hosts for the crayfish plague (Unestam, 1972), the causal agent of crayfish plague, and a disease lethal to European freshwater crayfish (Alderman, et al., 1990; Souty-Grosset, et al., 2006), causing dramatic population reduction and in many cases extinction (Holdich, et al., 1999). The signal crayfish exhibits a number of biological adaptions which allow it to tolerate extreme environmental conditions (McMahon, 2002). This flexibility may facilitate the further spread of both he crayfish and the crayfish plague.

The EU regulation on invasive alien species includes restrictions on keeping, importing, selling, breeding and growing listed species. Taking action as early as possible and preventing introduction will ensure that unnecessary suffering of animals is avoided and is more cost effective than eradication. On the other hand, member states will be required to take measures for early detection and rapid eradication of listed species. If a new population is detected there is an eradication obligation, whereas for widely spread species management measures must take place. The list mainly contains species already present in the EU, but future updates are expected to introduce more species not yet present in the EU. Member states select the measures appropriate to the local conditions and do not have an obligation to eradicate IAS of Union concern that are already widely spread in their territory.

Throughout Europe there have been several attempts to eradicate different crayfish species. Reviews of possible methods for controlling nuisance populations of alien crayfish are available (Holdich, et al., 1999; Hiley, 2003; Ribbens & Graham, 2004; Peay & Hiley, 2006; Freeman, et al., 2010; Stebbing, et al., 2014). These methods include different legislative, mechanical, biological and physical measures, including the use of biocides and pheromones. Mechanical methods, such as trapping, seining, and electrofishing can control, but not eradicate crayfish populations (Holdich, et al., 1999; Hiley, 2003; Peay & Hiley, 2006). It seems that only chemical based treatments offer any hope for effective eradication of invasive crayfish species (Peay, 2001).

Except for some eradications performed in the United Kingdom (Peay, et al., 2006) and Norway (Sandodden & Bardal, 2010; Sandodden & Johnsen, 2010), there has not been much effort put into eradication of invasive crayfish species throughout Europe using chemicals. The reasons for this are probably complex and differ between countries. Are the main reasons legislative constraints, unwillingness or lack of knowledge and experience? Is eradication of alien crayfish possible and desirable, and what is left to save in Europe?

Chemical methods of eradication include the use of biocides, surfactants and pheromones. Ribbens & Graham (2004) review the use of biocides for control of crayfish populations. Organophosphates and organochlorines are reported to be effective, but these chemicals are known to bioaccumulate through the food chain (Holdich, et al. 1999). Crayfish can bioaccumulate organochlorines and, as crayfish are eaten by many predators, this is obviously important in terms of biomagnification through the food

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chain (Ludke, et al., 1971) In contrast, both natural pyrethrum (Pyblast) and synthetic pyrethroids, which have been shown to be effective at very low doses, break down rapidly and do not bioaccumulate (Holdich, et al., 1999; Hiley, 2003; Peay & Hiley, 2006). Synthetic pyrethrum is based on the chemical structure and biological activity of natural pyrethrum, an extract of plants of the genus *Chrysanthemum* (Holdich, et al., 1999).

Eversole & Seller (1997) concluded, in a comprehensive study based on 35 different chemical groups, that synthetic pyrethroids were the most poisonous to crayfish. Both natural pyrethrum and synthetic pyrethroids have low toxicity to birds, mammals, plants and many invertebrates (Van Wijngaarden, et al., 2005). They are, however, in varying degrees toxic to non-target fauna, including crustaceans, insects, arthropods, fish and amphibians (Mayer & Ellersieck, 1986; Burridge & Haya, 1997). The environmental fate and degradation of pyrethroid insecticides were reviewed by Leahey (1979). He concluded that pyrethroids do not persist in the environment for long periods, do not accumulate in the biosphere and do not biomagnify in the food chain. Ecosystem recovery is fairly rapid, with the toxic effect of pyrethroids lasting from days to months, and all major animal groups recovering within a year (Gydemo, 1995). Holdich, et al. (1999) states that ecosystems can recover fairly rapidly from the toxic effects of pyrethroids. Compared to natural pyrethrum the synthetic forms are more toxic, less degradable by light, more readily available and less expensive (Morolli, et al., 2006). To date, no crayfish-specific biocide has been developed.

O`Reilly (2015) suggested that lower concentrations of the natural pyrethrum may be suitable to eradicate or control signal crayfish in small standing waterbodies. Where the risk of damage to non-target species is not an issue and the water is not being used for another purpose, cheaper alternative biocides such as synthetic pyrethroids could be used.

Two successful signal crayfish eradications have been performed in Norway. On the basis of these results and EU regulation 1143/2014, more focus should be put into identifying or creating island populations of special concern and preserve them for the future survival of European native crayfish populations. Eradication measures should and must be considered as an option in this work. The number of eradication attempts probably will increase in Europe as both the knowledge base and environmental impacts increase.

MATERIALS AND METHODS

Both successful eradications were performed in southern Norway close to the capital Oslo (Fig. 1). The locations consisted of several ponds and small streams and involved the application of the synthetic pyrethroid-based pharmaceutical BETAMAX VET®, which is a cypermethrin-based pharmaceutical originally developed for treatment of salmon louse (*Lepeophtheirus salmonis*) infestation of farmed Atlantic salmon (*Salmo salar*). Cypermethrin is a synthetic pyrethroid and a common agent in many insecticides licensed throughout Europe.

Both eradications involved two separate consecutive treatments separated by two weeks and a partial drainage of some of the ponds. The first eradication was performed in the Dammane watershed in Telemark County, during May 2008. The watercourse consists of a creek with five small ponds, the largest measuring approximately 2,000 m^2 (Table 1). The treatment is described in detail in Sandodden & Johnsen (2010). The second eradication was undertaken using the same pharmaceutical, methods and equipment in the Oslo & Akershus County at Ostøya, an island in Oslo-fjord, during October 2009. The treatment involved six ponds on a golf course. The two largest ponds were close to 2,200 m^2 (Table 1). The treatment is described in Sandodden & Bardal (2010, in Norwegian). All ponds were treated with the help of pumps placed in a boat or on the shore (Fig. 2). The chemical was dispersed both on the water surface, along the pond bottom and on a 10 m onshore belt around each pond. Continuous drip stations were placed at the most upstream location of each creek or seep to ensure treatment of the whole drainage basin. This ensured a continuous, constant dosage of BETAMAX VET® during treatment. In the smallest of seeps, enclosed water bodies and small upstream creeks, watering cans were used to dispense a dilution of the chemical. For more details regarding methods, see Sandodden & Johnsen (2010).

The requirements set by the Norwegian Food and Safety Authority for issuing an eradication confirmation after eradication of signal crayfish are described in Johnsen, et al., (2010) and state: 1. No crayfish caught during trapping five to five and a half years after eradication is performed. 2. Noble crayfish (*Astacus astacus*) placed in cages in the treated area have shown no signs of crayfish plague during the last three years of monitoring. 3. Analyses of water and sediments show no sign of crayfish plague spores five to five and a half years after eradication. The methodology is described in Vrålstad, et al., (2009). Based

Table 1 Area, mean depth, volume and BETAMAX VET® used during treatment of the ponds at Dammane and Ostøya locations.

		Area m^2	Mean depth metres	Volume m^3	BETAMAX litres
Dammane	Dam 1	371	0.82	303	0.14
	Dam 2	697	0.92	639	0.17
	Dam 3	1,146	2.27	2,602	1.41
	Dam 4	3,154	1.92	6,054	2.78
	Dam 5	1,346	1.73	1,996	0.57
Ostøya	Dam 14	2,242	3.00	6,726	3.56
	Dam 18	1,400	1.80	2,520	1.33
	Dam 13	990	1.80	1,782	0.94
	Dam 2	2,200	1.80	3,960	2.09
	Dam 1	1,054	1.80	1,897	1.00
	Dam 8	370	2.00	370	0.20

on the investigations involved in eradication confirmation, the Norwegian Food and Safety Authority can issue a self-declaration of freedom of disease (OIE, 2009).

RESULTS

Dammane

No surviving crayfish was observed or found during the second treatment or during drainage of the ponds. On the basis of the Norwegian Food and Safety Authority self-declaration of freedom for disease procedure, the County Governor carried out trials with caged live noble crayfish in 2010 and 2011. In 2010, a total of 31 male crayfish were placed in three cages in three of the treated ponds. The caged crayfish suffered a high mortality during the trials that lasted for 136 days. Analyses performed at the Norwegian Veterinary Institute showed that the cause of death was not crayfish plague. In 2011, a total of 30 male crayfish were placed in three cages in two of the treated ponds. The caged crayfish suffered a high mortality during the trials that lasted for 129 days. Analyses performed at the Norwegian Veterinary Institute showed that the cause of death was not caused by crayfish plague. In 2011 a trapping trial for crayfish was carried out in two of the treated ponds. No crayfish were caught.

Regarding eradication confirmation, the Norwegian Food and Safety Authority concluded on the basis of these results in December 2011 that they could either issue an eradication confirmation based on the results alone or carry out trials with caged crayfish for another year. The relatively new method using molecular investigations based on water samples in search of crayfish plague spores might be carried out as an addition, but the more realistic approach would be caged crayfish trials. The Norwegian Food and Safety Authority's final advice was to issue an eradication confirmation based on the results given in Dammane. They have not yet issued a formal letter or report declaring eradication confirmation (Jan Egil Aronsen, Norwegian Food and Safety Authority pers. comm., 2017).

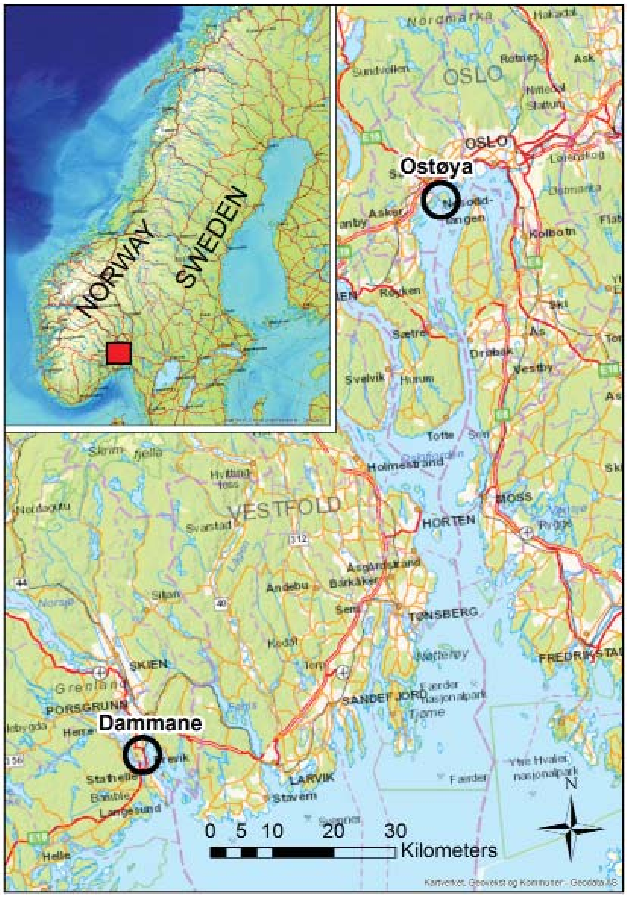

Fig. 1 Dammane and Ostøya locations in southern Norway.

Ostøya

No surviving crayfish was observed or found during the second treatment or during drainage of the ponds. The County Governor carried out trials with caged live noble crayfish in 2013 and 2014. Cages were placed in five of the treated ponds. No signs of disease or crayfish plague were detected. In 2014 a trapping trial for crayfish was carried out. No crayfish were caught. In June 2014 the Norwegian Veterinary Institute collected water samples for analyses of crayfish plague spores in two of the treated ponds (unpublished data). No spores were detected. On the basis of these results the County Governor concluded that the signal crayfish and the crayfish plague is eradicated form the infected ponds. These are unpublished results but summarized in a letter from the County Governor dated 17 March 2017 (ref. 2017/1978-1 M-NA).

DISCUSSION

What is left to save in Europe?

There are still significant native crayfish populations in Europe, which are being decimated through the spread of introduced invasive non-native crayfish (Gherardi, et al., 2011). Action to control invasive non-native crayfish would protect these rare and valuable species. Equally, the impacts from invasive non-native crayfish are wider, ranging from damage to river and flood defence banks (Guan, 1994), through to impact on recreational fisheries. So, there is a case for action based on both ecological and socio-economic factors.

Is eradication of alien crayfish possible and desirable?

As this paper shows, there are possibilities for crayfish eradication. We have the scientific evidence base regarding the species, their risks and impacts; we have the processes to make a robust case, tools, techniques and expertise to take action and now the powers under EU IAS regulations to make that a reality. It is possible to make robust cases to government and only by doing this can we tackle the final funding barrier. Reporting successful eradications should both inspire and motivate future eradication projects. To justify the use of chemicals, it is important to conduct and report the environmental impacts following the eradication attempts and evaluate these in comparison to not taking action.

Fig. 2 Boat mounted pump used to apply BETAMAX VET® during the Dammane and Ostøya treatments.

Lack of eradication projects

The answer to why there have been no eradication projects until now is probably mostly a combination of legislative constraints and lack of experience. Not all European countries are EU- members and most European countries have national legislation regulating the use of chemicals in freshwater. Both local and national regulatory agencies seem not to know where to start and which legislation to apply when trying to implement an eradication strategy involving chemicals. The answer to why not in future is, while complex, now only down to making a strong case to secure political backing and funding to take eradications forward.

Legislation is now a reason for crayfish control

To date, legislation has probably been one of the greatest constraints. Many countries lack effective legislation to carry out pro-active management/control of invasive non-native crayfish species. Legislation controlling import and trade of crayfish species, as well as introduction to the wild has, on the other hand, existed in several European countries (Edsman, 2008; Holdich & Sibley, 2009; Stentiford, et al., 2010), although there is no international regulatory framework for the trade of live animals (Chucholl, 2013). At least in principle, legislation has prevented introduction, and controlled exploitation to reduce risk of spread. Legislation to allow action to control spread or attempt to eradicate once invasive crayfish have been illegally introduced has been missing in many countries. That has all changed with the introduction of the EU invasive alien species regulations, which provide member states with mechanisms to issue Species Control Orders, and the powers behind them to take direct action to eradicate high risk invasive species. We have yet to see how this regulation will be enforced.

Ability to eradicate

Where the threats have been recognised, there seems to have been willingness to take action within the regulatory agencies and conservation bodies on the ground. However, that has been hampered by the lack of legislative powers, scientific evidence base and funding. This has been combined with a lack of public will (and therefore pressure on government) to take action. In most cases, the impacts from invasive crayfish species were not seen or understood by the general public.

Lack of knowledge?

This has probably been an important reason for not performing eradications. Lack of knowledge has been in three main areas: 1. a clear understanding of the species biology, impacts and risks; 2. leading on from this, an understanding of recognised processes such as risk assessments, risk management assessments, invasive species action plans etc. to build a robust case for action; and 3. a lack of knowledge regarding effective tools and techniques to translate that into action. Most of these areas we have now largely addressed or are working to do so. When a bigger experience base has been built more countries probably will try to address the legislative and funding issues necessary to reduce the detrimental effects of invasive non-native crayfish.

Lack of experience?

Historically this has been a factor. Biocide based programmes have only fairly recently become an alternative. Conventional means to manage aquatic invasive fish and crustaceans have been netting, trapping, electrofishing, draining waterways and liming etc. All of the above methods have been trialled to attempt eradication of invasive alien crayfish, but none have achieved more than population reduction (Peay, 2001). Eradication has not been feasible using conventional methods and long-term control is not financially or operationally sustainable, because of the costs associated and work load necessary. As in Norway, that is now changing, and the expertise, tools and techniques we have developed for application of rotenone-based pesticides are directly transferable to application of biocides (synthetic or natural pyrethrins) for crayfish management. These methods have been trialled and found to be very effective if applied correctly (Sandodden & Bardal, 2010; Sandodden & Johnsen, 2010). Total eradication of invasive alien crayfish in Europe is no longer feasible, but emphasis should be placed on sustaining viable island populations of native crayfish and creating new ones. Eradication programmes should be made an option throughout Europe during identification and establishment of suitable island populations and areas. Knowledge and experience to carry out successful crayfish eradications exists. The new EU regulation 1143/2014 is a new tool for securing necessary local legislation and funding.

CONCLUSION

There seems to be an increase in governmental willingness in Norway to conduct chemical eradications when projects are feasible and have acceptable short term environmental impacts. The opportunities for successful eradications should be weighed against not only the environmental impact but also the size and complexity of the waters holding the introduced species. Both legislative and funding constraints seem less prominent as successful eradications have been confirmed. Time will show if this trend will spread throughout Europe.

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REFERENCES

Alderman, D.J., Holdich, D. and Reeve, I. (1990) 'Signal crayfish as vectors in crayfish plague in Britain'. *Aquaculture* 86: 3–6.

Burridge, L.E. and Haya, K. (1997). 'The lethality of pyrethrins to larvae and post-larvae of the American lobster (*Homarus americanus*)'. *Ecotoxicology and Environmental Safety* 38(2): 150–154.

Chucholl, C. (2013). 'Invaders for sale: trade and determinants of introduction of ornamental freshwater crayfish'. *Biological Invasions* 15: 125–141.

Edsman, L. (2008). 'The Swedish story about import of live crayfish'. *Bulletin Francais Pêche Pesticide* 372–373: 281–288.

Freeman, M.A., Turnbull, J.F., Yeomans, W.E. and Bean, C.W. (2010). 'Prospects for management strategies of invasive crayfish populations with an emphasis on biological control'. *Aquatic Conservation Marine Freshwater Ecosystem* 20: 211–223.

Gherardi, F. (2010). 'Invasive crayfish and freshwater fishes of the world'. *Revue Scientifique et Technique International Office of Epizootics* 29(2): 241–254.

Gherardi, F., Aquiloni, L., Diéguez-Uribeondo, J. and Tricarico, E. (2011). 'Managing invasive crayfish: Is there a hope?' *Aquatic Sciences*. 73: 185–200.

Guan, R. Z. (1994). 'Burrowing behaviour of the signal crayfish, *Pacifastacus leniusculus* (Dana), in the River Great Ouse, England'. *Freshwater Forum* 4: 155–168.

Gydemo, R. (1995). *Effect on an Insecticide Induced Crayfish Kill*. Report to the Swedish Environmental Protection Agency and the Swedish Board of Fisheries. Stockholm University, Sweden: Department of Systems Ecology.

Hiley, P.D. (2003). 'The Slow Quiet Invasion of Signal Crayfish (*Pacifastacus leniusculus*) in England – Prospects for the White-clawed Crayfish (*Austropotamobius pallipes*)'. In: D.M. Holdich and P.J. Sibley (eds.) *Management and Conservation of Crayfish: Proceedings of a Conference held in Nottingham on 7 November 2002*, pp. 127–138. Bristol: Environment Agency.

Holdich, D.M., Gydemo, R. and Rogers, D.W. (1999) 'A Review of Possible Methods for Controlling Nuisance Populations of Alien Crayfish'. In: F. Gherardi and D.M. Holdich (eds) *Crustacean Issues 11*, *Crayfish in Europe as Alien Species*, pp. 245–270. Rotterdam: A.A. Balkema.

Holdich, D.M. and Sibley, P.J. (2009). 'ICS and NICS in Britain in the 2000s'. In: J. Brickland, D.M. Holdich and E.M. Imhoff (eds.) *Crayfish Conservation in the British Isles. Proceedings of a Conference Held on 25th March 2009 in Leeds, UK,* pp. 13–33. Leeds, UK.

Holdich, D.M., James, J., Jackson C. and Peay, S. (2014). 'The North American signal crayfish, with particular reference to its success as an invasive species in Great Britain'. *Ethology Ecology and Evolution* 26: 232–262.

Johnsen, S.I., Vrålstad, T. and Sandodden, R. (2010). *Prosedyre ved Funn eller Mistanke om Introduksjon av Signalkreps – Iverksetting av Tiltak og Eventuell Friskmelding av Lokalitet*. Norwegian Institute of Nature Research Report 572. Lillehammer, Norway: Norwegian Institute of Nature Research.

Leahey, J.P. (1979). 'The metabolism and environmental degradation of the pyrethroid insecticides'. *Outlook on Agriculture*. 10: 121–133.

Ludke, J.L., Finley, M.T. and Lusk, C. (1971). 'Toxicity of Mirex to crayfish, *Procambarus blandinggi*'. *Bulletin of Environmental Contamination and Toxicol*ogy 6(1): 89–96.

Mayer, F.L. and Ellersieck. M.R. (1986). 'Manual of Acute Toxicity: Interpretation and Database for 410 Chemicals and 88 Species of Freshwater Animals'. *US Department of Interior Fish and Wildlife Resources Publication* 160. Washington, D.C.

McMahon, B.R. (2002). 'Physiological Adaptations to Environment'. In: D.M. Holdich (ed.) *Biology of Freshwater Crayfish,* pp. 327–376. Oxford, UK: Blackwell Science Ltd.

Morolli, C., Quaglio, F., Della Rocca, G., Malvisi, J. and Di Salvio, J. (2006). 'Evaluation of the toxicity of synthetic pyrethroids to red swamp crayfish (*Procambarus clarkia*, Girard 1852) and common carp (*Cyprinus carpio*, L 1758)'. *Bulletin Francais Pêche Pesticide*. 380–381: 1381–1394.

OIE (2009). *Aquatic Animal Health Code Twelfth Edition*. Paris, France: World Organisation for Animal Health.

O`Reilly, S. (2015). 'Assessing the Toxicity of Biocides on the North American Signal Crayfish *Pacifastacus leniusculus* (Dana) to Aid Eradication'. MSc(R) thesis. Glasgow: University of Glasgow.

Peay, S. (2001). *Eradication of Alien Crayfish Populations*. Environment Agency, Bristol: Environment Agency and English Nature, R&D Technical Report W1-037/TR1.

Peay, S. and Hiley, P.D. (2006). 'Biocide Trial to Eradicate Signal Crayfish in the North Esk Catchment'. *Scottish Natural Heritage Commissioned Report* No. 0122. Aberdeen, Scotland: Scottish Natural Heritage.

Peay, S., Hiley, P.D., Collen, P. and Martin, I. (2006). 'Biocide treatment of ponds in Scotland to eradicate signal crayfish'. *Bulletin Français de la Pêche et de la Pisciculture* 380–381: 1363–1379.

Ribbens, J.C.H. and Graham J.L. (2004). 'Strategy for the Containment and Possible Eradication of American Signal Crayfish (*Pacifastacus leninusculus*) in the River Dee Catchment and Skyre Burn Catchment, Dumfries and Galloway'. *Scottish Natural Heritage Commissioned Report* No. 014. Dumfries, Scotland: Scottish Natural Heritage.

Sandodden, R. and Bardal, H. (2010). 'Bekjempelse av Signalkreps (*Pasifastacus leniusculus*) på Ostøya i Bærum Kommune'. *Veterinærinstituttets rapportserie* 1-2010. Oslo: Veterinærinstituttet.

Sandodden, R. and Johnsen, S.I. (2010). 'Eradication of introduced signal crayfish *Pasifastacus leniusculus* using the pharmaceutical BETAMAX VET®'. *Aquatic Invasions* 5(1): 75–81.

Souty-Grosset, C., Holdich, D.M., Noël, P.Y., Reynolds, J.D. and Haffner, P. (eds.) (2006). *Atlas of Freshwater Crayfish in Europe*. Paris: Museum National d´Histoire Naturelle.

Stebbing, P., Longshaw, M. and Scott, A. (2014). 'Review of methods for the management of non-indigenous crayfish, with particular reference to Great Britain'. *Ethology Ecology and Evolution* 26: 204–231.

Stentiford, G.D., Oidtmann, B., Scott, A. and Peelerb, E.J. (2010). 'Crustacean diseases in European legislation: Implications for importing and exporting nations'. *Aquaculture* 306: 27–34.

Unestam, T. (1972). 'On the host range and origin of the crayfish plague fungus'. *Report of the Institute of Freshwater Research Drottningholm* 52: 192–198.

Van Wijngaarden, R.P.A., Brock, T.C.M. and Van den Brink, P.J. (2005). 'Threshold levels for effects of insecticides in freshwater ecosystems: A review'. *Ecotoxicology* 14: 355–380.

Vrålstad, T., Knutsen, A.K., Tengs, T. and Holst-Jensen, A. (2009). 'A quantitative TaqMan® MGB realtime polymerase chain reaction-based assay for detection of the causative agent of crayfish plague *Aphanomyces astaci*'. *Veterinary Microbiology* 137: 146–155.

Modelling invasive plant alien species richness in Tenerife (Canary Islands) using Bayesian Generalised Linear Spatial Models

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Abstract Biological invasions are one of the major threats to biodiversity, especially on islands where the number of endemic species is the highest despite their small area. In the Canary Islands, the relationships among invasive alien species (hereafter IAS) and their environmental and anthropogenic determinants have been thoroughly described but robust provisional models integrating species spatial autocorrelation and patterns of IAS communities are still lacking. In this study, we developed a Generalised Linear Spatial Model for Invasive Alien Species Richness (IASR) under a Bayesian framework, using a methodological approach that encompass GIS and geostatistical analysis. In this study, we hypothesised that the inclusion of spatial autocorrelation can improve model performance thus obtaining more IASR-reliable predictions. In addition, this method provides uncertainty maps that prioritize areas where further sampling efforts are needed. Our model showed that IASR in Tenerife is mainly driven by a combination of anthropogenic and natural processes, highlighting favourable conditions for IAS from the coastline to about 800 m a.s.l., especially on the windward humid aspect. Among anthropogenic factors, a clear positive relationship between road kernel density estimation and IASR was found. Indeed, road density has recently increased especially in low to mid altitudinal zones on the Canary Islands, strictly associated with urban expansion and it has been widely demonstrated to be one of the main IAS pathways. Hence, higher road density can be related to increased 'propagule pressure' which is, together with source of disturbance, one of the most important factors explaining richness in alien species invasion success. Our main conclusions highlight the importance of considering spatial autocorrelation and researchers' prior knowledge to increase the predictive power of statistical models. From a practical perspective, these models and their related uncertainty, will serve as important management tools highlighting those portions of territories that will be more prone to biological invasions and where monitoring efforts should be directed.

Keywords: biodiversity, biogeography, biological invasions, geostatistics, GIS, kriging

INTRODUCTION

Islands display unique ecological and evolutionary processes, hosting more than 20% of the terrestrial plant and vertebrate species in the world, within less than 5% of the global terrestrial area (Courchamp, et al., 2014). Endemics on islands are present with a magnitude higher than on continents (Kier, et al., 2009). In fact, more than one third of biodiversity hotspots in the world are entirely, or largely, within islands (Bellard, et al., 2014).

Besides their high diversity, islands host extremely fragile environments: 50 out of 80 of the documented plant extinctions in the last 400 years occurred on islands and more than 2000 endemic island taxa are currently thought to be on the verge of extinction (Ricketts, et al., 2005; Whittaker & Fernández-Palacios, 2007; Fernández-Palacios, et al., 2015). Nowhere in Europe is this pattern more conspicuous than in Macaronesia, the biogeographic region that encompasses the oceanic islands of the Azores, Madeira, the Canaries and the Cape Verde archipelago (Whittaker & Fernández-Palacios, 2007; Fernández-Palacios, et al., 2015). Macaronesia is widely recognised as an outstanding biodiversity hotspot worldwide due to its high rates of endemism in angiosperms and in bryophytes (40% and 6.5%, respectively, Whittaker & Fernández-Palacios, 2007).

Invasive Alien Species (IAS) pose a serious threat to the conservation of biodiversity and ecosystem integrity worldwide (DAISIE, 2009; Scalera, et al., 2012). Island systems, in fact, are extremely susceptible to biological invasions due to low habitat diversity, their simplified trophic webs and higher rate of endemic species (Courchamp, et al., 2003; Millennium Ecosystem Assessment, 2005; Vilà & Lopez-Darias, 2006; Barni, et al., 2012; Bacaro, et al., 2015).

Oceanic islands perform as an *open-air laboratory* in the field of invasion biology, because of their long history of large-scale anthropogenic disturbances and the recent introduction of non-native species (Whittaker & Fernández-Palacios, 2007; Denslow, et al., 2009), allowing us to generalise about the outcome of biotic invasions and to test the consistency of invasive organisms' behaviours (Kueffer, et al., 2010).

Several factors may determine the composition and the abundance of alien floras, including climate, geology, land use, landscape context, human impact, competition with natives and natural or anthropogenic disturbance and residence time (Crawley, 1987; Pyšek, et al., 2002; Arévalo, et al., 2005). Anthropogenic factors, such as inhabitants and trade networks, were imputed as main drivers of plant IAS introduction and spread: most populated islands should have more opportunities to import (and export) novel species due to the high rate of trade and transport with mainland areas (Pyšek, et al., 2010). Roads are anthropogenic features that can have greater influence on the distribution of IAS, particularly increasing the IAS propagule pressure (Lockwood, et al., 2005) or promoting the spread of generalist species with short life cycles and high reproductive rates (Parendes & Jones, 2000; Pauchard & Alaback, 2004; Arèvalo, et al., 2005; Dietz & Edwards, 2006; Arteaga, et al., 2009). In the Canary archipelago, as well as worldwide (Pauchard, et

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al., 2009), elevation and topography are factors driving the structure and distribution patterns of alien species spread (Arévalo, et al., 2005; Rejmánek, et al., 2005; Arteaga, et al., 2009)

Ecologists agree on the need for preventive tools such as early alert systems, given that control or eradication of already-established populations is more difficult and costly (Hobbs & Humphries, 1995; Bax, et al., 2001). Predictive invasion models, in fact, allow for evaluating the present and future extent of plant invasions. Furthermore, their outcomes are useful tools supporting the development of eradication/control programmes (Wace, 1977; Alpert, et al., 2000; Rejmánek & Pitcairn, 2002).

Spatial autocorrelation (SAC) is rarely included in ecological models thus potentially leading to biased parameter estimates. Furthermore, classic geostatistical models assume that data are Gaussian distributed, which may be an unrealistic assumption for count data, such as species richness. Generalised linear spatial models (GLSMs) provide a more robust model definition able to cope with response variables belonging to the exponential family distribution (Diggle, et al., 1998, 2003; Zhang, 2002; Christensen & Waagepetersen, 2002; Diggle & Ribeiro, 2007). By definition, the GLSM is a generalised linear mixed model in which the random effects are derived from a spatial process. The Bayesian approach allows parameter estimation by combining information coming from the observed data (via the likelihood function) as well as information coming from other prior sources (i.e. previous studies, subjective judgments) which is formalised through prior distributions. Therefore, Bayesian GLSMs (BGLSM) offer a flexible and robust approach for incorporating spatial correlation and prior knowledge into the modelling approach. In addition, the possibility of obtaining uncertainty maps may provide useful information where data are missing and further sampling efforts should be addressed. In this study, we hypothesised that the inclusion of SAC can improve model performance and therefore more reliable predictions, assuming that a variable selection process has been adopted. Specifically, we investigated alien species richness distribution on Tenerife (Canary Islands) using a multidisciplinary approach encompassing Geographic Information Systems (GIS), geostatistical calculation and statistical modelling. The main goals of this study are: i) to compute an ecologically and spatially reliable model of ASR spatial pattern in the island ii) to test if the inclusion of SAC into the modelling framework improves model performance.

MATERIALS AND METHODS

Study area

The study was carried out on Tenerife, the largest (2,033 km^2) island of the volcanic Canary archipelago situated in the subtropics ca. 70 km off the northwest coast of Africa (27–29° N, 13–18° W; Fig. 1). It is characterised by steep altitudinal gradient and it has a triangle-based pyramid shape with a truncated apex at 2,000 m a.s.l. at Las Cañadas, from which the volcano Teide rises (3,718 m a.s.l.)

The climate on Tenerife is semiarid to humid Mediterranean type (Arteaga, et al., 2009), with mean annual temperature reaching 19° C on the windward aspect and 21 °C on the leeward one. Mesoclimate is affected by trade winds that create a contrast between the northern and windward aspect (more humid and cloudy) and the southern and leeward aspect (more arid and cloudless).

Strong variation in elevation and aspect, which define local mesoclimatic zones and land use, are primary factors in structuring both native and alien plant communities on the Canary Islands (Whittaker & Fernández-Palacios, 2007). On Tenerife, vegetation can be simplified into five ecosystems based mainly on elevation and orientation gradients: succulent coastal scrub (0–700 m a.s.l.), thermophilous forest (200–600 m), laurel forest or *laurisilva* (500–1,000 m), Canarian pine forest (800–2,000 m), and summit or high-mountain scrub (> 2,000 m) (Fernández-Palacios, 1992; del Arco Aguilar, et al. 2006).

Statistical methods

Response variable

The distribution of Invasive Alien Species on the Canary Islands is available at ATLANTIS (Gobierno de Canarias, 2016). This database contains the occurrences of alien species within a grid of 500 m × 500 m square cells covering the entire archipelago. Species records span from 1970 to 2013. Invasive Alien Species Richness (IASR) on Tenerife was obtained by aggregating species occurrences in those ATLANTIS grid cells covering Tenerife Island land (5,514 cells out of 8,519 selected). Seventy-two species are present in the dataset (out of 701 alien species reported for the entire archipelago; Arechavaleta, et al., 2010).

Predictor variables

Three sets of abiotic variables, namely landscape, anthropogenic and climatic predictors, were derived in order to take into account all the the potential factors

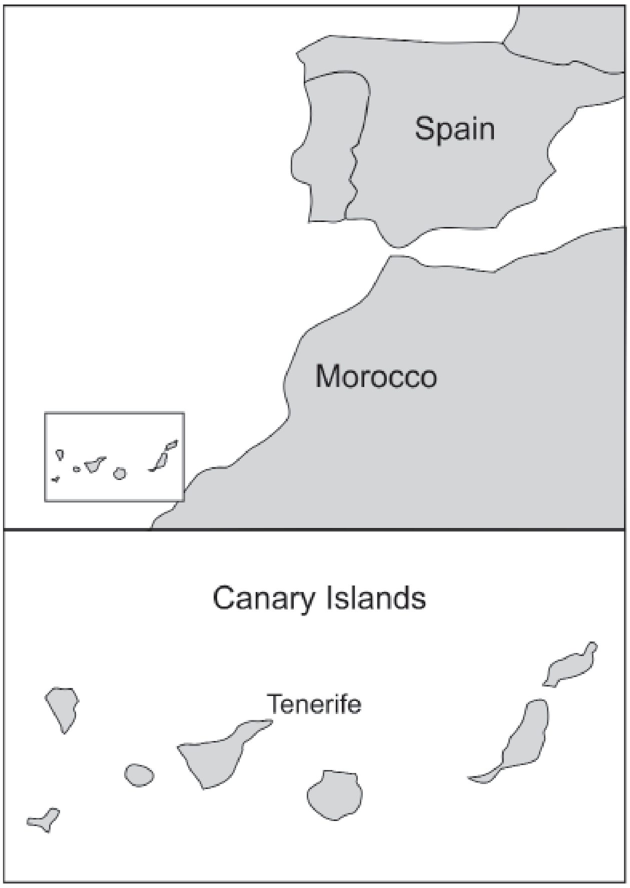

Fig. 1 Canary Islands and position of Tenerife Island within the Canary archipelago.

influencing alien species richness. Specifications of the variables chosen are addressed below.

Landscape predictors

The Digital Elevation Model (DEM) was downloaded from Cartográfica de Canarias S.A. (GRAFCAN, <https://www.grafcan.es/>). Aspect and slope were derived from the DEM for each 10 ×10 m pixel using QGIS 2.16.0 with GRASS 7.0.4 (Quantum GIS Development Team, 2016). The standard deviation of slope was calculated as an index of roughness (Grohmann, et al., 2010).

All the predictors were resampled to 500 m of spatial resolution using the nearest neighbour algorithm, accordingly to the spatial resolution of the species abundance grid. The relative abundance of vegetation classes (del Arco Aguilar, et al., 2010) within each cell was used to classify each grid cell, while the percentage of protected area per cell was used as a proxy of landscape nature conservation.

Anthropogenic predictors

As a proxy of anthropogenic impacts (e.g. fragmentation, Bacaro, et al., 2011) the Shannon index based on the relative abundance of land use classes within each cell was computed using the R package "*vegan"* (Oksanen, et al., 2017). We calculated a density proxy for roads using a Kernel density estimation (Rosenblatt, 1956; Parzen, 1962) using four regularly distributed classes of sample points on the road network distant from each other 5, 10, 20 and 50 km. As above, data were downloaded from Cartográfica de Canarias S.A.

Climate predictors

Climatic data were obtained from Agencia Estatal de Metereologia (AEMET) spanning from 2005 to 2014. Since recorded data showed many gaps throughout the entire time series of every single weather station, we used only those weather stations having records covering at least 80% of the full-time series for Precipitation (P) and 60% for Temperatures (T).

For each dataset mean annual (ma), mean seasonal (Winter: December, January, February (DJF); Spring: March, April, May (MAM); Summer: June, July, August (JJA)) were calculated. In order to obtain continuous representation of the phenomena, the co-kriging spatial interpolation technique (Myers, 1984) was applied using elevation, slope and aspect as covariates using "*geoR*" R package (Ribeiro & Diggle, 2001).

Data analysis and modelling

Spatial autocorrelation in explanatory variables was checked by computing Moran's I, using R package "*spdep*" (Bivand & Piras, 2015). In order to avoid multicollinearity, a forward variable selection with a double-stopping criterion approach (Blanchet, et al., 2008) was adopted in order to select the reduced set of predictors using "*adespatial*" (Dray, et al., 2017).

This procedure consists of computing the global model explained by all explanatory variables via a constrained ordination such as Redundancy Analysis and, if the resulting model is significant, calculating the adjusted coefficient of multiple determination (R^2_{adj}). Then variables were added to a null model (including only the intercept) using a forward procedure: the procedure stops when no more significant variables were founded (for a given alpha level) or when the R^2_{adj} of the model is greater than the global model R^2_{adj}. This double-stopping criterion should prevent the selection method from being too liberal and consequently inflating type I error rates. Once the reduced set of predictors was obtained, this was further evaluated via AIC comparisons using an iterative automatic routine (package "*glmulti*", Calcagno & de Mazancourt, 2010). The set of predictors thus obtained was then used for computing the BGLSM. The resulting model was used as a starting point for the BGLSM.

Unfortunately, probably as an effect of the high number of predictors retained in model selection, we came across issues in algorithm convergence. For this reason, we decided to further reduce the number of predictors chosen, among the reduced list previously obtained, to three which are known to be important drivers of the alien species community along the elevation gradient (Arévalo, et al., 2005, 2010; Barni, et al., 2012, Bacaro, et al., 2015). Thus, only the roads 10 km kernel density, P_{MAM} and elevation were included in the final model (Table 1).

To take into account the spatial correlation of count data, a BGLSM using the Langevin-Hastings Markov Chain Monte Carlo (MCMC) algorithm was computed using the "*geoRglm"* R package (Christensen & Ribeiro, 2002). To complete the Bayesian model formulation of the geostatistical models, a strong-informative uniform prior distribution (Rocchini, et al., 2017a) based on the result of the geostatistical model was specified. Simulations were run with the following specifications: four chains, 20,000 iterations, burn-in period of 6,000 iterations and a thinning rate of 100. To ensure a good mixing of the chains, convergences were assessed both visually and with Geweke's diagnostic (Geweke, 1992), along with the autocorrelation within the chains through "*coda*" package (Plummer, et al., 2006). The Bayesian framework also allows uncertainty of the model to be taken into account, that is the uncertainty of the prediction in the sampling units (Gelman & Hill, 2006). This statistic is crucial for correctly interpreting results and avoiding inappropriate decision-making.

Finally, the linear relationship between Predicted vs Observed IASR values was evaluated and the R^2 was calculated as a measure of goodness of fit.

All analyses were performed using the R 3.4 environment (R Core Team, 2017).

RESULTS

A total of 72 IAS were present in the dataset, with a mean of 4.18 species per cell (range: 1–27). The most common species on the island are *Opuntia maxima* (3,161

Table 1 Summary statistics of predictors used in the MAM. The variables units are shown in the last column.

	Mean	1st quantile	3rd quantile	Min	Max	Units
Roads 10 km kernel density	0.06523	0.02160	0.08953	0.00010	0.41928	-
$P_{MAM2005\text{-}2014}$	25.808	14.878	36.546	1.647	63.103	mm
Elevation	578.9934	240.0000	794.0460	0.5685	2421.3621	m

occurrences), *Ageratina adenophora* (2,239 occurrences) and *Ricinus communis* (1,615 occurrences). On average, the northern (and windward) part of the island has higher values than the southern (Fig. 2), where the biggest cities are (Santa Cruz de Tenerife and San Cristóbal de La Laguna on the NNE coast, Puerto de La Cruz on the NW coast). Furthermore, a decreasing altitudinal trend in IASR was also observed, with higher values of IASR near the coast and lower values above 1,500 m a.s.l. SAC in IASR values were confirmed by the Moran's Index value (I = 0.873, $p < 0.001$).

The inclusion of SAC in the minimum adequate model resulted in a consistent improvement in general model performance (ΔAIC 7,366). In the BGLMS, Markov chains show good mixing and convergence as highlighted by Geweke's diagnostics. Positive linear relationships of IASR were observed with road kernel density estimation (10 km) and $P_{MAM2005\text{-}2014}$, while elevation showed a negative trend (Table 2). Suitable areas for IAS appear to be located in urban areas, especially on the humid leeward aspect of the island (Fig. 3). Model output summarised in Table 3 and Fig. 4 shows the uncertainty in the predicted IASR

Table 2 Model output derived from the maximum likelihood analysis: τ^2 is the nugget, σ^2 is the sill, Φ is the range and gives information about the spatial autocorrelation of the sampling units.

Variables	Coefficients
Intercept	3.4966
Roads 10 km kernel density	18.7518
$P_{MAM2005\text{-}2014}$	0.0126
Elevation	-0.0017
τ^2	5.7100
σ^2	10.7700
Φ	2.8950

Table 3 Descriptive statistics of model outputs.

	Mean	1st quantile	3rd quantile	Min	Max
Predicted	3.47	1.29	4.44	0.33	27.39
Uncertainty	0.75	0.40	0.90	0.14	4.48

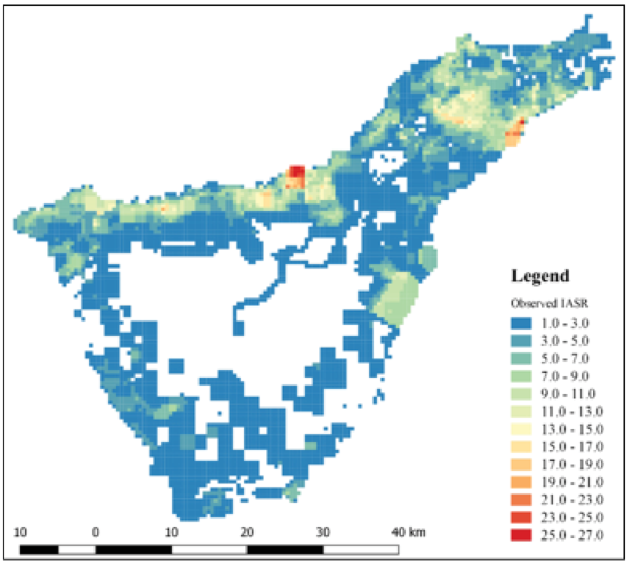

Fig. 2 Spatial pattern of alien species richness in Tenerife island.

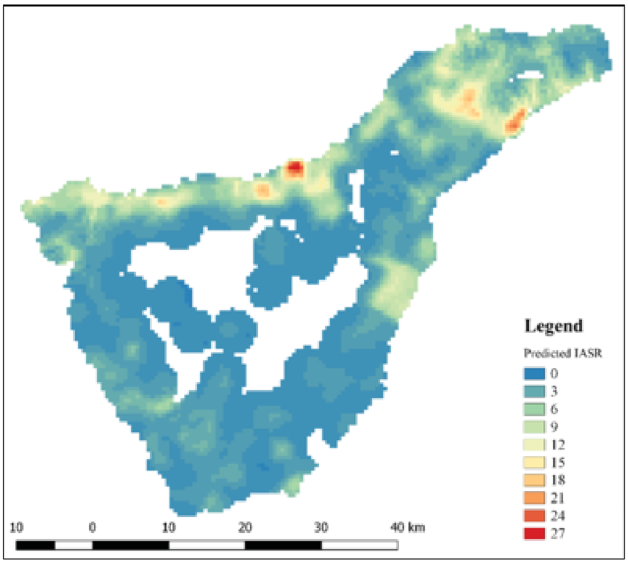

Fig. 3 Spatial pattern of predicted invasive alien species richness by BGLSM.

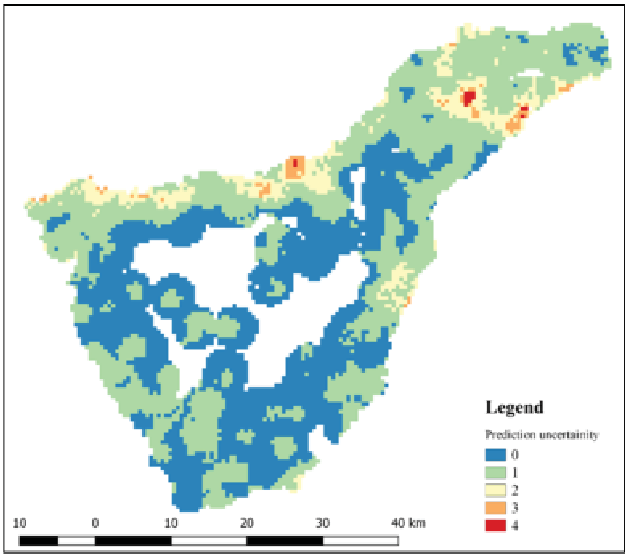

Fig. 4 Spatial pattern of uncertainty of the prediction of invasive alien species richness by BGLSM.

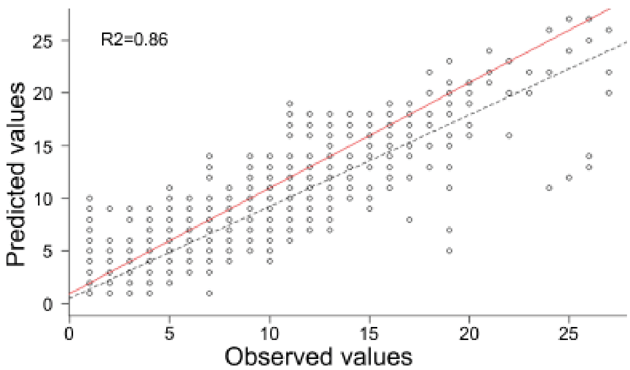

Fig. 5 Predicted vs. observed alien species richness. The solid line represents best prediction line, dashed line the fitted linear model.

values. Figure 5 shows the predicted *versus* observed values scatterplot suggesting a good performance of model fit.

DISCUSSION

The approach used in the selection of covariates and the incorporation of the spatial autocorrelation leads us to build a reliable ecological model to understand the IASR behaviour on Tenerife Island (Fig. 5). The outcomes of the model largely agree with most of the results previously published in the literature, taking into consideration both natural and anthropogenic processes. However, the ΔAIC suggests that incorporating SAC into GLM allows a consistent improvement in general model performance. Moreover, it allows us to obtain maps of the predictions that can be easily consulted by local governments. The map of uncertainty of the prediction provided in the Bayesian framework represents a powerful tool to highlight those areas where sampling efforts should be directed, providing valuable guidance in the decision-making process. On average, uncertainty in the model was quite low and evenly dispersed across the island. The areas where the uncertainty was higher are where human-related land uses occur, mainly in the arid coastal belt at low elevations (Fernández-Palacios & Nicolás, 1995; Rocchini, et al., 2017b).

The Canary Islands, particularly Tenerife Island, are chiefly characterised by a steep altitudinal gradient causing potential variations in several abiotic conditions such as water availability, temperature, precipitation, and solar radiation even over relatively short distances (Alexander, et al. 2009). IASR is inversely proportional with elevation as already observed in Arévalo, et al. (2005), Arteaga, et al. (2009) and Bacaro, et al. (2015), among others. The positive relationship between elevation and limiting factors such as drought, low temperatures and solar radiation were thoroughly investigated. Accordingly, it has been observed that at higher elevations, thermic and hydric stresses reduce the number of successful colonisations of alien species in different regions of the world (Fernandez-Palacios, 1992; Alpert, et al., 2000; Godfree, et al., 2004; Pauchard & Alaback, 2004; Becker, et al., 2005).

In general, mild environmental conditions associated with reduced drought stress enhance alien establishment and spread (Whittaker & Heegaard, 2003). These conditions were found at ca. 800–1,000 m a.s.l. (Arévalo, et al., 2005; Arteaga, et al., 2009). It has been observed that invasion success is mainly linked to the biogeographical affinities and environmental tolerances of the species (Wilson, et al., 1992; Arévalo, et al., 2005; Daehler, 2005). Accordingly, we found a positive relationship between P_{MAM} and IASR, and the BGLSM highlights as suitable the humid areas below 1000 m a.s.l, especially on the windward aspect, whereas the model did not predict suitable areas above 1500 m, except where roads are present. These findings reflect well the same pattern already observed in other studies performed both in Tenerife and in other oceanic islands (e.g. Arévalo, et al., 2005, Pauchard, et al., 2009; Bacaro, et al., 2015). In addition, Daehler (2005) observed similar patterns on the islands of Hawaii, where the relative importance of temperate species on the community composition increased strongly above 1,400 m a.s.l to detriment of the tropical ones.

Other authors (e.g. Nogués-Bravo, et al., 2008; Marini, et al., 2013) pointed out that relationships between IASR and anthropogenic factors are concentrated at low elevations, consequently increasing the opportunities for the introduction and establishment of propagules. Accordingly, our results showed a peak of alien species richness at a relatively low elevation. Species might have been introduced in the lowlands from different sources and in several historical periods. The kernel road density estimation shows a clear positive relationship with IASR. Bacaro, et al. (2015) reported that alien species were absent from plots located at higher elevation in plots sampled near the main Tenerife road network, consistent with previous observations (Pauchard & Alaback, 2004). Road density has increased especially in low to mid elevation belts of the Canary Islands, strictly associated with urban expansion and, consequently, to the spread of exotic plants. Roads may facilitate the dispersal of propagules of alien species via three main mechanisms: 1) as a source of disturbance that creates new environmental conditions that are suitable to ruderal and pioneer species; 2) they may facilitate the dispersal of propagules via air movement associated with the transit of vehicles; and 3) they may boost the rate of invasion by reducing competitiveness of native species that can cause the potential disappearance of even entire stands (Trombulak & Frissel, 2000; Bacaro, et al., 2015).

In this study, we assessed the incorporation of SAC into an ecological model built using ecologically reliable predictors. The incorporation of SAC improved general model performance and allows for for uncertainty to be accounted for in the model framework, providing a way to prioritize areas where more survey is needed along with further monitoring actions in order to reduce uncertainty.

Mild environmental conditions may be responsible for quick establishment and dispersal of aliens on islands. Accordingly, compared with current literature, our results showed higher alien species richness in mild environmental conditions and at a relatively low elevation. This can be also due to the fact that human land use is concentrated at low elevations, consequently increasing the opportunities for the introduction and establishment of propagules. To cope with plant alien species invasion, local governments have tried different approaches (Foxcroft, et al., 2007) but the most effective method still remains mechanical or hand removal (Gobierno de Canarias, 2014). In a global warming scenario, a modelling approach that takes into account spatial autocorrelation of data may play an even more crucial role in alien species monitoring, highlighting those portions of territories that are more prone to biological invasions, especially in fragile ecosystems such as in the Canary Islands.

REFERENCES

Alexander, J.M., Naylor, B., Poll, M., Edwards, P.J. and Dietz, H. (2009). 'Plant invasions along mountain roads: The altitudinal amplitude of alien Asteraceae forbs in their native and introduced ranges'. *Ecography* 32(2): 334–344.

Alpert, P., Bone, E. and Holzapfel, C. (2000). 'Invasiveness, invasibility and the role of environmental stress in the spread of non-native plants'. *Perspectives in Plant Ecology, Evolution and Systematics* 3(1): 52–66.

Arechavaleta, M., Rodríguez, S., Zurita, N. and García, A. (2010). *Lista de Especies Silvestres de Canarias. Hongos, Plantas y Animales Terrestres*. Santa Cruz de Tenerife, España: Gobierno de Canarias.

Arévalo, J.R., Delgado, J.D., Otto, R., Naranjo, A., Salas, M. and Fernández-Palacios, J.M. (2005). 'Distribution of alien vs. native plant species in roadside communities along an altitudinal gradient in Tenerife and Gran Canaria (Canary Islands)'. *Perspectives in Plant Ecology, Evolution and Systematics* 7(3): 185–202.

Arévalo, J.R., Otto, R., Escudero, C., Fernández-Lugo, S., Arteaga, M., Delgado, J.D. and Fernández-Palacios, J. M. (2010). 'Do anthropogenic corridors homogenize plant communities at a local scale? A case studied in Tenerife (Canary Islands)'. *Plant Ecology* 209(1): 23–35.

Arteaga, M.A., Delgado, J.D., Otto, R., Fernández-Palacios, J.M. and Arévalo, J.R. (2009). 'How do alien plants distribute along roads on oceanic islands? A case study in Tenerife, Canary Islands.' *Biological Invasions* 11(4): 1071–1086.

Bacaro, G., Santi, E., Rocchini, D., Pezzo, F., Puglisi, L. and Chiarucci, A. (2011). 'Geostatistical modelling of regional bird species richness: Exploring environmental proxies for conservation purpose'. *Biodiversity and Conservation* 20(8): 1677–1694.

Bacaro, G., Maccherini, S., Chiarucci, A., Jentsch, A., Rocchini, D., Torri, D., Gioria, M., Tordoni, E., Martellos, S., Altobelli, A., Otto, R., Escudero, C.G., Fernández-Lugo, S., Fernández-Palacios, J.M. and Arévalo, J.R. (2015). 'Distributional patterns of endemic, native and alien species along a roadside elevation gradient in Tenerife, Canary Islands'. *Community Ecology* 16(2): 223–234.

Barni, E., Bacaro, G., Falzoi, S., Spanna, F. and Siniscalco, C. (2012). 'Establishing climatic constraints shaping the distribution of alien plant species along the elevation gradient in the Alps'. *Plant Ecology* 213(5): 757–767.

Bax, N., Carlton, J.T., Mathews-Amos, A., Haedrich, R.L., Howarth, F.G., Purcell, J.E., Rieser, A. and Gray, A. (2001). 'The control of biological invasions in the world's oceans'. *Conservation Biology* 15(5): 1234–1246.

Becker, T., Dietz, H., Billeter, R., Buschmann, H. and Edwards, P.J. (2005). 'Altitudinal distribution of alien plant species in the Swiss Alps'. *Perspectives in Plant Ecology, Evolution and Systematics* 7(3): 173–183.

Bellard, C., Leclerc, C. and Courchamp, F. (2014). 'Impact of sea level rise on the 10 insular biodiversity hotspots'. *Global Ecology and Biogeography* 23(2): 203–212.

Bivand, R. and Piras, G. (2015). 'Comparing implementations of estimation methods for spatial econometrics'. *Journal of Statistical Software* 63(18): 1–36.

Blanchet, F.G., Legendre, P. and Borcard, D. (2008). 'Forward selection of explanatory variables'. *Ecology* 89(9): 2623–2632.

Calcagno, V. and de Mazancourt, C. (2010). 'glmulti: An R package for easy automated model selection with (generalized) linear models'. *Journal of Statistical Software* 34(12): 1–29.

Christensen, O.F., and Waagepetersen, R. (2002). 'Bayesian prediction of spatial count data using generalized linear mixed models'. *Biometrics* 58(2): 280–286.

Christensen, O.F. & Ribeiro Jr., P.J. (2002) 'geoRglm: A package for generalised linear spatial models'. *R News* 2 (2): 26–28.

Courchamp, F., Chapuis, J.L. and Pascal, M. (2003). 'Mammal invaders on islands: impact, control and control impact'. *Biological Reviews* 78(03): 347–383.

Courchamp, F., Hoffmann, B.D., Russell, J.C., Leclerc, C. and Bellard, C. (2014). 'Climate change, sea-level rise, and conservation: keeping island biodiversity afloat'. *Trends in Ecology and Evolution* 29(3): 127–130.

Crawley, M.J. (1987). 'What Makes a Community Invasible?'. In: A.J. Gray, M.J. Crawley, and P.J. Edwards (eds.) *Colonization, Succession and Stability,* 26th BES symposium 1987, pp. 429–453. Oxford: Blackwell Scientific.

Daehler, C.C. (2005). 'Upper-montane plant invasions in the Hawaiian Islands: patterns and opportunities'. *Perspectives in Plant Ecology, Evolution and Systematics* 7(3): 203–216.

DAISIE. (2009). *Handbook of Alien Species in Europe*. Berlin: Springer.

del Arco Aguilar, M.J., Pérez-de-Paz, P.L., Acebes, J.R., González-Mancebo, J.M., Reyes-Betancort, J.A., Bermejo, J.A., de Armas, S. and González-González, R. (2006). 'Bioclimatology and climatophilous vegetation of Tenerife (Canary Islands)'. *Annales Botanici Fennici* 43(3): 167–192.

del Arco Aguilar, M.J., González-González, R., Garzón-Machado, V. and Pizarro-Hernández, B. (2010). 'Actual and potential natural vegetation on the Canary Islands and its conservation status'. *Biodiversity and Conservation* 19(11): 3089–3140.

Denslow, J.S., Space, J.C. and Thomas, P.A. (2009). 'Invasive exotic plants in the tropical Pacific islands: patterns of diversity'. *Biotropica* 41(2): 162–170.

Dietz, H. and Edwards, P.J. (2006). 'Recognition that causal processes change during plant invasion helps explain conflicts in evidence'. *Ecology* 87(6): 1359–1367.

Diggle, P.J. and Ribeiro Jr., P.J. (2007) *Model-based Geostatistics*. New York: Springer.

Diggle, P.J., Tawn, J.A. and Moyeed, R.A. (1998). 'Model-based geostatistics'. *Journal of the Royal Statistical Society: Series C (Applied Statistics)* 47(3): 299–350.

Diggle, P.J., Ribeiro Jr, P.J. and Christensen, O.F. (2003). 'An Introduction to Model-based Geostatistics'. In: J. Møller (ed.) *Spatial Statistics and Computational Methods, Lecture Notes in Statistics Vol.173*. pp. 43–86. New York: Springer.

Dray, S., Blanchet, G., Borcard, D., Guenard, G., Jombart, T., Larocque, G., Legendre, P., Madi, N. and Wagner H.H. (2017). 'adespatial: Multivariate Multiscale Spatial Analysis'. R package version 0.0-8. <https://CRAN.R-project.org/package=adespatial>.

Fernández-Palacios, J.M. (1992). 'Climatic responses of plant species on Tenerife, the Canary Islands'. *Journal of Vegetation Science* 3(5): 595–603.

Fernández-Palacios, J. M., and Nicolás, J. P. (1995). Altitudinal pattern of vegetation variation on Tenerife. *Journal of Vegetation Science* 6(2), 183–190.

Fernández-Palacios, J.M., Rijsdijk, K.F., Norder, S.J., Otto, R., Nascimento, L., Fernández-Lugo, S., Tjørve, E. and Whittaker, R. J. (2015). 'Towards a glacial-sensitive model of island biogeography'. *Global Ecology and Biogeography* 25(7): 817–830.

Foxcroft, L.C., Hoffmann, J.H., Viljoen, J.J. and Kotze, J.J. (2007). 'Environmental factors influencing the distribution of *Opuntia stricta*, an invasive alien plant in the Kruger National Park, South Africa'. *South African Journal of Botany* 73(1): 109–112.

Gelman, A. and Hill, J. (2006). *Data Analysis Using Regression and Multilevel/Hierarchical Models.* Cambridge: Cambridge University Press.

Geweke, J. (1992). 'Evaluating the Accuracy of Sampling-Based Approaches to the Calculation of Posterior Moments'. In: J.M. Bernardo, J.O. Berger, A.P. Dawid and A.F.M. Smith (eds.) *Bayesian Statistics*, Vol. 4., pp. 169–193. Oxford: Clarendon Press.

Gobierno de Canarias (2014). 'Base de Datos de Especies Introducidas en Canarias'. <http://www.interreg-bionatura.com/especies/>. Accessed 31 March 2016.

Gobierno de Canarias (2016). 'Banco de Datos de Biodiversidad de Canarias'. <http//:www.biodiversidadcanarias.es/>. Accessed 31 March 2016.

Godfree, R., Lepschi, B. and Mallinson, D. (2004). 'Ecological filtering of exotic plants in an Australian sub-alpine environment'. *Journal of Vegetation Science* 15(2): 227–236.

Grohmann, C.H., Smith, M.J. and Riccomini, C. (2010). 'Multiscale analysis of topographic surface roughness in the Midland Valley, Scotland'. *Institute of Electrical and Electronics Engineers Transactions on Geoscience and Remote Sensing* 49(4): 1200–1213.

Hobbs, R.J. and Humphries, S.E. (1995). 'An integrated approach to the ecology and management of plant invasions'. *Conservation Biology* 9(4): 761–770.

Kier, G., Kreft, H., Lee, T.M., Jetz, W., Ibisch, P.L., Nowicki, C. and Barthlott, W. (2009). A global assessment of endemism and species richness across island and mainland regions. *Proceedings of the National Academy of Sciences* 106(23): 9322–9327.

Kueffer, C., Daehler, C.C., Torres-Santana, C.W., Lavergne, C., Meyer, J.Y., Otto, R. and Silva, L. (2010). 'A global comparison of plant invasions on oceanic islands'. *Perspectives in Plant Ecology, Evolution and Systematics* 12(2): 145–161.

Lockwood, J.L., Cassey, P. and Blackburn, T. (2005). 'The role of propagule pressure in explaining species invasions'. *Trends in Ecology and Evolution* 20(5): 223–228.

Marini, L., Bertolli, A., Bona, E., Federici, G., Martini, F., Prosser, F. and Bommarco, R. (2013). 'Beta-diversity patterns elucidate mechanisms of alien plant invasion in mountains'. *Global Ecology and Biogeography* 22(4): 450–460.

Millennium Ecosystem Assessment. (2005). *Ecosystems and Human Well-being: Biodiversity Synthesis*. Washington DC: World Resources Institute.

Myers, D.E. (1984). 'Co-kriging – New Developments'. In: G. Verly, M. David, A.G. Journel and A. Marechal (eds.) *Geostatistics for Natural Resources Characterization,* pp. 295–305. Netherlands: Springer.

Nogués-Bravo, D., Araújo, M.B., Romdal, T. and Rahbek, C. (2008). 'Scale effects and human impact on the elevational species richness gradients'. *Nature* 453(7192): 216–219.

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, R., Legendre, P., McGlinn, Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., Henry, M., Stevens, H., Szoecs, E. and Wagner, H. (2017). 'vegan: Community Ecology Package'. R package version 2.4-3. <https://CRAN.R-project.org/package=vegan>.

Parendes, L.A. and Jones, J.A. (2000). 'Role of light availability and dispersal in exotic plant invasion along roads and streams in the HJ Andrews Experimental Forest, Oregon'. *Conservation Biology* 14(1): 64–75.

Parzen, E. (1962). 'On estimation of a probability density function and mode'. *The Annals of Mathematical Statistics* 33(3): 1065–1076.

Pauchard, A. and Alaback, P.B. (2004). 'Influence of elevation, land use, and landscape context on patterns of alien plant invasions along roadsides in protected areas of South-Central Chile'. *Conservation Biology* 18(1): 238–248.

Pauchard, A., Kueffer, C., Dietz, H., Daehler, C.C., Alexander, J., Edwards, P.J. and Jakobs, G. (2009). 'Ain't no mountain high enough: Plant invasions reaching new elevations'. *Frontiers in Ecology and the Environment* 7(9): 479–486.

Plummer, M., Best, N., Cowles, K. and Vines, K. (2006). 'CODA: Convergence diagnosis and output analysis for MCMC'. *R News* 6: 7–11.

Pyšek, P., Jarošík, V. and Kučera, T. (2002). 'Patterns of invasion in temperate nature reserves'. *Biological Conservation* 104(1): 13–24.

Pyšek, P., Jarošík, V., Hulme, P. E., Kühn, I., Wild, J., Arianoutsou, M., Bacher, S., Chiron, F., Didziulis, V., Essli, F., Genovesi, P., Gherardi, F., Hejda, M., Kark, S., Lambdon, P.W., Desprez-Loustaun, M., Nentwig, W., Pergl, J., Poboljšaj, K., Rabitsch, W., Roques, A., Roy, D.B. Shirley, S., Solarz, W., Vilà, M. and Winter, M. (2010). 'Disentangling the role of environmental and human pressures on biological invasions across Europe'. *Proceedings of the National Academy of Sciences* 107(27): 12157–12162.

Quantum GIS Development Team (2016). 'Quantum GIS Geographic Information System'. Open Source Geospatial Foundation Project. < http://qgis.osgeo.org>.

R Core Team (2017). 'R: A Language and Environment for Statistical Computing'. R Foundation for Statistical Computing, Vienna, Austria. < http://www.R-project.org/>.

Rejmánek, M. and Pitcairn, M.J. (2002). 'When is eradication of exotic pest plants a realistic goal'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 249–253. IUCN ISSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Rejmánek, M., Richardson, D.M. and Pyšek, P. (2005). 'Plant invasions and invasibility of plant communities'. *Vegetation Ecology* 20: 332–355.

Ribeiro Jr, P.J. and Diggle, P.J. (2001). 'geoR: A package for geostatistical analysis'. *R News* 1(2): 14–18.

Ricketts, T.H., Dinerstein, E., Boucher, T., Brooks, T.M., Butchart, S.H., Hoffmann, M., Lamoreux, J.F., Morrison, J., Parr, M., Pilgrim, J.D., Rodrigues, A.S.L., Sechrest, W., Wallace, G.E., Berlin, K., Bielby, J., Burgess, N.D., Church, D.R., Cox, N., Knox, D., Loucks, C., Luck, G.W., Master, L. L., Moore, R., Naidoo, R., Ridgely, R., Schatz, G.E., Shire, G., Strand, H., Wettengel, W. and Wikramanayake, E. (2005). 'Pinpointing and preventing imminent extinctions'. *Proceedings of the National Academy of Sciences of the United States of America* 102(51): 18497–18501.

Rocchini, D., Garzon-Lopez, C.X., Marcantonio, M., Amici, V., Bacaro, G., Bastin, L., Chiarucci, A., Foody, G.M., Hauffe, H.C., He, K. S., Ricotta, C., Rizzoli, A. and Rosà, R. (2017a). 'Anticipating species distributions: Handling sampling effort bias under a Bayesian framework'. *Science of the Total Environment* 584: 282–290.

Rocchini, D., Bacaro, G., Chirici, G., Da Re, D., Feilhauer, H., Foody, G.M., Galluzzi, M., Garzon-Lopez, C.X., Gillespie, T.W., He, K.S., Lenoir, J., Marcantonio, M., Nagendra, H., Ricotta, C., Rommel, E., Schmidtlein, S., Skidmore, A.K., Van De Kerchove, R., Wegmann, M., Rugani, B. (2017b). 'Remotely sensed spatial heterogeneity as an exploratory tool for taxonomic and functional diversity study'. *Ecological Indicators* 85: 983–990.

Rosenblatt, M. (1956). 'Remarks on some nonparametric estimates of a density function'. *The Annals of Mathematical Statistics* 27(3): 832–837.

Scalera, R., Genovesi, P., Essl, F. and Rabitsch, W. (2012). 'The Impacts of Invasive Alien Species in Europe'. *European Environment Agency Technical Report* 16. Luxembourg: Publications Office of the European Union.

Trombulak, S.C. and Frissell, C.A. (2000). 'Review of ecological effects of roads on terrestrial and aquatic communities'. *Conservation Biology* 14(1): 18–30.

Vilà, M. and López-Darias, M. (2006). 'Contrasting biogeography of endemic and alien terrestrial species in the Canary Islands'. *Orsis: Organismes y Sistemes* 21: 91–101.

Wace, N. (1977). 'Assessment of dispersal of plant species—the car-borne flora in Canberra'. *Proceedings of the Ecological Society of Australia* 10: 167–186.

Whittaker, R.J. and Heegaard, E. (2003). 'What is the observed relationship between species richness and productivity? Comment'. *Ecology* 84(12): 3384–3390.

Whittaker, R.J. and Fernández-Palacios, J.M. (2007). *Island Biogeography: Ecology, Evolution, and Conservation*. Oxford: Oxford University Press.

Wilson, J.B., Rapson, G.L., Sykes, M.T., Watkins, A.J., and Williams, P.A. (1992). Distributions and climatic correlations of some exotic species along roadsides in South Island, New Zealand. *Journal of Biogeography* 19(2): 183–193.

Zhang, H. (2002). 'On estimation and prediction for spatial generalized linear mixed models'. *Biometrics* 58(1): 129–136.

Using expert knowledge and field surveys to guide management of an invasive alien palm in a Pacific Island lowland rainforest

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Abstract Invasive alien ornamental plants are a global problem, especially on oceanic islands, and can have severe impacts on native biodiversity. *Pinanga coronata,* is an ornamental palm tree that can form mono-dominant stands in its native habitat and is widely cultivated throughout the tropics. Here we investigate the introduction, spread, impact and management of this invasive palm in the Fiji Islands, using extensive discussions with local experts and field surveys. *Pinanga coronata* was introduced in the 1970s to the Colo-i-Suva area, eastern Viti Levu island, Fiji´s principal island, and has since become invasive in mahogany plantations and lowland rainforest. It has also been introduced and is becoming invasive on the western side of that island. However, the distribution of *P. coronata* remains geographically limited to the immediate vicinity of introduction sites but it is rapidly spreading. In each location, the species has formed mono-dominant stands in the understorey and appears to be displacing native plant species, as suggested by a negative correlation of its abundance with that of native tree ferns. This highlights the need for rapid control of *P. coronata* in Fiji. Local experts state management should involve manual removal of seedlings and saplings, killing of adult palms by injection of herbicide, and education and legislation to prevent the further spread of the species. Based on these recommendations and field data, management actions to control *P. coronata* are proposed and steps to develop these into a management plan are discussed. Given *P. coronata* threatens native biodiversity in Fiji and has the potential to invade other rainforest ecosystems in the tropics, proposed management approaches are urgent and relevant for other tropical countries.

Keywords: biodiversity loss, biological invasion, island biology, mahogany plantation, management plan, mono-dominant protected area, ornamental

INTRODUCTION

The tropical Pacific islands are highly vulnerable to the impacts of invasive alien plant species because of the region's geographic isolation and evolutionary traits that leave species vulnerable to competition (Gurevitch & Padilla, 2004; Denslow, et al., 2009; Caujapé-Castells, et al., 2010; Woinarski, 2010; Minden, et al., 2010b). The region has the greatest rate of increase in the number of invasive alien plant introductions with respect to area in the world (Van Kleunen, et al., 2015). Furthermore, invasive species are a major cause of extinction on Pacific Islands (Tye, 2009). With increasing economic development in nations of the Pacific, the diversity and impact of invasive plants in the region is likely to increase (Kueffer, et al., 2010).

Pinanga coronata, or ivory cane palm, is an ornamental palm tree that is cultivated and traded throughout the tropics (Palmpedia, 2017). The palm is native to lowland rainforests in Java and Sumatra (Kimura & Simbolon, 2002) and has been identified as a potentially invasive alien plant species on Pacific and other oceanic islands (Meyer, et al., 2008). Introduced to Fiji in the 1970s for its ornamental properties, the palm has become invasive in lowland rainforest and mahogany plantations (Keppel & Watling, 2011) but the current extent of its distribution is unknown. The Fiji Islands have a unique and highly diverse biota that is severely threatened by habitat loss, exploitation, pollution and invasive species (Myers, et al., 2000; Keppel, et al., 2014).

Pacific small island developing states (SIDS) have limited information, funding and trained professionals for invasive species management and conservation in general (Tye, 2009; Keppel, et al., 2012). However, local communities and experts often have extensive knowledge about their environment (Lefale, 2010; Keppel, 2014; Keppel, et al., 2015). In Fiji, expert knowledge plays a crucial role in conservation and protected area management (Keppel, 2014) and has provided important information about the conservation status of rare trees (Keppel, et al., 2015).

Using results from a quantitative field survey and qualitative expert knowledge, we demonstrate that *P. coronata* is threatening biodiversity and displacing native tree ferns in lowland rainforests and mahogany plantations. We then combine these two lines of evidence as the basis for a management framework to address the *P. coronata* invasion and its impact on native biodiversity in Fiji. Acknowledging the current challenges in the Pacific for invasive species management (Tye, 2009), the framework incorporates methods that are economically viable, develops capacity building and promotes awareness for all stakeholders related to the *P. coronata* invasion.

METHODS

Site description

The Fiji Islands include over 300 islands and islets in the western South Pacific (Mueller-Dombois & Fosberg, 1998). Fiji has a tropical climate with a wetter season from November to April and a drier season from May to October. Due to the south-east trade winds orographic rainfall produces higher precipitation in the south-east of topographically more complex islands (Mataki, et al.,

In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) (2019). Island invasives: scaling up to meet the challenge, pp. 417–423. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

2006). The capital of Fiji, Suva, is in the south-east of the archipelago's largest island, Viti Levu, and has an average annual rainfall of about 3,000 mm and an average surface temperature of 25.4°C (Mataki, et al., 2006).

The Colo-i-Suva area is approximately 12 km north of Suva (Fig. 1) and has four protected areas. The Colo-i-Suva Forest Park has no legal status but comprises the Colo-i-Suva Forest Reserve (FR; 370 ha) and the Maranisaqa-Wainiveiota FR (77 ha). Adjacent are the Savura FR (448 ha) and the Vago FR (24.7 ha). The area is mountainous and the vegetation communities are fragmented, lowland rainforest and mahogany plantations amongst agricultural and urban landscapes. The Savura and Vago FRs are mostly comprised of lowland native rainforest with minimal disturbance. These FRs constitute a major catchment for the Savura Creek, which secures the water supply for the capital city (Keppel, et al., 2005). Colo-i-Suva Forest Park is a mahogany plantation that has not been commercially logged since its establishment in the 1960s (Tuiwawa & Keppel, 2012). The Forest Park has conservation values because the mahogany plantations support a rich native understorey (Tuiwawa & Keppel, 2012), and is also frequented for recreational activities including local and international tourism (Malani, 2002).

Study species

Pinanga coronata, is native to western Indonesia, on Java and Sumatra where it is abundant in the rainforest understorey and occurs from sea-level to about 1800 m (Kimura & Simbolon, 2002). The species can be found on steep hillsides, lowland flats and exposed ridges, and juvenile palms are found at higher densities on lower slopes and moist areas (Kimura & Simbolon, 2002). Tolerating low light conditions, *P. coronata* forms mono-dominant clusters in the rainforest understorey, where it can reproduce sexually and asexually from vegetative shoots (Kimura & Simbolon, 2002; Kimura & Simbolon, 2003; Witono & Kondo, 2007). *Pinanga coronata* has rapid growth rates, reaches fecundity at <1 m in height, and can then continuously reproduce (Kimura & Simbolon, 2002).

Pinanga coronata has shown signs of becoming invasive in Hawaii and Tahiti (Daehler & Baker, 2006; Meyer, et al., 2008), but is not believed to currently be threatening biodiversity (US Forest Service, 2015). In Fiji, *Pinanga coronata* was introduced to the Colo-i-Suva area on Viti Levu during the 1970s for its ornamental properties

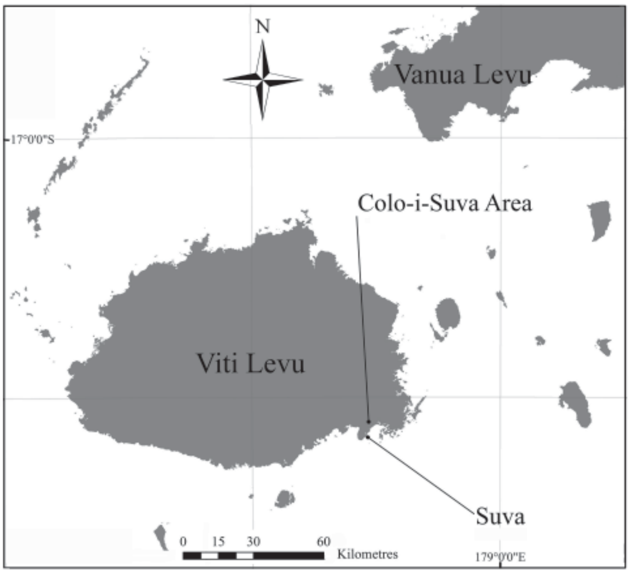

Fig. 1 Study locations on Fiji's largest island, Viti Levu.

(Keppel & Watling, 2011) and was first recognised as a potentially invasive species in 1992 (Watling & Chape, 1992). Since its introduction the palm has become dominant in the understorey of mahogany plantations (Watling, 2005) and invasive in native lowland rainforest, forming mono-dominant stands (mature palms and saplings) of several metres in diameter (Keppel & Watling, 2011).

Interviews

Experts were consulted by the lead author through informal discussions during fieldwork, and in semi-structured interviews in the office from July to September 2016, regarding the invasion history of *P. coronata*. Discussions were open ended but the key themes were the introduction history, distribution and dispersal, impact on native flora and recommended management of *P. coronata*.

Field survey

A systematic field survey was conducted in the Colo-i-Suva Forest Park and Savura FR. The aim was to identify areas of management priority and to determine if *P. coronata* is spreading. Using the Fishnet Tool in ArcMap 10.2.2, a 300 x 300 m grid with 11 columns and 15 rows was overlaid on the boundaries of Colo-i-Suva Forest Park and Savura FR. The centre point of each grid cell was imported into a Garmin Etrex 30® as waypoints. A 5 x 5 m plot was placed at each waypoint within the boundaries of the two forest reserves.

The abundance of *P. coronata* was recorded in ninety-two 5 x 5 m plots in Colo-i-Suva Forest Park (54 plots) and Savura FR (38 plots). The abundance of the palm was determined as the number of mature (stem > 1 m in height), juvenile (>0.5–<1.0 m) and seedling (< 0.5 m) palms, calculated by counting their numbers in each plot. The abundance of tree ferns was estimated by counting the number of mature (caudex > 1 m), juvenile (0.1–1 m) and tree fern saplings (< 0.1 m) (Ash, 1986; Ash, 1987). Additionally, opportunistic sightings of isolated *P. coronata* palms were recorded on a GPS Etrex 30®. Palms were only considered isolated if they were not near other *P. coronata* palms and were not a part of a mono-dominant stand.

RESULTS

Introduction history

Although there are no official records about the exact location and year *P. coronata* was introduced to Fiji, it was likely first introduced to a quarantine station north of Fiji's capital Suva (Fig. 2), for the propagation and trade of exotic palm trees. Palms were likely sold from this location to horticulturalists around Fiji. *P. coronata* is believed to have spread from the site through the surrounding, now cleared, mahogany plantation. Although the quarantine site has been abandoned and is now surrounded by an agricultural landscape, *P. coronata* is still present around the remains of the buildings.

The first official record of *P. coronata* in Fiji is a specimen in the South Pacific Regional Herbarium (number DA 18579) collected from the former Emperor Gold Mine guesthouse at Colo-i-Suva (about 2 km north of the former quarantine station) by Saula Vodonivalu on 16 February 1975. The habitat was described as a roadside and the specimen was flowering, with the tallest palm being approximately seven feet. This specimen originated from plantings around the mine's guesthouse, which were planted for ornamental purposes. This guesthouse was on the site of what is now an agricultural property that grows fresh produce for Joe's Farm supermarkets (Fig. 2). At some stage *P. coronata* was also introduced to a residential property within the interior of Colo-i-Suva Forest Park, which had a diverse collection of exotic ornamental plant species. This garden is still private property but the lease will return to the Ministry of Fisheries and Forests.

Distribution and dispersal

We believe that the distribution of *P. coronata* is currently restricted to the Colo-i-Suva area but is spreading rapidly. Our observations record that the species has now spread through the Savura and Vago FRs, occupying a total area of about 1,500 ha, and is most dense in the mahogany plantations near Joe's Farm and in the north of Colo-i-Suva Forest Park (Fig. 2).

P. coronata is cultivated ornamentally in several gardens in Suva, including the University of the South Pacific, Laucala Campus. It has escaped from cultivation in and around the Garden of the Sleeping Giant near Nadi Airport, on the western side of Viti Levu (Fig.1) and is distributed as an ornamental by landscapers and horticulturists in Suva and across Fiji, especially to hotels and tourism resorts. No estimates of the numbers of palms dispensed is available.

Within Colo-i-Suva Forest Park, we observed *P. coronata* to be most dense along streams and watercourses. We believe that one means of dispersal for the species is by seeds falling into waterways leading to establishment downstream. Once established near streams, *P. coronata* probably expands its distribution by moving up slopes bordering water courses.

Birds are believed to disperse *P. coronata* seeds. DW found a *P. coronata* seedling sprouted in his garden approximately nine kilometres from the introduction locations and main infestations. In the Colo-i-Suva area (Colo-i-Suva Forest Park and Savura FR) dense patches of *P. coronata* seedlings are commonly found below the canopy of tall native (especially *Gymnostoma vitiense*; Casuarinaceae) and exotic trees (*Maesopsis eminii*; Rhamnaceae) used as perching locations by native members of the Columbidae family, suggesting that fruits are eaten by birds that forage in the lower canopy and understorey . The island thrush, (*Turdus poliocephalus*) and the red-vented bulbul (*Pycnonotus cafer*) are likely dispersers in mature and open/edge forests, respectively.

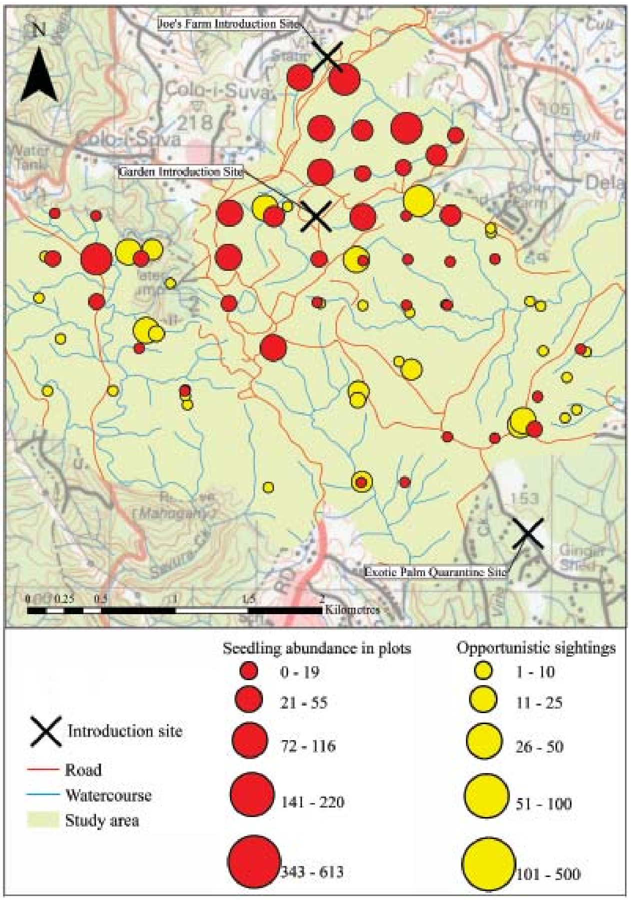

Fig. 2 The distribution of palm seedlings in plots and isolated populations in Colo-i-Suva Forest Park and Savura Forest Reserve.

Impact on native flora

We found mono-dominant stands of *P. coronata* in the understorey around all introduction sites in the Colo-i-Suva area. In the north of Colo-i-Suva Forest Park, *P. coronata* comprises up to 70% of the understorey and is outcompeting native understorey plants, especially tree ferns, and reducing their sapling regeneration (Mathieu 2015). Similarly, the palm is also abundant and outcompeting native species in the understorey of lowland rainforests. Therefore, *P. coronata* is considered to have the potential to become dominant and outcompete native plant species in Fiji's native lowland rainforests.

Pinanga coronata was present in 54 % of the plots surveyed in Colo-i-Suva Forest Park and 17 % of plots in Savura FR. It was mono-dominant in the understorey of 19 plots (21 % of all plots), 18 of which were in the north of Colo-i-Suva Forest Park (dominated by mahogany plantations) and the other was in the north of Savura FR (consisting of native lowland rainforest). Visual inspection of the distribution map (Fig. 2), shows the highest density near putative source locations and several isolated populations in both forest reserves.

Palm cover in the understorey displayed a strong negative correlation with all three tree fern classes, tree fern saplings ($\rho \geq -0.26$, $p < 1.2 \times 10^{-2}$), juvenile tree ferns ($\rho \geq -0.38$, $p < 3.7 \times 10^{-4}$) and mature tree ferns ($\rho \geq -0.33$, $p < 1.7 \times 10^{-3}$). With increasing palm cover in the understorey, the abundance of tree ferns decreased, especially when palm cover exceeded 50 % (Fig. 3). Therefore, results from the plots surveyed reinforce our field observation-based belief that *P. coronata* is displacing native species.

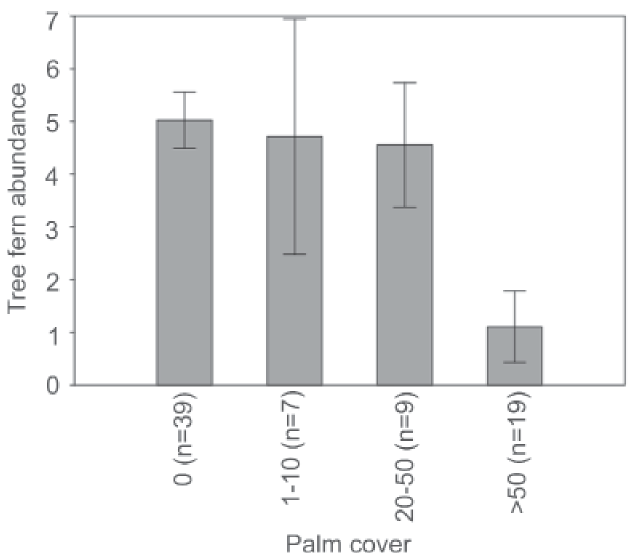

Fig. 3 The comparison of tree fern saplings and palm cover in the understorey. Tree fern abundance on the y-axis and palm cover on the x-axis. n = number of plots in the class.

Recommendation to manage *Pinanga coronata*

We believe that *P. coronata* must be listed as a pest species and be controlled as a matter of urgency. Control could be via manual removal of *P. coronata*, starting with seedlings and juvenile palms in isolated populations. Chemicals may be required to kill adult palms, because they are difficult to remove and the species can reproduce vegetatively from the base. The exotic tree *Maesopsis eminii*, which is spread throughout the Colo-i-Suva Forest Park, may need to be controlled concurrently, as this tree attracts frugivorous birds and *P. coronata* seedlings are often abundant at its base.

There also needs to be action to stop further propagation and reintroductions by horticulturalists. Furthermore, education for communities, tourism operators and the Biosecurity Authority of Fiji about the palm and its threats to native biodiversity is essential to solicit maximum support from the public and the government.

DISCUSSION

In our study, qualitative and subjective data are supported and reinforced by quantitative and objective field data. Combined, these results make a strong case that *P. coronata* is continuing to spread through mahogany plantations and native lowland rainforest in the Colo-i-Suva area. Both observations and a negative correlation between the abundance of palms and native tree ferns suggest that the introduced palm is displacing native tree ferns. Hence, both expert opinion and field data demonstrate the detrimental impact and potential threat of *P. coronata*, highlighting the need for swift and effective management actions.

However, before any management can be effectively implemented, knowledge of the exact distribution of *P. coronata* (Panetta & Lawes, 2005) and consensus among key stakeholders about the need for urgent management is needed. Knowing the palm's distribution in the Colo-i-Suva area will not only define the target area for management but also determine the stakeholders that need to be involved. Support from the most influential stakeholders will be essential for establishing and implementing a successful conservation and management plan (Keppel, et al., 2012; Moon, et al., 2015; Lenz, 2016). All major stakeholders, especially the Fiji Forestry Department and Biosecurity Authority of Fiji, need to agree that *P. coronata* is a major threat to native biodiversity and an urgent management priority. Assuming that these pre-requisites will be attained, we propose a management framework (Table 2) using a decision and risk analysis based on our knowledge and available quantitative data (Maguire, 2004; Stohlgren & Schnase, 2006; Lenz, 2016). We hope that proposing this framework will hasten the development and implementation of an effective management plan.

Management framework

Considering the high threat of *P. coronata* to native biodiversity and that its distribution is still relatively restricted, the overarching aim of a management plan should be to eradicate the species (Keppel & Watling, 2011). Eradication is defined as the total removal of all individuals, including seeds, and ensuring that reintroduction will not occur (Myers & Bazely, 2003; Meyer, 2014). However, there have been very few successful invasive plant eradications in the Pacific Islands and these were restricted to species confined to small geographic areas (Meyer, 2014). Additionally, eradication is not achievable without containment (Panetta & Lawes, 2005). Therefore, a feasibility study combining the known biological information of *P. coronata* with the total extent of the invasion (invasion syndrome) will need to be conducted to determine if eradication is achievable with the resources available (Panetta, 2015). Due to this uncertainty about the feasibility of eradication, we focus our discussion about management on control measures to reduce the abundance and spread of *P. coronata*.

Prior to control, stakeholders including the Ministry of Fisheries and Forests, NatureFiji-MareqetiViti (NFMV) and the University of the South Pacific (USP) should develop a management plan. The framework presented here could serve as a starting point. Coordinated efforts by multiple parties will be more efficient (Stohlgren & Schnase, 2006) and have a greater chance of success when decision makers for protected areas support the strategy (Foxcroft, et al., 2008). The involvement of NFMV is important because they have a strong record of effectively engaging with communities and decision-makers to achieve positive conservation outcomes (Morrison, et al., 2012). It is recommended that NFMV be the primary coordinators because invasive species management facilitated by a non-governmental organisation (NGO) that promotes education and stakeholder communication has greater chances of success (Epanchin-Niell, et al., 2010).

The second stage of management should aim to investigate the best method to control *P. coronata* through a feasibility trial. Currently there is a paucity of information on best practice for palm control (Meyer, et al., 2008) and most methods are species-specific (Langeland, et al., 2011). A pre-control feasibility study is necessary to determine which method will be the most effective (Meyer, 2014), as we have outlined in Table 1. Physical removal of palm seedlings is one of the methods to be trialled, as this has been shown to be successful at reducing seedling and juvenile abundance (Langeland, et al., 2011).

Table 1 Recommended control measures for *P. coronata* that should be trialled in a feasibility study based on literature on invasive palm management (Dovey, et al., 2004, Langeland & Stocker, 1997, Langeland, et al., 2011) and opinions from the authors. Suggestions from literature = LT and opinion from experts = AU. *Biocontrol is mentioned because it could be a successful control method if an appropriate control agent is found. However, biocontrol is not recommended at this stage and should only be considered if all other methods are ineffective and not feasible.

Management aim	Age target	Method
Control	Seedlings and juveniles	Hand pulling and removed from the area [AU]
Control	Seedlings and juveniles	Hand pulling and tied to a tree [AU]
Control	Mature individuals and clumps	Crown removal and apply herbicide to the stem [LT and AU]
	Mature individuals and clumps	Inject herbicide into the apical bud [LT]
Reduce seed load	All fruiting palms	Removal of flowers [LT]
Control		Biocontrol*[LT]

Table 2 Summary of the recommended management framework proposed to control *P. coronata* in the Colo-i-Suva area, on Viti Levu, Fiji. NFMV=NatureFiji-MareqetiViti.

Aim: Control *P. coronata* in the Colo-i-Suva area, on Viti Levu, Fiji	
Objectives	**Actions**
STAGE 1: Producing a management plan through stakeholder communication	
Create a management plan through stakeholder engagement NFMV to formalise the management plan and education programmes	Facilitate a formal discussion between stakeholders to develop a management plan. Develop an education programme for stakeholders and the community. Formalise a regular method for communication between stakeholders.
STAGE 2: Pre-control feasibility study	
Conduct a pre-control feasibility study	Trial different control methods (Table 1).
STAGE 3: Control *P. coronata* in the Colo-i-Suva area	
Control *P. coronata* in the Colo-i-Suva area	Target isolated and juvenile *P. coronata* populations. Remove the low-density populations in the centre and south-east of Colo-i-Suva Forest Park. Progressively control palms towards the dense populations in the north of the Colo-i-Suva area Simultaneously reduce the seed load and foliage area in the dense populations in the north of the reserves.
STAGE 4: Post-control monitoring and reducing the threat of reinvasion	
Post-control monitoring** Reduce the threat of reestablishment**	Periodically monitor areas where *P. coronata* has been controlled and investigate responses in native vegetation. Plant native tree ferns in areas that are vulnerable to reinvasion and monitor propagation success.
STAGE 5: Prohibiting the trade of *P. coronata* in local horticulture	
Ensure that *P. coronata* is not reintroduced into the natural environment	Ban the trade of *P. coronata* in the horticulture and tourism industries. This will require the species to be listed as a pest with involvement from the Biosecurity Authority of Fiji. Find a native non-invasive palm that can replace the trade of *P. coronata*.

**Stages three and four should be conducted simultaneously. After control efforts have removed isolated palms monitoring should take place before the dense *P. coronata* populations are managed.

There are two approaches commonly used with herbicide applications for managing clonal palms that should be trialled. The first method is cutting the palm below the crown and treating the cut stem with herbicide and the second method is injecting herbicide directly into the palm's apical bud (Langeland & Stocker, 1997). In Indonesia, densities of the invasive palm *Arenga obtusifolia* decreased and native rainforest vegetation successfully regenerated, when the palm was injected with herbicide (Konstant, 2014; Nardelli, 2016). A combined approach of applying chemical herbicides and physical removal could be implemented but has had varied success for palm species, like *P. coronata*, that can vegetatively reproduce (Langeland & Stocker, 1997; Langeland, et al., 2011).

After determining the most effective method, the third stage of management should attempt to control the spread and reduce the distribution of *P. coronata*. We consider that juvenile plants in isolated populations should be controlled first. Prioritising low-density populations will be the most efficient at containing the invasion and reducing the threat to endemic, rare and threatened plant species without significantly increasing the cost (Higgins, et al., 2000). Targeting mature and juvenile individuals in isolated populations is a recommended strategy for other invasive plants in the Pacific Islands and likely to be more successful than removing large stands (Meyer, et al., 2011).

Although not recommended as an initial control method for *P. coronata*, biocontrol may be required if physical efforts fail to control the spread. Dovey, et al., (2004) recommends the use of biocontrol in the Pacific Islands because it is resource efficient and can strengthen stakeholder partnerships. Biocontrol is typically applied when the distribution of an invasive plant is too large to be controlled by physical methods but is expensive and requires time-consuming host-specificity tests to ensure native plants from the same family will not be negatively affected (Meyer, 2014). However, reduced foliage cover of an invasive tree due to biocontrol has resulted in the regeneration of understorey species in other Pacific Island rainforests (Meyer, et al., 2012).

Monitoring (stage 4) should take place as control efforts (stage 3) of the different *P. coronata* populations progress. It is critical that management efforts are long-term and control sites are periodically monitored to ensure that the palm does not regenerate from its seed bank, the longevity of which is currently not known, and to understand changes in the vegetation community in response to efforts (Blossey, 1999; Foxcroft, et al., 2008). When invasive alien plants are removed from Pacific rainforests follow-up control efforts are often required (Minden, et al., 2010a, b).

Management should aim to stop *P. coronata* reinvading controlled areas. In healthy native forest ecosystems, the succession by native flora will naturally occur, but management may be required to reduce the likelihood of invasive plants re-establishing (Awanyo, et al., 2011). Planting native species is an expensive but effective method of reducing the risk of reinvasion (Langeland, et al., 2011). In the Colo-i-Suva region, planting tree ferns could be a novel and appropriate approach to reduce the likelihood of *P. coronata* re-establishing, because tree ferns are native and abundant in the area, especially on disturbed sites (Tuiwawa, 1999; Keppel, et al., 2005).

The final stage is to ensure that *P. coronata* is not introduced into the environment again. This would require the species to be listed as a pest plant under legislation outlined in the Biosecurity Promulgation 2008 act (Biosecurity Authority of Fiji, 2008), ideally at the beginning of the management process to provide legal support for any efforts (Lenz, 2016). Adequate enforcement of the legislation may require training and improved technical expertise within Biosecurity Authority of Fiji. Current palm stocks in local nurseries and ornamental plantings should be identified and controlled to ensure that *P. coronata* is not reintroduced (Meyer, et al., 2008; Lenz, 2016). The latter will be difficult on privately owned properties. Involving the horticultural industry is fundamental for success because they are integral in preventing continuous reintroduction through ongoing plantings (Meyer, et al., 2008).

CONCLUSION

Our opinions and field data agree on the considerable threat that *Pinanga coronata* is posing to native biodiversity. They also show that the palm is expanding its distribution and spreading into native rainforest ecosystems. There is little doubt that it will continue to do so, unless it is effectively and swiftly managed. Such management would require a thorough and effective management plan suitable to the SIDS in the Pacific and developed through participation by all key stakeholders. Given the evidence that the palm is threatening biodiversity, we propose a framework that could serve as a roadmap for developing and implementing a management plan.

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REFERENCES

Ash, J. (1986). 'Demography and production of *Leptopteris wilkesiana* (Osmundaceae), a tropical tree-fern from Fiji'. *Australian Journal of Botany* 34**:** 207–215.

Ash, J. (1987). 'Demography of *Cyathea hornei* (Cyatheaceae), a tropical tree-fern in Fiji'. *Australian Journal of Botany* 35**:** 331–342.

Awanyo, L., Attuah, E.M. and McCarron, M. (2011). 'Rehabilitation of forest-savannas in Ghana: The impacts of land use, shade, and invasive species on tree recruitment'. *Applied Geography* 31**:** 181–190.

Blossey, B. (1999).'Before, during and after: The need for long-term monitoring in invasive plant species management'. *Biological Invasions* 1**:** 301–311.

Caujapé-Castells, J., Tye, A., Crawford, D.J., Santos-Guerra, A., Sakai, A., Beaver, K., Lobin, W., Vincent Florens, F.B., Moura, M. and Jardim, R. (2010). 'Conservation of oceanic island floras: Present and future global challenges'. *Perspectives in Plant Ecology, Evolution and Systematics* 12**:** 107–129.

Daehler, C.C. and Baker, R.F. (2006). 'New records of naturalized and naturalizing plants around Lyon Arboretun, Manoa Valley, O'ahu'. *Bishop Museum Occasional Papers* 87**:** 3–18.

Denslow, J.S., Space, J.C., & Thomas, P.A. (2009). 'Invasive exotic plants in the tropical Pacific islands: patterns of diversity'. *Biotropica* 41(2): 162–170.

Dovey, L., Orapa, W. and Randall, S. (2004). 'The Need to Build Biological Control Capacity in the Pacific'. In: J.M. Cullen, D.T. Briese, D.J. Kriticos, W.M. Lonsdale, L. Morin, and J.K. Scott, (eds.) *Proceedings of the XI International Symposium on Biological Control of Weeds*, pp. 146–153. Canberra, Australia.

Epanchin-Niell, R.S., Hufford, M.B., Aslan, C.E., Sexton, J.P., Port, J.D. and Waring, T.M. (2010). 'Controlling invasive species in complex social landscapes'. *Frontiers in Ecology and the Environment* 8**:** 210–216.

Biosecurity Authority of Fiji. (2008). *Biosecurity Promulgation 2008, No. 28 of 2008*, Republic of Fiji Islands Government Gazette, Vol. 8, No. 94.

Foxcroft, L.C., Richardson, D.M. and Wilson, J.R. (2008). 'Ornamental plants as invasive aliens: Problems and solutions in Kruger National Park, South Africa'. *Environmental Management* 41**:** 32–51.

Gurevitch, J. and Padilla, D.K. (2004). 'Are invasive species a major cause of extinctions?'. *Trends in Ecology and Evolution* 19**:** 470–474.

Higgins, S.I., Richardson, D.M. and Cowling, R.M. (2000). 'Using a dynamic landscape model for planning the management of alien plant invasions'. *Ecological Applications,* 10**:** 1833–1848.

Keppel, G., Navuso, J.C., Naikatini, A., Thomas, N.T., Rounds, I.A., Osborne, T.A., Batinamu, N. and Senivasa, E. (2005). 'Botanical diversity at Savura, a lowland rain forest site along the PABITRA Gateway Transect, Viti Levu, Fiji'. *Pacific Science* 59**:** 175–191.

Keppel, G. and Watling, D. (2011). 'Ticking time bombs – current and potential future impacts of four invasive plant species on the biodiversity of lowland tropical rainforests in south-east Viti Levu, Fiji'. *The South Pacific Journal of Natural and Applied Sciences* 29**:** 42–45.

Keppel, G., Morrison, C., Watling, D., Tuiwawa, M.V. and Rounds, I.A. (2012). 'Conservation in tropical Pacific Island countries: why most current approaches are failing'. *Conservation Letters* 5**:** 256–265.

Keppel, G. (2014). 'The importance of expert knowledge in conservation planning – Comment to an article by C.J. Klein, et al.'. *Marine Policy* 48: 202–203.

Keppel, G., Morrison, C., Meyer, J.-Y. and Boehmer, H.J. (2014). 'Isolated and vulnerable: The history and future of Pacific island terrestrial biodiversity'. *Pacific Conservation Biology* 20: 136–145.

Keppel, G., Naikatini, A., Rounds, I.A., Pressey, R.L. and Thomas, N.T. (2015). 'Local and expert knowledge improve conservation assessment of rare and iconic Fijian tree species', *Pacific Conservation Biology* 21**:** 214–219.

Kimura, M. and Simbolon, H. (2002). 'Allometry and life history of a forest understory palm *Pinanga coronata* (Arecaceae) on Mount Halimun, West Java'. *Ecological Research* 17**:** 323–338.

Kimura, M. and Simbolon, H. (2003). 'Patterns of leaf and inflorescence production of *Pinanga coronata* (Arecaceae) in a tropical montane rainforest in West Java'. *Tropics* 12**:** 179–184.

Konstant, B. (2014). 'Indonesia: If you cut it, they will come'. *The Horn Spring* 12**:** 4.

Kueffer, C., Daehler, C.C., Torres-Santana, C.W., Lavergne, C., Meyer, J.-Y., Otto, R. and Silva, L. (2010). A global comparison of plant invasions on oceanic islands'. *Perspectives in Plant Ecology, Evolution and Systematics* 12**:** 145–161.

Langeland, K. and Stocker, R. (1997). ‚Control of non-native plants in natural areas of Florida'. University of Florida. Cooperative Extensions Services, report no. ST 242, Gainesville, Florida.

Langeland, K., Ferrell, J., Sellers, B., Macdonald, G. and Stocker, R. (2011). *Integrated management of non-native plants in natural areas of Florida*. Institute of Food and Agricultural Sciences, University of Florida, Gainseville, FL. SP 242**:** 1–27.

Lefale, P.F. (2010). ‘Ua ‘afa le Aso Stormy weather today: traditional ecological knowledge of weather and climate. The Samoa experience’. *Climatic Change* 100: 317–335.

Lenz, M. (2016): *Risk Assessment of the Invasive Alien Ivory-cane Palm* (Pinanga coronata) *in the Islands of the Tropical South Pacific.* The University of the South Pacific, Faculty of Science, Technology & Environment (FSTE) and University of Goettingen, Department of Geography, Germany.

Maguire, L.A. (2004). ‘What can decision analysis do for invasive species management?’. *Risk Analysis* 24**:** 859–868.

Malani, M. (2002). ‘Ecotourism in Fiji’. In*:* T. Hundloe (ed.) *Linking green productivity to ecotourism: Experiences in the Asia-Pacific Region*, 6**:** 45–55 . Tokyo: Asian Productivity Organization.

Mataki, M., Koshy, K.C. and Lal, M. (2006). ‘Baseline climatology of Viti Levu (Fiji) and current climatic trends’. *Pacific Science* 60**:** 49–68.

Mathieu, J. (2015): ‘Diversity of Understory Vegetation in a Submontane Tropical Rainforest under Impact of an Invasive Alien Palm Species. (Bachelor dissertation), Fiji: University of the South Pacific and Canada: University of Quebec.

Meyer, J-Y., Lavergne, C. and Hodel, D.R. (2008). ‘Time bombs in gardens: Invasive ornamental palms in tropical islands, with emphasis on French Polynesia (Pacific Ocean) and the Mascarenes (Indian Ocean)’. *Palms* 52: 71–83.

Meyer, J-Y., Loope, L. and Goarant, A. (2011). ‘Strategy to control the invasive alien tree *Miconia calvescens* in Pacific Islands: eradication, containment or something else’. In*:* C.R. Veitch, M.N. Clout and D.R. Towns (eds.), *Island invasives: eradication and management.* Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Meyer, J-Y., Fourdrigniez, M. and Taputuarai, R. (2012). ‘Restoring habitat for native and endemic plants through the introduction of a fungal pathogen to control the alien invasive tree *Miconia calvescens* in the island of Tahiti’. *BioControl* 57**:** 191–198.

Meyer, J.-Y. (2014). ‘Critical issues and new challenges for research and management of invasive plants in the Pacific Islands’. *Pacific Conservation Biology* 20**:** 146–164.

Minden, V., Hennenberg, K.J., Porembski, S. and Boehmer, H.J. (2010a). ‘Invasion and management of alien *Hedychium gardnerianum* (kahili ginger, Zingiberaceae) alter plant species composition of a montane rainforest on the island of Hawai’i’. *Plant Ecology* 206**:** 321–333.

Minden, V., Jacobi, J.D., Porembski, S. and Boehmer H.J. (2010b): ‘Effects of invasive alien kahili ginger (*Hedychium gardnerianum*) on native plant species regeneration in a Hawaiian rainforest’. *Applied Vegetation Science* 13 (1): 5–14.

Moon, K., Blackman, D.A. and Brewer, T.D. (2015). ‘Understanding and integrating knowledge to improve invasive species management’. *Biological Invasions* 17**:** 2675–2689.

Morrison, C., Rounds, I. and Watling, D. (2012). ‘Conservation and management of the endangered Fiji sago palm, *Metroxylon vitiense*, in Fiji’. *Environmental Management* 49**:** 929–941.

Mueller-Dombois, D. and Fosberg, R. (1998). *Vegetation of the tropical Pacific Islands.* New York: Springer.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G. and Kent, J. (2000). ‚Biodiversity hotspots for conservation priorities’. *Nature* 403**:** 853–858.

Myers, J.H. and Bazely, D. (2003). *Ecology and control of introduced plants*. Cambridge: Cambridge University Press.

Nardelli, F. (2016). ‘Current status and conservation prospects for the Javan rhinoceros *Rhinoceros sondaicus* Desmarest’. *International Zoo News* 63**:** 180–202.

Palmpedia. 2017. ‘*Pinanga coronata*’. *Palmpedia – Palm Grower's Guide.* <http://www.palmpedia.net/wiki/Pinanga_coronata> Accessed 5 November, 2016.

Panetta, F.D. and Lawes R. (2005). ‘Evaluation of weed eradication programs: the delimitation of extent’. *Diversity and Distributions* 11**:** 435–442.

Panetta, F. D. (2015). ‘Weed eradication feasibility: Lessons of the 21st century’. *Weed Research* 55**:** 435–442.

Stohlgren, T.J. and Schnase, J.L. (2006). ‘Risk analysis for biological hazards: What we need to know about invasive species’. *Risk Analysis* 26: 163–173.

Tuiwawa, M. (1999). ‘The Flora, Ecology and Conservation of the Botanical Biodiversity of Waisoi and the Southeasterm Slopes of the Korobasaga Range in Namosi Province, Fiji’. MSc thesis, Suva: University of the South Pacific.

Tuiwawa, S.H. and Keppel, G. (2012). ‘Species diversity, composition and the regeneration potential of native plants at the Wainiveiota Mahogany Plantation, Viti Levu, Fiji Islands’. *The South Pacific Journal of Natural and Applied Sciences* 30**:** 51–57.

Tye, A. (2009). *Guidelines for Invasive Species Management in the Pacific: A Pacific Strategy for Managing Pests, Weeds and Other Invasive Species*. Apia, Samoa: Secretariat of the Pacific Environment Programme.

US Forest Service. (2015). ‘*Pinanga coronata*’. Pacific Island Ecosystems at Risk (PIER) <http://www.hear.org/pier/species/pinanga_coronata.htm> Accessed 30 March 2016.

Van Kleunen, M., Dawson, W., Essl, F., Pergl, J., Winter, M., Weber, E., Kreft, H., Weigelt, P., Kartesz, J. and Nishino, M. (2015). ‚Global exchange and accumulation of non-native plants’. *Nature* 525**:** 100–103.

Watling, D. (2005). *Palms of the Fiji Islands,* Suva, Fiji: Environmental Consultants Fiji.

Watling, D. and Chape. S. (1992). *Fiji: State of the Environment Report*. Suva: Government of Fiji.

Witono, J.R. and Kondo, K. (2007). ‘Genetic relationships of *Pinanga coronata* accessions by RAPD markers’. *Chromosome Botany* 2**:** 93–97.

Woinarski, J. (2010). ‘Biodiversity conservation in tropical forest landscapes of Oceania’. *Biological Conservation* 143: 2385–2394.

Persistence, accuracy and timeliness: finding, mapping and managing non-native plant species on the island of South Georgia (South Atlantic)

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Abstract The South Georgia ecosystem-based habitat restoration project is a major project that began with the eradication of invasive rats (*Rattus norvegicus*) and reindeer (*Rangifer tarandus*), 2011–2017. As part of this restoration programme a non-native plant management strategy was developed and implemented. With only 8% of the whole South Georgia landmass suitable for vascular plants (ca. 283 km²) due to permanent ice and bare rock, there have been 25 indigenous vascular plants and 41 non-native plants recorded from earlier surveys. Following removal of grazing pressure from introduced mammals, surveys were conducted to quantify the current status and distribution of non-native plant populations and enable a non-native plant control strategy to be developed for the island. Due to the vast scale of the island, multiple seasons were required to carry out rapid surveys of key indicators such as species, area of plant coverage in square metres and age class (mature or juvenile). Survey and control data were entered into a spatial database to enable analysis, allow data-informed management decisions and be used for long-term control-based monitoring of outcomes. During this series of surveys, 44 naturalised, non-native plant species were identified and mapped. Of these, 34 species are now being managed at zero density with 56,851 m^2 at 184 sites controlled to date; four are managed at specific sites with 22,443 m^2 controlled to date, three require confirmation of species and the remaining three species are widely established and receive limited control. Spatially quantifying the distribution and control of non-native plants has enabled the development and implementation of an effective management strategy which contributes to the restoration of South Georgia's native biodiversity.

Keywords: Atlantic Ocean, control-based monitoring, habitat restoration, non-native plants, South Atlantic, South Georgia

INTRODUCTION

South Georgia (3,533 km², 54°21' S, 36°42' W) is located in the South Atlantic Ocean approximately 1,450 km south-east of the Falkland Islands (Fig. 1). South Georgia is a United Kingdom Overseas Territory (UKOT) managed by the Government of South Georgia and the South Sandwich Islands (GSGSSI). The island is mountainous and glaciated, and it is only the coastal fringes that are snow free in the summer months and able to support vegetation. An estimated 8% of the land mass of South Georgia (i.e. 283 km²) provides suitable habitat for vascular plants (GSGSSI GIS, 2007) and, in spite of the sub-Antarctic climate, many non-native species have naturalised or persisted for many years.

The first non-native plant species recorded on South Georgia was *Poa annua* in 1902 (Walton, et al., 1973) and this may have been introduced with early sealing expeditions. Increasing disturbance due to the activity of shore-based whaling operations after their establishment from 1904 (Burton, 2012) likely contributed to many of the later introductions. Greene (1964) classified 51 vascular species for the island with 24 as listed as native and 27 as non-native or introduced. There are now considered to be 25 native vascular species with the addition to Greene's list of the hybrid *Acaena magellanica* × *A. tenera* (Galbraith, 2011; Burton, 2012)

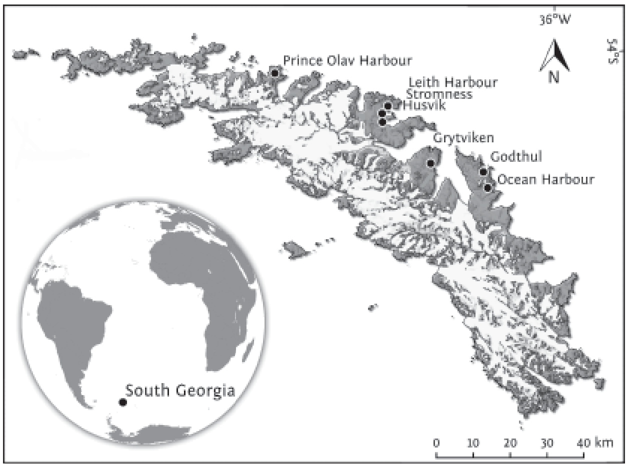

Fig. 1 Location of South Georgia Island and main whaling station locations on the island.

Osborne, et al. (2009) recorded 24 introduced vascular plant species during the survey undertaken in 2009 as part of the Royal Society for the Protection of Birds (RSPB) South Atlantic Invasive Species Project.

Local management of selected non-native plant species on South Georgia has been undertaken since 2004 when efforts to control bittercress (*Cardamine glacialis*) were initiated. In 2010 the efforts to control bittercress were increased and other non-natives were targeted at selected sites (GSGSSI, 2016).

In 2014, GSGSSI obtained funding from the UK Government-funded Darwin Plus initiative (www.darwininitiative.org.uk) for the project 'Strategic Management of Invasive Alien Plants on South Georgia'. This project enabled a more strategic approach to island-wide non-native plant control. As well as on-going control of low incidence species, comprehensive surveys were completed and the distribution and range of non-native plant species on the island were mapped. This paper outlines the processes that were undertaken to determine the extents of non-native plant species to support the development of a non-native plant management strategy for the island.

METHODS

Desktop review

The first step to determine the plant species present was a desktop review of all documents available that had location information for non-native species recorded on

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South Georgia. These included published and unpublished reports, herbarium records, and personal communications with researchers, staff and visitors. Many of these records had limited spatial accuracy and were recorded by general location only, which restricted the ability to map these records. Osborne, et al. (2009) had good spatial data for the sites they visited although they had not been able to access the restricted areas in some of the historic whaling stations. All the records found in the desktop review were compiled into a filterable dataset which generated a species list by locality which was then checked for presence/absence during any visits to that area. This dataset provided the initial non-native plant species list and search targets, with more recent records given a higher search priority. While not many of these records were very accurate spatially they gave a good starting point for surveying.

Field surveys

Using the compiled historical records, surveys were undertaken to obtain accurate information on the current abundance and distribution of non-native plant species in each search area. Priority survey locations were given to those with recent records, areas of human disturbance particularly whaling stations, and where reindeer had been present. All the surveys were conducted on foot with access to some areas, where required, provided by ship or small boat.

A rapid GPS survey technique was used to cover the large areas involved. The resolution for survey points was based on plant population size whereby the larger the population, the further apart points were recorded to reduce the time required for recording. GPS waypoints were taken at the centre of each non-native plant infestation with separate waypoints for each species present. Key indicators for each waypoint were recorded i.e. the plant species, the area of plant coverage in square metres, and age class (mature or juvenile not capable of reproduction). Coverage was estimated from the ground cover of the infestation, and is the ground area covered by the plant if forming a monoculture. In the case of scattered plants the percentage cover is used to estimate the total square meters of the infestation.

From the spatial information collected non-native species were classified into 5 classes depending on population size and distribution (Table 1). Rather than determine whether a non-native species was likely to be invasive, the precautionary principle (Williams, 1997) was adopted and all non-native vascular plants have been classified as part of the strategy.

The main surveys were conducted in February–March 2015 and in February–March 2016, when most non-native plants were in flower; this made it easier to assess distribution, and the flower structures provided the diagnostic characteristics to differentiate non-native species from closely related native plants.

Accurate geographical coordinates were essential for relocating infestations; coordinates were recorded using hand-held global positioning system receivers (Garmin GPS62 & 64). To manage these data a GPS Exchange Format file (gpx) import and export capability was developed in the recording database to facilitate data storage and display. Data were collected continuously during different control and survey visits as weather and logistics allowed more time to check areas more thoroughly. Additional data were collected where necessary during these visits to improve spatial knowledge of the infested areas and outlier plants.

Control based monitoring

Along with surveys, control was undertaken on selected known sites and all control activities were recorded using the same key indicators that had been used during survey data collection (GPS coordinates, coverage in square metres and age class), with the addition of the type of herbicide, the application rate used, and the volume of water used.

Management units

In order to manage the site-led control of Class Two species, South Georgia was divided into 117 management units. The management units were determined by a two-step process; firstly the island was divided into eight eco-geographic zones, defined primarily by climate, vegetation and the historic presence of introduced mammals (Martin, et al., 2009). For the purpose of non-native plant management, these zones were further divided into smaller units based on the level of historic human disturbance, presence of non-native plants, geographical features and ease of logistical access.

RESULTS

South Georgia's vegetation is mostly short grassland or low-growing rush and sedge communities, apart from the tall stands of coastal tussac (*Parodiochloa flabellata*). Many of the non-native plants are also low-growing which makes detection very difficult; persistent surveying is required to locate all individuals. Sometimes a number of visits are needed as many species are not very visible until flowering, and timing is critical to finding and controlling these species before seed becomes viable. New infestations and new non-native species have also continued to be found which highlights the need for persistent surveys. Due to the size of the island, multiple seasons were required to survey the priority areas. Following repeated surveys between

Table 1 Classification of non-native plants on South Georgia and number of non-native plants in each weed strategy class.

Class	Description	Number of species
One	Priority species; require species-led control at the island-wide level, to control all plants before they reach maturity. All sites with these species have a 'Site Tag' in the Weeds Database, for management of follow-up visits.	34
Class Two – Site-Led	Species of moderate distribution, requiring site-led control. Priority populations are those at high-use visitor sites, and sites with small infestations where control will reduce further dispersal.	4
Class Three – Site-Led	Species which are widespread and abundant, and require management at high-use visitor sites and at some remote outlier sites where appropriate.	3
Research	More information required before classification, to confirm status.	3
Historic	Historic species, not seen for at least 10 years. A re-sighting promotes the species to Species-Led – Class One.	35

2014 and 2017, we consider there to be 44 non-native species present on the island or that have recent records from the last 10 years, with a further 35 species recorded historically but no longer present (Appendix 1).

There have now been 4,245 non-native survey locations recorded to date. Following the survey these non-native species have been classified depending on population size and distribution (Table 1).

From the survey results a non-native plant management strategy (GSGSSI, 2016) and an associated environmental impact assessment were developed. In line with GSGSSI requirements, these documents were peer-reviewed to ensure they met best practice standards. After their finalisation, more widespread control of non-native plant species was undertaken across the island.

There are 34 Class One species occurring at 184 control sites; these are managed on a species-led basis by targeting them across their entire known range on South Georgia. Each of the Class One species is managed at zero density whereby all plants are controlled where found.

Fig. 2 shows the small increase in new Class One sites found and treated each season, along with the proportion that were active (some plants found) and not active (no plants seen at that site that season).

Control-based monitoring data show that 49,202 m^2 of Class One species have now been controlled on South Georgia with 850 m^2 of follow-up required in 2016/17 (Fig. 3). The majority of the treated species controlled in 2015/16 were *Rumex acetosella* since this was the most widespread Class One species and control was undertaken only once the full extent of the infested area of this species was known after surveys that season.

There are currently four Class Two species with 221 control locations, these records total 44,903 m^2 of plants treated over the seasons shown in Fig. 3.

DISCUSSION

Spatially quantifying the distribution and control of non-native plants has enabled the development and implementation of the 'South Georgia Non-Native Plant Management Strategy 2016-2020' (GSGSSI, 2016) which contributes to the restoration of South Georgia's native biodiversity.

Control of Class Two species is prioritised according to the potential dispersal risk posed by small populations and the threat they present to surrounding areas. Spatial data from the surveys overlaid with the units was essential in presenting this information to enable decision-making for the strategic management of the surveyed species.

There were 44 non-native plant species detected during the surveys and of these, 34 are currently being controlled using a range of methods with the aim being to eradicate them from South Georgia. Many remote areas have not been able to be visited yet, and although the risk of non–native infestations at these sites is considered to be low, based on historic records, all the vegetated areas of the island will eventually need to be surveyed. This may take many years due to the logistical difficulties of accessing the island's remote areas.

To ensure success of the non-native plant strategy, persistence is required in treating all target plants until their seed bank is fully diminished. Control-based monitoring will assist in determining success by utilising the data recorded on plant coverage, age class, herbicide rates and volumes in order to measure progress season by season.

We are confident that most of the non-native species and infestations have now been located. However, due to the large size of the areas to be searched, new records are not unexpected and the weed strategy has been designed to be adaptive based on the data available.

While all high priority areas for non-native species have now been surveyed, continued checks will be required to ensure all infestations are located around the island. Also, as vegetation communities are likely to recover from grazing following the reindeer eradication, further searches for non-native plants will be required across the estimated 4,500 ha of vegetated landscape on the Barff Peninsula and in the Stromness Bay area (3,250 ha) where the reindeer were present.

Monitoring new incursions and unknown infestations will be ongoing and this persistence can be achieved only if there is a long-term commitment to providing necessary resources, as is currently the case with the present control programme funded by the Government of South Georgia and the South Sandwich Islands.

Timeliness is also vital for ensuring that populations are successfully controlled. All control operations and surveys must take place during the optimum time for locating and treating non-native targets. For South Georgia, this is between December and February.

Accuracy is also essential, all target species need to be spatially documented using GPS waypoint data to aid in re-locating plants. While there are some small differences in estimating plant coverage by observers, regular comparisons between people improve accuracy and consistency of

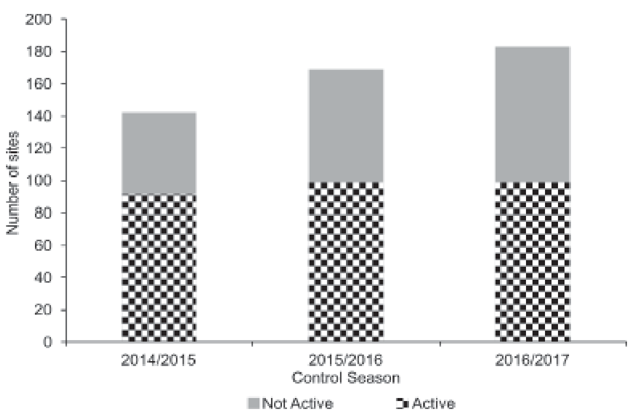

Fig. 2 Number of Class One non-native plant sites on South Georgia 2014–2017.

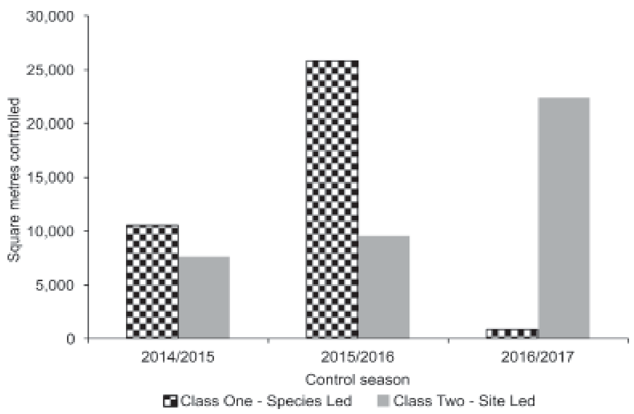

Fig. 3 Area of Class One and Class Two non-native plant sites controlled on South Georgia over the last three seasons (2014/15–2016/17).

measuring. Having data in quantifiable measures allows changes in the size and number of infested areas to be monitored as control efforts are undertaken. Control-based monitoring provides quantitative information for managing the target species and enables the comparison of control and survey data. This information will assist with further refinement of the management strategy and enable data driven decision making.

Finally, as with all eradication projects, strong biosecurity to prevent new introductions to South Georgia and the movement of already established non-native plant species between areas is essential. In South Georgia, there is a wide range of biosecurity measures in place from cargo packing facilities in the UK and mandated equipment cleaning before every landing to a bespoke biosecurity facility on the island itself. Ongoing education and awareness raising is key to ensure that all visitors to the island are aware of their biosecurity obligations and the vital role it plays in protecting native biodiversity.

ACKNOWLEDGEMENTS

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REFERENCES

Burton, R. (2012). *South Georgia*. Government of South Georgia and the South Sandwich Islands.

Galbraith, D. (2011). *A Field Guide to the Flora of South Georgia*. Great Britain: South Georgia Heritage Trust.

Greene, S.W. (1964). *The Vascular Flora of South Georgia*. British Antarctic Survey Scientific Report No. 45.

GSGSSI GIS (2007). *GIS Vegetation Layer*. NDVI data composed of NDVI calculations made on Landsat 8 Images. Government of South Georgia and the South Sandwich Islands.

GSGSSI (2016). *South Georgia Non-Native Plant Management Strategy 2016–2020*. Government of South Georgia and the South Sandwich Islands.

Martin, A.R., Poncet, S., Barbraud, C., Foster, E., Fretwell, P. and Rothery P. (2009). 'The white-chinned petrel (*Procellaria aequinoctialis*) on South Georgia: population size, distribution and global significance'. *Polar Biology* 32: 655.

Osborne, J., Borosova, R., Briggs, M. and Cable, S. (2009). *Introduced Vascular Plants: Survey for Baseline Information on Introduced Vascular Plants and Invertebrates: South Georgia.* Unpublished report to South Atlantic Invasive Species Project. The Lodge, Bedfordshire, UK: RSPB.

Walton, D.W.H. and Smith, R.I.L. (1973). 'Status of the Alien Flora of South Georgia.' *British Antarctic Survey Bulletin* 36: 79–97.

Williams, P. A. (1997). 'Ecology and Management of Invasive Weeds'. *Conservation Sciences* 7. Wellington, New Zealand: Department of Conservation.

Appendix 1 Naturalised non-native vascular plants on South Georgia species list, 2017.

Latin Name	Common Name	Family	Strategy Class
Achillea millefolium	yarrow	Asteraceae	One
Achillea ptarmica	sneezewort	Asteraceae	One
Agrostis vinealis	brown bent	Poaceae	One
Allium schoenoprasum	chives	Amaryllidaceae	One
Anthoxanthum odoratum	sweet vernal grass	Poaceae	One
Anthriscus sylvestris	cow parsley	Apiaceae	One
Capsella bursa-pastoris	shepherd's purse	Brassicaceae	One
Cardamine glacialis	bittercress	Brassicaceae	One
Carex aquatilis	water sedge	Cyperaceae	One
Carex nigra	common sedge	Cyperaceae	One
Carex sp.	sedge unknown (not flowering)	Cyperaceae	One
Carex vallis-pulchrae	marsh sedge	Cyperaceae	One
Dactylis glomerata	cocksfoot	Poaceae	One
Deschampsia cespitosa	tufted hair-grass	Poaceae	One
Deschampsia flexuosa	wavy hair-grass	Poaceae	One
Elytrigia repens	couch grass	Poaceae	One
Empetrum rubrum	diddle dee	Ericaceae	One
Festuca rubra	red fescue	Poaceae	One
Juncus filiformis	thread rush	Juncaceae	One
Leptinella scariosa	feathery buttonweed	Asteraceae	One
Lobelia pratiana	berry lobelia	Campanulaceae	One
Luzula multiflora var congesta	heath wood-rush	Juncaceae	One
Nardus stricta	mat grass	Poaceae	One
Ranunculus acris	meadow buttercup	Ranunculaceae	One
Ranunculus repens	creeping buttercup	Ranunculaceae	One
Rumex acetosella	sheep’s sorrel	Polygonaceae	One
Rumex crispus	curled dock	Polygonaceae	One
Sagina procumbens	pearlwort (procumbent)	Caryophyllaceae	One
Scorzonerioides autumnalis	autumn hawkbit	Asteraceae	One
Stellaria media	common chickweed	Caryophyllaceae	One
Trifolium repens	white clover	Fabaceae	One
Tripleurospermum inodorum	scentless mayweed	Asteraceae	One
Vaccinium vitis-idaea	cowberry	Ericaceae	One
Veronica serpyllifolia	thyme-leaved speedwell	Scrophulariaceae	One
Agrostis capillaris	common bent	Poaceae	Two
Deschampsia parvula	punk grass	Poaceae	Two
Poa pratensis	smooth meadow grass	Poaceae	Two
Trisetum spicatum	spike trisetum	Poaceae	Two
Cerastium fontanum	common mouse-ear	Caryophyllaceae	Three
Poa annua	annual meadow grass	Poaceae	Three
Taraxacum officinale	dandelion	Asteraceae	Three
Agrostis? unknown	unknown grass - TBC	Poaceae	Research
Galium saxatile	heath bedstraw	Rubiaceae	Research
Holcus lanatus	Yorkshire fog	Poaceae	Research
Aegilops sp.	goat grass	Poaceae	Historic
Alchemilla monticola	velvet lady’s mantle	Rosaceae	Historic

Appendix 1 (continued) Naturalised non-native vascular plants on South Georgia species list, 2017.

Latin Name	Common Name	Family	Strategy Class
Alopecurus geniculatus	marsh foxtail	Poaceae	Historic
Artemisia sp.	mugwort	Asteraceae	Historic
Avena fatua	wild-oat	Poaceae	Historic
Brassica cf. *napus*	rape	Brassicaceae	Historic
Carum carvi	caraway	Apiaceae	Historic
Centella sp.	centella	Apiaceae	Historic
Cerastium arvense	field mouse-ear	Caryophyllaceae	Historic
Daucus carota	carrot	Apiaceae	Historic
Festuca ovina	sheep's fescue	Poaceae	Historic
Hypericum tetrapterum	square-stemmed St John's-wort	Clusiaceae	Historic
Lactuca sp.	wild lettuce	Asteraceae	Historic
Lamium purpureum	red dead-nettle	Lamiaceae	Historic
Lolium multiflorum	Italian rye grass	Poaceae	Historic
Lolium temulentum	darnel ryegrass	Poaceae	Historic
Lotus corniculatus	bird's foot trefoil	Fabaceae	Historic
Lupinus sp.	lupin	Fabaceae	Historic
Matricaria discoidea	pineapple weed	Asteraceae	Historic
Phleum pratense	timothy grass	Poaceae	Historic
Pisum sativum	pea	Fabaceae	Historic
Plantago sp.	hoary plantain	Plantaginaceae	Historic
Poa trivialis	rough meadow grass	Poaceae	Historic
Raphanus sp.	radish	Brassicaceae	Historic
Rorippa islandica	northern yellow-cress	Brassicaceae	Historic
Rumex alpinus	alpine dock	Polygonaceae	Historic
Senecio vulgaris	common groundsel	Asteraceae	Historic
Sinapis arvensis	charlock	Brassicaceae	Historic
Solanum tuberosum	potato	Solanaceae	Historic
Sonchus sp.	sow thistle	Asteraceae	Historic
Stellaria graminea	grass leaf starwort	Caryophyllaceae	Historic
Thlaspi arvense	field penny-cress	Brassicaceae	Historic
Trifolium hybridum	alsike clover	Fabaceae	Historic
Urtica dioica	common nettle	Urticaceae	Historic
Urtica urens	annual nettle	Urticaceae	Historic

Eradication programmes complicated by long-lived seed banks: lessons learnt from 15 years of miconia control on O'ahu Island, Hawai'i

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Abstract The invasive tree *Miconia calvescens* (Melastomataceae) is a priority for control on the Hawaiian Island of Oʻahu due to its potential to replace native ʻōhiʻa (*Metrosideros polymorpha*, Myrtaceae) forests and degrade watershed function if allowed to establish. The Oʻahu Invasive Species Committee (OISC) is attempting to eradicate this species from the island of Oʻahu. OISC uses a buffer strategy based on estimated seed dispersal distance to determine the area under surveillance. This strategy has worked well enough to suppress the number of trees reaching reproductive age. The number of mature trees removed annually is now less than the number initially removed when the programme started in 2001. In 2016, just 12 mature trees were removed from 54.71 km^2 surveyed compared to 2002, when 40 mature trees were removed from 8.26 km^2 surveyed, a 96% drop in mature trees per square kilometre surveyed. However, miconia has a long-lived seed bank and can germinate after 20 years of dormancy in the soil. Funding shortages and gaps in surveys due to refusal of private property owners to allow access have resulted in some long-range extensions. OISC's results suggest that seed bank longevity is an important factor when prioritising invasive species risk and that allocating more resources at the beginning of a programme to eradicate a species with long-lived seed banks may be a better strategy than starting small and expanding.

Keywords: invasive species, invasive plants, watershed, outreach, cloud water interception, *Miconia calvescens, Metrosideros polymorpha*

INTRODUCTION

The tree miconia (*Miconia calvescens* – Melastomataceae) has been recognised as a threat to forests on Pacific Islands where it has been introduced (Meyer, et al., 2011; Medeiros, et al., 1997). Native to tropical Central and South America, it is under control programmes in French Polynesia, New Caledonia and Hawaiʻi (Meyer, et al., 2011). In areas where miconia has invaded, it has formed monospecific stands, shading out all plant species beneath it (Meyer, 1996). A miconia-dominated forest would likely not perform watershed services as well as Hawaiʻi's multi-layered native forests. Runoff and water would likely increase and replenishment of the islands' freshwater aquifer through cloud water interception would likely decrease. (Nanko, et al., 2013; Takahashi, et al., 2011). Because of its potential to outcompete native forest flora and its potential deleterious effects on watershed function, miconia has been prioritised for eradication on the Hawaiian Island of Oʻahu. Miconia was introduced to Oʻahu at the Wahiawā botanical garden in 1961 (Medeiros, et al., 1997). It was not until the late 1990s that its invasive potential became known and efforts to control it began (Medeiros, et. al., 1997). Here we describe the results of the island-wide eradication programme for miconia implemented by the Oʻahu Invasive Species Committee (OISC) since 2002.

The Koʻolau Range forms the eastern spine of the island of Oʻahu and is the location of the island's primary aquifers supplying water to the urban centre of Honolulu (Board of Water Supply, 2016). Data from miconia's native and invaded ranges shows that this species occurs in tropical areas with more than 1,500 mm of rainfall (Libeau, et al., 2017). Oʻahu rainfall data indicates that most of the Koʻolau Range, including the areas encompassing the island's most important aquifers, could support miconia (Giambelluca, et al., 2013).

Miconia's potential to replace forest ecosystems with monospecific stands is evident from its invasion history in French Polynesia (Meyer, et al., 2011). There, dense stands occur over 80,000 ha from sea level to 1,400 m (Meyer, et al., 2011). To put those numbers in perspective, the forested area of the Koʻolau Mountains is approximately 40,469 ha and its highest peak is 960 m (Koʻolau Mountain Watershed Partnership, 2017). The rainfall and elevation of the Koʻolau Range are similar to those areas in Tahiti where miconia has formed monospecific stands and is therefore vulnerable to the transformative effects of a miconia invasion (Fig. 1).

Miconia leaves can reach up to one metre in length (Chimera, et al., 2000) (Fig. 2). These large leaves reduce light levels so dramatically that understorey and groundcover vegetation under a miconia canopy are severely reduced (Meyer 2004; Nanko, et al., 2013). Rainwater collects on the large leaves and funnels it to

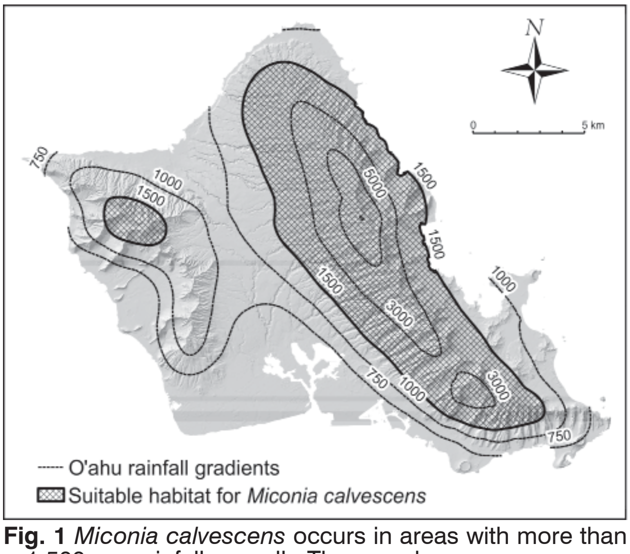

Fig. 1 *Miconia calvescens* occurs in areas with more than 1,500 mm rainfall annually. The area shown encompasses almost all of the Ko'olau mountain range on the eastern side of the island. (Rainfall data from Giambelluca, et al., 2011)

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the bare ground with a velocity high enough to accelerate erosion when hitting bare ground (Nanko, et al., 2013).

Water recharge of the island's aquifers may also be at risk. A study on Hawaiʻi Island found that native-dominated ʻōhi'a forest intercepted 454 mm more cloud water than strawberry guava (*Psidium cattleyanum* – Myrtaceae) dominated forests due to the differences in bark structure and tree shape (Takahashi, et al., 2011). Miconia has smooth bark similar to strawberry guava and would likely have similar rates of cloud water interception. This is important as cloud water interception may contribute up to 32% of total precipitation in Hawaiʻi's montane wet forests (Giambelluca, et al., 2011). Based on these studies, we surmise that a structurally complex, native forest is likely better at condensing fog and cloud drip and directing rain into the islands' aquifer than a forest dominated by monospecific stands of miconia.

CONTROL OF MICONIA ON OʻAHU

Control of miconia in the Hawaiian Islands began in 1991 after scientists and conservationists saw the damage it was causing in Tahiti (Medeiros, et al., 1997). On Oʻahu, miconia was planted at three botanical gardens, at two private residences and a commercially operated park (Medeiros, et al., 1997). All voluntarily destroyed their trees when requested by the state Departments of Agriculture (HDOA) and Land and Natural Resources (DLNR). Follow-up surveillance and control were conducted by volunteers and HDOA and DLNR employees until the Oʻahu Invasive Species Committee was formed as a project of the University of Hawaiʻi in 2001.

The miconia eradication project strategy is based on delimitation, defined as conducting enough surveillance to be sure that we know how far the invasion extends; containment, defined as containing the population by removing plants before they can mature; and extirpation, defined as removing immatures until the seed bank is exhausted (Panetta & Lawes, 2005; Panetta, 2007). In order to achieve the benchmarks of delimitation, containment and extirpation, OISC designates areas within a certain radius around reproductive trees for ground or helicopter surveys and conducts outreach to property owners and outdoor enthusiasts. The search area is currently at 91.39 km^2 and encompasses 4,000 different private property lots for which we must acquire permission to access in order to survey (Fig. 3).

Ground surveys are conducted for 800 m around every mature tree and 500 m around every immature tree every three years. The 500 m or 800 m radius around trees is called the ground buffer. An analysis of OISC's miconia field data shows that 99% of immature trees fall within 350 m of a mature tree (Fujkawa, pers. comm. 2017), confirming that the size of the search area is large enough.

Fig. 2 Typical leaf size for *Miconia calvescens*.

The frequency interval of every three years is necessary since miconia can mature in as little as four years (Meyer, 1996). Areas within this 800 m ground buffer that are too steep to survey by ground are surveyed by helicopter every two years.

Ground crews locate miconia visually during both ground and air surveys. In addition to their large size, miconia's leaves have vibrant purple undersides and this makes it fairly easy to detect on both types of surveys

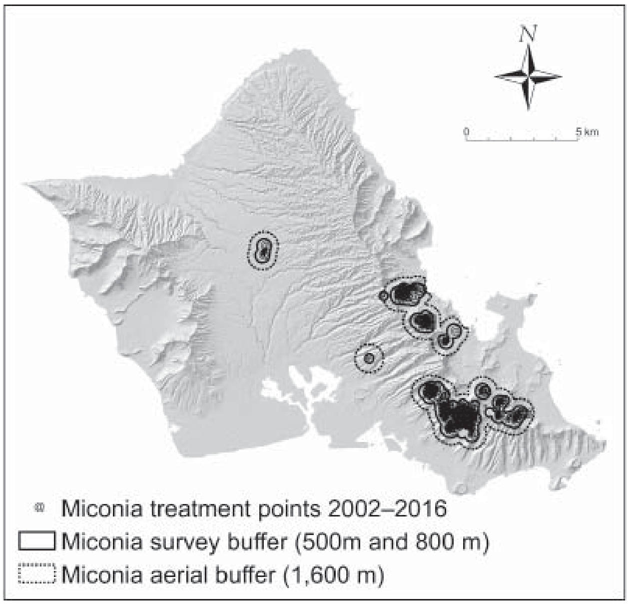

Fig. 3 The O'ahu Invasive Species Committee search area for *Miconia calvescens.*

Fig. 4 A sapling growing out of a patch of *Clidemia hirta* on O'ahu. The large leaves and striking purple undersides make miconia fairly detectable, although in heavy vegetation, trees can still be missed.

(Fig. 4). However, on ground surveys trees are sometimes missed due to dense vegetation that limits visibility. Steep terrain can also make trees difficult to detect as simply getting up a vertical slope safely may distract the surveyor from finding trees. Large trees that have already breached the canopy are also difficult to spot from the ground.

Surveyors can visually find miconia trees from a helicopter but the helicopter must fly very low and slow above the canopy. The large leaves are visible from the air and the rotor wash from the helicopter often blows the leaves around so the purple undersides are visible. Immature trees as tall as four metres have been spotted on helicopter surveys. One disadvantage of helicopter surveys is that in areas with a thick canopy, trees growing beneath may be missed. OISC's observations are that once a large tree has broken through the canopy it often matures very quickly, so areas designated as too steep for ground surveys within the ground buffer are flown every two years to compensate for the fact that trees will not be found until they are older.

Another 800 m from all mature trees is flown by helicopter to check for outliers. Despite the high cost of paying for helicopter time, the per-hectare cost is actually less than ground surveys because so much area can be covered quickly. Residential areas designated for helicopter surveys are done by ground or road in order not to disturb the residents. If a tree is found during an outlier survey, then an 800 m buffer is drawn around it and it becomes part of the area that is searched by ground.

Outreach to hikers, hunters and other outdoor recreationalists has been helpful in receiving reports of miconia. OISC engages organised groups of hikers and hunters through presentations, educational materials and social media with the aim of informing people how to identify and report miconia. We also present to schools and set up educational booths at community festivals in the areas where we are surveying. We believe that outreach also assists in gaining entry to private land. Our observation is that property owners who have heard about the invasive species problem before we call and ask their permission to survey, are more likely to let us on, although we have not specifically measured this.

RESULTS

OISC hired its first staff in November of 2001 and surveys started in 2002. The number of square kilometres surveyed per year has grown as more funding became available. Since 2002, OISC has been able to achieve a 96% reduction in mature plants from 4.8 mature trees per km^2 surveyed to 0.2 mature trees per km^2 surveyed (Table 1). There were 40 mature trees found and removed over 8.26 km^2 in 2002 and just 12 found and removed over 54.71 km^2 in 2016.

OISC also counts and takes GPS points for trees that are immature but over two metres tall. Trees over two metres that are missed will likely be mature the next time the field crew surveys. Therefore, the number of trees over two metres should also be at zero in order to achieve and ensure containment. OISC has achieved an 81% decrease in trees over two metres from 6.8 to 1.3 trees per km^2.

Three significant range extensions have recently occurred. As stated above, OISC's data shows that 99% of immature plants fall within 350 m of a mature plant. However, in 2015, one immature tree was found 6,900 m from the nearest mature tree. In 2016, another was found 1,600 m from the nearest mature tree. In 2017, a small patch of mature and immature plants was found 2,400 m from the nearest mature tree. The 6,900 m and 2,400 m extensions were found while the crew was surveying for other plant species.

DISCUSSION

Having the source trees removed from the botanical gardens and the few private properties, as well as detection of mature trees by agencies and volunteers before OISC was even formed in 2001, was a tremendous help to the eradication project. OISC was able to apply its strategy around the historical points and get a head start on delimitation. By 2010, the surveillance and delimitation phase of the project was complete. OISC did not have the resources to survey all suitable habitat, however, we took the steps described below to ensure that all known populations were mapped. We interviewed fellow natural resource agencies and hiker groups working in suitable habitats to ensure there was not a population in areas we did not have the time to look at. We conducted binocular surveys outside our survey areas in prime miconia habitat looking for large patches. We also calculated the distance of immature trees to the nearest mature tree in 2009. We found that 99% of trees fell within 400 m of a mature tree and maximum distance of an immature tree was just short of 1,600 m (Fujikawa, et al., 2009). This gave us confidence that by 2010, delimiting was complete.

After 2010, the project moved into the containment phase, but it has been difficult to achieve containment as defined by eliminating all mature plants. Although OISC has been able to achieve a significant decrease in the number of mature plants per square kilometre surveyed, we have not been able to completely suppress maturation.

Trees that are missed during one survey cycle are sometimes missed due to human error—thick vegetation and steep terrain are two factors that may decrease the efficacy of surveillance. Although our success rate with getting property owners to agree to let field crews survey their property is 95%, there are some property owners who have been reluctant to agree to surveys. In one case, it took

Table 1 Number of mature, >2m tall trees and km^2 surveyed 2002–2016.

Year	No. mature trees	No. trees >2m	Km^2 surveyed
2002	40	94	8.26
2003	4	21	9.37
2004	7	14	9.00
2005	9	54	21.00
2006	6	27	25.16
2007	6	25	29.37
2008	0	37	20.53
2009	4	89	14.07
2010	1	48	23.25
2011	3	27	27.00
2012	5	83	14.87
2013	2	94	22.20
2014	5	97	21.59
2015	12	123	39.44
2016	12	94	54.71

several years to acquire access from a property owner who owned an entire valley. By the time the crew was able to survey, trees had matured. Sometimes the 5% that say no or take a long time to say yes can be critical. In some years decreases in funding meant we did not have the resources to survey the area required by our strategy. The combination of funding fluctuations and time spent negotiating property access allowed some trees to mature. The presence of mature trees may have resulted in range extensions into new watersheds from long-distance dispersal events.

A review of the distances between immature and mature trees conducted in 2017 resulted in 99% of immature miconia falling within 350 m of a mature tree (Fujikawa pers. comm. 2017), which was similar to our 2009 results. However, the furthest immature miconia was now 6,900 m away from the nearest mature tree.

Miconia's long-lived seed bank is a complicating factor in achieving containment. Research by Meyer (2010) has estimated the seed bank at 16 years in French Polynesia, but observations on Oʻahu suggest it may be as long as 20 years. The Wahiawā botanical garden where miconia was originally introduced has found seedlings and reported them to OISC as late as 2016. They removed their mature tree in 1996 (Medeiros, et. al., 1997) and OISC has surveyed the entire area at least three times without finding any miconia whatsoever, so the likelihood that the seedlings are from the 1996 mature tree is very high.

Lessons learnt

The long-term work done on miconia in French Polynesia is key to OISC's success in preventing a full-scale miconia invasion on Oʻahu. The research was critical to raising the alarm about the species and mobilising control efforts early. Miconia is one of the few species that OISC has taken on where the seed-bank longevity is known thanks to the long-term studies done by Jean-Yves Meyer (Meyer, et al., 2011). Research from Tahiti also formed the basis of the outreach narrative. For example, one of the key talking points for outreach was the enormous area of forest that had been turned into monotypic stands of miconia and a photograph of a landslide in Tahiti was a mainstay of state-wide outreach to explain the potential erosion effects. Both Tahiti and Oʻahu offer lessons to islands where miconia might be dispersed in the future.

If possible, having adequate funds at the beginning of an eradication project to complete delimiting as soon as possible may shorten the containment phase. It took OISC eight years to be sure where the miconia population was. Private property was a complicating factor. A small percentage of larger landowners would not let the field crew survey in a timely manner and trees were allowed to mature while we negotiated access. Delimiting and containing a species before it spreads to additional private property owners will be immensely helpful in achieving eradication as quickly as possible. Taking on additional species can also be helpful in detecting long-distance dispersal events. OISC volunteered to do aerial surveys for a forest pathogen because it would require us to fly over all habitat suitable for miconia. The small patch of mature and immature trees 2,600 m away from the nearest known mature tree was discovered during this survey.

Outreach has also been helpful for the survey effort and programme managers should consider dedicating funds and employees for outreach from the beginning of the programme. OISC did not have a full-time outreach specialist until four years after the programme started. OISC has had trees reported to us but more importantly, we believe outreach has helped OISC get access to survey on private land. People seem more willing to grant access if they have heard of the invasive species problem and miconia before they receive the call asking for permission to survey.

OISC's outreach is a combination of talking directly to community groups, hiking and hunting groups and schools through presentations, participating at events and social media. We have not had the ability to scientifically test the outreach and see which methods or messages work best. Anecdotally, we believe that explaining the larger ecosystem effects and the possible effects on the island's water supply will persuade a wider group of people than those who might be motivated by preserving the native flora of Hawaiʻi. Research about which messages would be received best would be welcomed.

Although completely suppressing maturation of miconia has proved difficult due to the long-lived seed bank, OISC has been able to achieve a 96% reduction in mature trees per square kilometre since the programme began in 2002. In 2016, only 12 mature trees were found across 54.71 km^2. OISC has been able to keep the density of mature trees very low, but the long-lived seed bank means that a missed tree due to human error, an area that the field crew cannot survey due to lack of funding or the inability to get access to private property will likely result in a mature tree once the crew has access to the area. Containment for species with long-lived seed banks will be long-term projects.

For this reason, when evaluating feasibility, prioritising species for control and planning eradications, seed bank longevity should always be taken into account (Panetta and Timmins 2004). Policymakers deciding which species should be restricted for import should also consider seed bank longevity as a critical factor. While a long-lived seed bank is certainly not the only factor that makes a species invasive, if a species with a long-lived seedbank starts to become a problem, eradication will be a long-term and expensive project. Seed-bank longevity is not known for many species, and conducting longevity studies to answer that question for species under management would be very helpful to plan eradication efforts.

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REFERENCES

Board of Water Supply (2016). *Water Master Plan*. Honolulu, USA: Board of Water Supply. <www.boardofwatersupply.com/water-resources/water-master-plan>.

Chimera, C.G., Medeiros, A.C., Loope, L.L. and Hobdy, R.H. (2000). 'Status of management and control efforts for the invasive alien tree *Miconia calvescens* DC. (Melastomataceae) in Hana, East Maui'. *Pacific Cooperative Studies Unit Technical Report #128*. <https://scholarspace.manoa.hawaii.edu/handle/10125/7122>.

Fujikawa, J.Y., Iott, S., Luechauer, Z., Beebe, C., Kanakaokai, G., Conley, A., Arnott, C. and Neville, R. (2009). 'Delimiting surveys, control and eradication of *Miconia calvescens* using pre-determined buffer areas on the island of Oʻahu, Hawaiʻi'. *Poster presented at the 2009 International Miconia Conference,* Keanae, Maui, Hawaii, May 4–7, 2009.

Giambelluca, T.W., DeLay, J.K., Nullet, M.A., Scholl, M.A. and Gingerich, S.B. (2011). 'Canopy water balance of windward and leeward Hawaiian cloud forests on Haleakalā, Maui, Hawaiʻi'. *Hydrological Processes* 25: 438–447.

Giambelluca, T.W., Chen, Q., Frazier, A.G., Price, J.P., Chen, Y.-L., Chu, P.-S., Eischeid, J.K, and Departe, D.M. (2013). 'Online rainfall atlas of Hawaiʻi'. *Bulletin of the American Meteorological Society* 94: 313–316.

Koʻolau Mountain Watershed Partnership (KMWP) (2017). *Koʻolau Mountain Watershed Partnership Action Plan 2016–2017*. <http://koolauwatershed.org/wp-content/uploads/KMWP-2016-2017-Action-Plan.pdf>.

Libeau, M., Pouteau, R., Taputurai, R. and Meyer, J.-Y. (2019). 'Predicting the risk of plant invasion on islands: the case of *Miconia calvescens* in the Marquesas, French Polynesia (South Pacific)'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* p. 716. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Medeiros, A.C., Loope, L.L., Conant, P. and McElvaney, S. (1997). 'Status, ecology, and management of the invasive plant *Miconia calvescens* DC. (Melastomataceae) in the Hawaiian Islands'. *Records of the Hawaii Biological Survey for 1996. Bishop Museum Occasional Papers* 48: 28–36.

Meyer, J.-Y. (1996). 'Status of *Miconia calvescens* (Melastomataceae), a dominant invasive tree in the Society Islands (French Polynesia)'. *Pacific Science* 50: 66–76.

Meyer, J.Y. (2004). 'Threat of invasive alien plants to native flora and forest vegetation of Eastern Polynesia'. *Pacific Science* 58: 355–375.

Meyer, J.-Y. (2010). 'The *Miconia* Saga: 20 Years of Study and Control in French Polynesia (1998–2008)'. In: L.L Loope, J.-Y. Meyer, B.D. Hardesty and C.W. Smith (eds.) *Proceedings of the International Miconia Conference, Keʻanae, Maui, Hawaiʻi*, May 4–7, 2009, Maui Invasive Species Committee and Pacific Cooperative Studies Unit, University of Hawaiʻi at Mānoa. <http://www.hear.org/conferences/miconia2009/proceedings/>.

Meyer, J-Y., Loope, L. and Gourant, A-C. (2011). 'Strategy to control the invasive alien tree *Miconia calvescens* in Pacific islands: eradication, containment or something else?' In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 91–96. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Nanko, K., Giambelluca, T.W., Sutherland, R.A., Mudd, R.G., Nullet, M.A. and Ziegler, A.D. (2013). 'Erosion potential under *Miconia calvescens* stands on the island of Hawaiʻi'. *Land Degradation & Development* 26: 218–226.

Panetta, F.D. and Timmins, S.M. (2004). 'Evaluating the feasibility of eradication for terrestrial weed incursions'. *Plant Protection Quarterly* 19: 5-11.

Panetta, F.D. and Lawes, R. (2005). 'Evaluation of weed eradication programs: the delimitation of extent'. *Diversity and Distributions* 11: 45–442.

Panetta, F.D. (2007). 'Evaluation of weed eradication programs: containment and extirpation'. *Diversity and Distributions* 13: 33–41.

Takahashi, M., Giambelluca, T.W., Mudd, R.G., DeLay, J.K., Nullet, M.A. and Asner, G.P. (2011). 'Rainfall partitioning and cloud water interception in native forest and invaded forest in Hawaiʻi Volcanoes National Park'. *Hydrological Processes* 25: 448–464.

Weed eradication on Raoul Island, Kermadec Islands, New Zealand: progress and prognosis

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Abstract During the 45 years that the Raoul Island weed eradication programme has been underway, eleven species have been eradicated. To complete the restoration of Raoul Island's unique ecosystems supporting significant seabird biodiversity and endemic biota, nine further transformer weeds must be eradicated. In this review of progress to date, we examine the feasibility of eradication of these transformers and identify that four species are on target for eradication: African olive (*Olea europaea* subsp. *cuspidata*), yellow guava (*Psidium guajava*), castor oil plant (*Ricinus communis*) and grape (*Vitis vinifera*). However, for four more species more staff resources are required to achieve eradication as currently infestations are establishing faster than they are being eliminated: purple guava (*Psidium cattleianum*), black passionfruit (*Passiflora edulis*), Brazilian buttercup (*Senna septemtrionalis*) and Mysore thorn (*Caesalpinia decapetala*). The ninth species, Madeira vine (*Anredera cordifolia*), is being contained but presents logistical difficulties for effective control – herbicide resistant tubers and cliff locations requiring rope access in unstable terrain. Increasing the resources for this programme now to enable eradication of these transformer weeds will reduce the total long-term cost of the programme. Eradication of rats, the 2006 eruption, recent greater cyclone frequency, increased tourism requiring biosecurity management, and staffing reductions have all impacted progress on weed eradication. Myrtle rust (*Austropuccinia psidii*), confirmed in March 2017 as the latest invasive species on Raoul Island, is establishing on Kermadec pohutukawa (*Metrosideros kermadecensis*), the dominant canopy species. The impact of this species on the weed eradication programme is unknown at this point.

Keywords: conservation, dispersal, eradication feasibility, invasive species, rats, seabirds, seedbank, transformer

INTRODUCTION

Raoul Island, the largest and northernmost of the main islands in the Kermadec Group (29° 15' S, 177° 55' W), was once home to vast seabird colonies. However, the impacts of whalers and settlers from 1800 AD through the introduction of goats (*Capra hircus*), pigs (*Sus scrofa*), cats (*Felis catus*), and Norway rats (*Rattus norvegicus*) extirpated most indigenous seabird species and a number of indigenous land birds (Veitch, et al., 2004). The goats had a major impact on the endemic vascular plants too (Parkes, 1984). Many vascular plant species were introduced for food and animal forage (Sykes et al., 2000). Twenty-five vascular plant species are endemic to Raoul Island (Sykes, et al., 2000), and most of these make up the forest that clothes the island, dominated by Kermadec pohutukawa (*Metrosideros kermadecensis*). The latest invasive species to arrive on Raoul Island is myrtle rust (*Austropuccinia psidii*). This species was first detected in March 2017 and noticed because of canopy die-off of a small area of mature Kermadec pohutukawa in Denham Bay. Myrtle rust has the potential to alter the dynamics of many native and introduced biota on the island by releasing plants from suppression by the pohutukawa canopy and reduced flowering and nectar production which will impact some land birds. Raoul Island is an active volcano, last erupting in 2006, and it is located in the path of seasonal cyclones (December to May).

Because of its unique ecosystems, Raoul Island was declared a Flora and Fauna Reserve (now Nature Reserve) in 1934. The New Zealand government has funded the eradication of all introduced feral mammals: goats were eradicated in 1984 (Sykes & West, 1996), and rats and cats were eradicated in 2002 and 2004, respectively (Broome, 2009). The eradication of goats greatly assisted recovery of endemic plant species, rescuing several from the brink of extinction. As a consequence of the rat and cat eradications indigenous seabirds and land birds are returning to Raoul Island (Veitch, et al., 2011), significantly beginning the recovery of this ecosystem. Several terrestrial birds now occupy extensive areas of Raoul and are likely to have significant impacts on ecosystem dynamics.

However, a small suite of transformer weed species (sensu Pyšek, et al., 2004) currently impedes full restoration of ecosystem functioning on Raoul Island. The vascular plant flora of Raoul Island currently comprises 118 indigenous species and 196 introduced species (of which c.10% are transformer species). A weed eradication programme has been underway since 1972 (West, 2011) and, to date, 11 species have been eradicated (Table 1), the majority of which were transformers (West & Thompson, 2013). New incursions or detection of exotic species are evaluated for impact and eradication potential as per DOC weed-led systems (Owen, 1998). Biosecurity to prevent new incursions is a priority and weed control to protect threatened plant species in non-forested, coastal ecosystems is important.

The eradication programme is now focussed on nine transformer species that have a major impact on forest ecosystems, four of which are vines (Table 2, and see West, 1996 for more background on these species). Given that seabirds are now beginning to return to Raoul Island to breed, it is particularly important to ensure that vines, which can entangle landing seabirds as well as smother native vegetation including forest, are eliminated.

Weed eradication programmes can take a long time, and many have failed (Panetta, 2015). The Raoul Island weed eradication programme has been formally reviewed twice since it began 45 years ago (West, 1996; West & Havell, 2013). Each time, the species being targeted have been evaluated to understand impacts of the species, and effectiveness of the eradication methodology. Both reviews have resulted in changes to the management programme and revised lists of species to focus on for eradication. The latest review restricted the focus to nine species where eradication will have the biggest impact on biodiversity. Changes to staffing were also recommended, so that there would be six months overlap of some experienced staff with new staff. This recommendation has been actioned for the contracted staff but the volunteer programme (six-month term) was discontinued in 2015 and replaced with

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seconded staff (three-month term), effectively halving this additional effort for weed control.

In between the two formal reviews, the programme is constantly evaluated in relation to all management on the island. For example, grape (*Vitis vinifera*) was added to the list of target transformer species before rats were eradicated because the two rat species present were preventing fruit development on the grape vines (West and Havell, 2011). The year in which eradication commenced for each of the nine transformer species is shown in Table 3.

The option of eradication of transformer species is more appealing in the long-term than ongoing control to zero-density, as eradication means that financial investment in weed detection and control can cease once the species have been eliminated. To achieve this, sufficient resourcing is required to not only achieve the goal but also reduce total costs (Panetta, 2015). The feasibility of eradication of alien plants from Raoul Island was evaluated 30 years after the programme began (West, 2002). At that time, all necessary conditions (listed in West, 2002) appeared to be met, and application to the task was what was needed.

Preventing reinvasion is entirely achievable for all nine remaining target species, given the remoteness of Raoul Island and the strict biosecurity protocols that are in place. But how well are the species being extirpated and contained within Raoul Island as eradication proceeds? Panetta (2015) describes a model for categorising species in terms of the 'technical' feasibility of eradication by taking into consideration the relative feasibilities of extirpation and containment. He notes that eradication occurs via two processes: (i) extirpation (the elimination of the target in both space and time) and (ii) containment, which is the prevention of further occupancy of space (i.e. spread).

This approach is a useful one to apply to the nine target transformer species on Raoul Island as the work is done and reported on a plot basis, and it is an advancement on the methodology proposed by Holloran (2006) for reporting progress. There are currently 13 weeding blocks comprising 153 plots of varying size (0.1–83.2 ha), covering almost 834 ha which is 28.3% of the total area of Raoul Island (Fig. 1). Plot size varies based on terrain and travelling time; typically each plot can be carefully grid-searched in one day (see West, 2002 for more detail

Table 1 Species eradicated from Raoul Island. For each species, the year eradication began and the year in which the species was last recorded are given. Eradication was formally declared in 2013 by West and Thompson (2013).

Species	Common name	Eradication began	Last recorded
Cortaderia selloana	pampas grass	1984	1993
Ficus macrophylla	Moreton Bay fig	1996	1999
Foeniculum vulgare	fennel	1969	1999
Furcraea foetida	Mauritius hemp	1974	2002
Gomphocarpus fruticosus	swan plant	1979	2002
*Macadamia tetraphylla**	macadamia	1996	2003 (2015)
Phoenix dactylifera†	date	1995	1999
Phyllostachys aurea	bamboo	1996	2001
Populus nigra	poplar	1995	2003
Senecio jacobaea	ragwort	1980	1980
Vitex lucens	puriri	1997	1997

* One macadamia seedling was found in 2015 in the same location as the original small stand of trees.
† Wild dates have been eradicated but the species is still present at two historic sites as apparently non-reproductive individuals.

Table 2 Transformer weeds currently being eradicated on Raoul Island. The juvenile period, seed persistence and dispersal mechanism of each species is used to estimate the feasibility of eradication (Panetta 2015). Species are listed in order of feasibility of eradication: most to least.

Species	Common name	Growth form	Juvenile period	Seed persistence	Dispersal	Feas-ibility	Goal
Olea europaea subsp. *cuspidata*	African olive	Tree	5 years	2.4 years[1]	bird	3	eradicate
Psidium cattleianum	purple guava	Small tree	2–3 years	6–7 months[2]	bird	3	eradicate
P. guajava	yellow guava	Shrub	1–2 years	c. 1 year[3]	bird	4	eradicate
Passiflora edulis	black passionfruit	Vine	9 months	a few weeks[4]	bird	4	eradicate
Ricinus communis	castor oil plant	Small tree	5–6 months	>19 years[5]	explosive*	6	eradicate
Senna septemtrionalis	Brazilian buttercup	Shrub	c. 2 years	>16 years[6]	explosive*	5 or 7	eradicate
Caesalpinia decapetala	Mysore thorn	Vine	4–6 months	>12 years[7]	explosive*	6	eradicate
Anredera cordifolia	Madeira vine	Vine	< 1 year	15 years (tubers)[8]	gravity*	6 or 8	contain
Vitis vinifera	grape	Vine	1 year	5 years[9]	bird	8	eradicate

*Occasional long-distance dispersal by wind, bird, water or accidental-human vectors: for Brazilian buttercup and Madeira vine this occasional longer distance dispersal has resulted in considerable range extension, therefore, two feasibility estimates are given to cover both the normal and not uncommon dispersal events. [1]Cuneo, et al., 2010; [2]Uowolo & Denslow, 2008; [3]CABI, 2017b; [4]CABI, 2017a; [5]Kammili & Jatothu, 2015; [6]Ewart, 1908; [7]no published data: this estimate is from an isolated infestation of known age on Raoul Island; [8]Harden, et al., 2004; [9]no published data found for *Vitis vinifera*: this estimate is for *Vitis aestivalis* (Haywood, 1994).

of the plot-based searching methodology). Using Panetta's model (Panetta, 2015), each plot can be evaluated to see if the species has been extirpated from it, and the distribution of a species among the plots can be evaluated to see if the species is being contained or is expanding its range. Then, the relative relationship between extirpation and containment can be evaluated for each target species to determine if eradication can be achieved (Panetta, 2015).

METHODS

On-island weed searching

Details of weed searching and removal are given in West (2002) and Holloran (2006) and here we restate briefly what the annual plan and actions are: that weeding plots should be grid-searched on the ground a minimum of once each year with plots containing the target transformer species to be searched twice. Within plots, known infestations are marked (including GPS coordinates) and specifically searched during grid-searching or between grid-searches. Finds of immature plants outside of infestations are recorded as random finds. New infestations are created when mature, fruiting plants or localised seed banks are found and, if a new infestation is large, a new plot may be created. If no target species have been found at an infestation for the period of the suspected viability of the propagule bank based upon database records, or the site has been destroyed by a landslide or volcanic eruption, the infestation is retired. GPS tracks of grid-searching are downloaded to Arc-GIS and used to identify any gaps in search coverage and to document search effort.

Feasibility of eradication

Data on the factors used by Panetta (2015) to determine feasibility of eradication were compiled from published information and, where necessary or more appropriate, from our observations on Raoul Island. The two key biological factors relevant to extirpation are the length of the juvenile period (i.e. how quickly can plants produce more viable propagules?) and seed persistence (i.e. how long can seed remain viable?). The biological factor that is most relevant to containment is dispersal modes (i.e. is spread likely to be short- or long-distance; predictable – e.g. water or wind – or unpredictable?). Evaluating the data for these three factors enables identification of eradication feasibility on a scale from most feasible (a score of 1) to least feasible (a score of 8 – see Panetta, 2015, p. 232).

Evaluation of progress towards eradication

All data on the number of individuals removed per plot for the nine target weed species from 1 January 1998 to 31 December 2016 were extracted from the Raoul Island weed database. Recording of the number of individuals in three stage classes – seedlings, adolescents (taller than 30 cm but not yet flowering) and mature (flowering and/or fruiting) – began in October 1997, so the database now holds more than 20 years of continuous data.

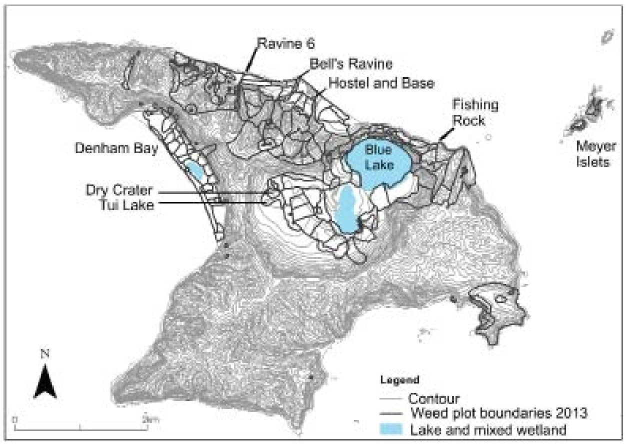

Fig. 1 Raoul Island showing places mentioned and the distribution of weed plots.

The number of times each plot was searched per year, from 1998 to 2016 was extracted from the database. Also, the number of active, retired and random infestations in each plot was summarised for the same period. This information was used to interpret the data on number of individuals per target species per plot and per year.

RESULTS

On-island weed searching

From 1998 to 2011 the mean number of grid searches per plot exceeded one a year and exceeded two in 2003 (Fig. 2). However, since 2012 the number of plots grid-searched has dropped well below a single search each year culminating in less than one third of plots being grid-searched in 2016.

Feasibility of eradication

The data for the nine transformer species targeted for eradication show a wide range of feasibility (Table 2), from feasible, e.g. African olive (*Olea europaea* subsp. *cuspidata*) and purple guava (*Psidium cattleianum*) to much less feasible e.g. Madeira vine (*Anredera cordifolia*) and grape. Species with long juvenile phase (> 2 years), short seed persistence (< 3 years) and short distance or largely human-mediated propagule dispersal score lower, and are therefore more feasible to eradicate. Conversely,

Table 3 Percentage of plots occupied by each species in 1997–2000 (from West 2002) and 1998–2016 as well as the year in which eradication began.

Species	Common name	Eradication began	% plot occupancy	
			1997–2000	1998–2016
Olea europaea subsp. *cuspidata*	African olive	1973	19.6	13.6
Psidium cattleianum	purple guava	1973	22.4	25.0
P. guajava	yellow guava	1972	11.9	8.6
Passiflora edulis	black passionfruit	1980	32.9	36.4
Ricinus communis	castor oil plant	1990	7.7	6.4
Senna septemtrionalis	Brazilian buttercup	1978	72.0	72.1
Caesalpinia decapetala	Mysore thorn	1974	18.2	20
Anredera cordifolia	Madeira vine	1995	2.1	2.1
Vitis vinifera	grape	1998	8.4	8.6

species with short juvenile phase (< 2 years), long seed persistence (> 3 years) and long distance dispersal score higher. The short juvenile phase means searching has to be more frequent; the long seed persistence means the duration of the programme is longer and is extended every time new seed is added to the soil if a fruiting individual is not found in time; the long-distance dispersal means that a greater area must be searched.

Evaluation of progress towards eradication

All transformer species

The number of active and retired infestations for each species gives a good indication of progress towards achieving eradication (Fig. 3). Five species – African olive, yellow guava (*Psidium guajava*), castor oil plant (*Ricinus communis*), purple guava and black passionfruit (*Passiflora edulis*) – have considerably more retired than active plots. Brazilian buttercup (*Senna septemtrionalis*) and Mysore thorn (*Caesalpinia decapetala*) – the two species with the greatest seed longevity – have proportionally more active plots. Control of grape began later than the other species (Table 3), hence the high proportion of active plots compared to retired. Madeira vine is the only species with more active than retired plots.

The random infestations give an indication of dispersal beyond the immediate vicinity of mature plants (Fig. 3) and the effectiveness of the programme to control weed reproduction. Black passionfruit and purple guava, both bird-dispersed, have a relatively high proportion of random finds. Mysore thorn has the highest proportion of random finds reflecting not its dispersal ability but its highly cryptic nature when growing among the tall ground ferns which grow densely in parts of Denham Bay, and its extensive original distribution.

African olive

Numbers of this species detected and removed since 1998 are very low (Fig. 4). The last mature individual was removed in 2008, and two adolescent plants from different locations in 2010 and 2011. All of these finds were from within the historic range of this species before eradication commenced and the percentage of active plots for this species has decreased (Table 3).

Purple guava

Numbers of this species were low (West 2002) but began to increase in 2008, increasing an order of magnitude in 2011, and with very high numbers recorded in 2016 (Fig. 4) from just a few infestation plots mostly within the crater around the shores of Blue Lake and Tui Lake. Most of the purple guava detected in 2015 were from new detections in the dry crater near Tui Lake, adjacent to old infestations where purple guava was last detected in 2002. More than half the numbers detected in 2016 were from this infestation. Purple guava infestations are still being found within the known historical range of the species, but the percentage of plots occupied has increased slightly (Table 3). Infestations of purple guava and buffers of up to 100 metres have been intensively searched since 2015, and, subject to resourcing, additional areas within the crater are likely to be checked. Seedlings of this species are very cryptic (look very similar to two of the endemic species) so careful searching is required.

Yellow guava

There is an increased number of yellow guava “seedling” detections since 2011 but overall the numbers are quite low (Fig. 4). The last mature individual detected was in 2008 with no further detections at that site. The seedlings recorded are generally suckers from roots: those recorded in 2011 were suckers from just two plants. A yellow guava shoot was discovered in 2015 in a crack in a concrete path close to the Hostel. This may have originated from root suckers from a relic guava root system in adjacent gardens, as ongoing persistence and lack of other finds indicates. However, it is also possible that an undetected mature plant may be present within the range of local birds. Yellow guava is active in fewer plots within its historical range (Table 3).

Black passionfruit

The number of black passionfruit being detected and removed began to increase markedly from 2004 (Fig. 4), with the biggest number found so far, in 2011, coming primarily from one infestation where eight mature vines had been removed the year before. This site is still very active. To date, despite the increase in numbers detected, black passionfruit has not materially exceeded its historic range. Although it now occupies more plots than previously (Table 3), these are plots within the bounding polygon that describes the historic range of this species.

Castor oil plant

Numbers of this species are low (Fig. 4), with the last mature plants removed in 2003 all from one site. With the exception of three adolescent plants in 2011 and 2012, all other seedlings and adolescents removed since 2004 have come from this site. Castor oil plant has not expanded beyond its historic range and has fewer active plots than previously (Table 3).

Brazilian buttercup

This species has been the most numerous since the eradication programme began. Numbers were declining

Fig. 2 Mean number of grid searches per plot from 1998–2016.

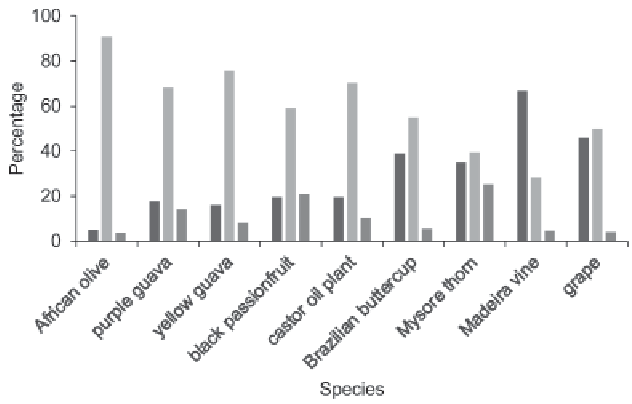

Fig. 3 Percentage of active (black bars), retired (mid grey bars) and random (dark grey bars) infestations within the weeding plots for each transformer species from 1998–2016.

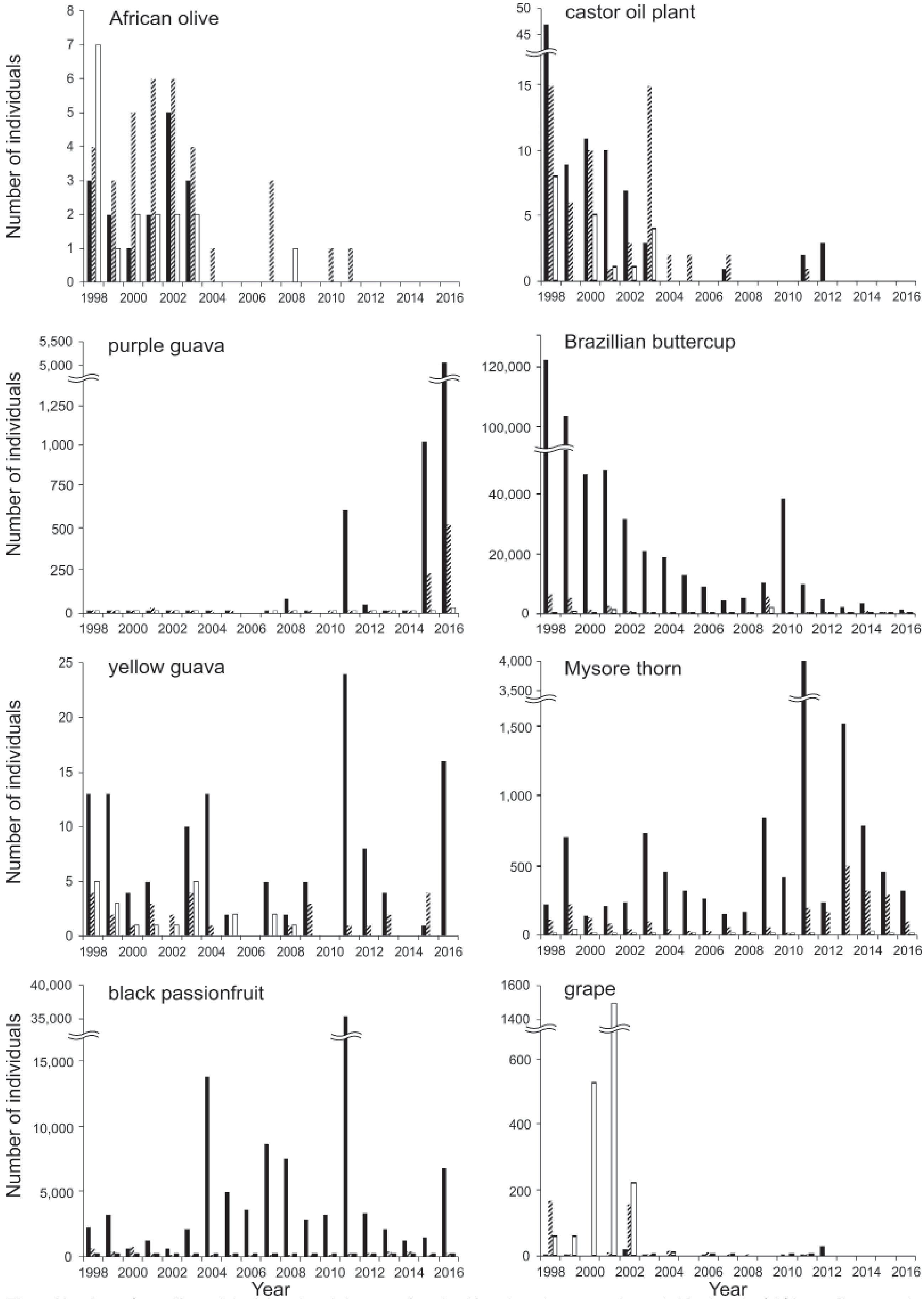

Fig. 4 Number of seedlings (black bars), adolescent (hatched bars) and mature plants (white bars) of African olive, purple guava, yellow guava, black passionfruit, castor oil plant, Brazillian buttercup, Mysore thorn and grape removed in the period 1998–2016 from Raoul Island. For Brazilian buttercup the data for the same period are for removal from Raoul Island and the Meyer Islets: this is the only one of the target species found on these islets that are c. 1 km NE of Raoul Island.

effectively until 2008 when they began to increase, reaching a new peak in 2010 – primarily seedlings (Fig. 4). In 2009, the significant increase in mature and adolescent plants is due to the discovery of three outlier populations that extended the range of this species beyond its historic range. Two of these sites – the westernmost weed plot (see Fig. 1) and a plot on the cliffs at the southern end of Denham Bay were found during helicopter surveillance. The other site, below bluffs at the northern end of Denham Bay was found during a routine search of a nearby plot at the back of the bay. Brazilian buttercup is also on North and South Meyer Islets and is the only one of the target species found off Raoul Island. This species occupies a marginally greater percentage of plots than previously (Table 3): the new plots described above have been virtually cancelled out by the retirement of some plots due to slips and the 2006 eruption.

Mysore thorn

Mysore thorn numbers have fluctuated through time but reached a new peak in 2011 as a result of the high number of mature vines found in 2010 (Fig. 4): the highest number of mature plants recorded since 2001 (Fig. 5). Mysore thorn is confined to Denham Bay now that an infestation at the head of Ravine 6 has been eradicated. Within Denham Bay, however, this species' range has increased slightly, with helicopter surveillance in 2002 and 2009 leading to the detection of two sites on the cliffs above the bay, including the southernmost site known (see Fig. 1). However, it is not these newly discovered infestations that are contributing the higher numbers of all size classes since 2010, it is a number of the historic plots on the flat and towards the cliffs north of Denham Bay swamp. Mysore thorn occupies a slightly higher percentage of plots than previously (Table 3).

Madeira vine

The weight of tubers removed in 1998 was not recorded but a file note halfway through that year mentions 60 sacks of tubers had been removed. The amount of tubers removed is overall less in the past decade than in the previous one (Fig. 6) as the more accessible plots are controlled. Various methods for killing the tubers have been trialled and used, including composting (in black bins using an accelerant), burning, desiccation followed by burning of the desiccated tubers, and freezing (the current method). Madeira vine has not expanded beyond its historic range of two locations and has the same percentage plot occupancy as previously (Table 3). Madeira vine has almost been eradicated from Bell's Ravine with only small finds in 2015.

Grape

Grape vines are now in very low numbers, with no new sites since 2012 (Fig. 4). Most of the infestation sites are in Denham Bay and three are in old settlement areas on the north side of Raoul Island, reflecting past human occupancy. The percentage plot occupancy for grape has increased very slightly (Table 3), reflecting a single mature vine found in 2011 during grid-searching in Denham Bay.

DISCUSSION

Panetta's model (Panetta, 2015) is a very useful framework to evaluate eradication feasibility but when using it, we have been very conscious of the lack of accurate data on seed longevity in the soil in Raoul Island's environmental conditions. We have observed that some of the transformer species being targeted on Raoul Island have seedling banks, e.g. African olive and black passionfruit, and others have the ability to resprout from underground roots, e.g. yellow guava and grape. It could be useful for these mechanisms to be added to the model, perhaps as propagule persistence (replacing seed persistence) given that resprouts can appear more than three years after any other stem material has been present above-ground, and seedlings can remain in a seedling bank for more than three years until a light gap is created allowing the seedlings to rapidly grow to into mature plants.

The graphs of species abundance through time (Figs 4 & 5) combined with life history data indicating feasibility of eradication (Table 2) as well as plot occupancy (Table 3) and the number of active, retired and random infestations (Fig. 3) indicate that eradication is very achievable for four species: African olive, yellow guava, castor oil plant and grape. Note that species differ in their life history traits so therefore have different eradication feasibility scores (Table 2). Grape has the highest score (least feasible) but is eradicable because the biomass of all grapes was reduced to essentially zero before the rats were eradicated (West & Havell, 2011). Dispersal of this species has not been possible because all resprouts are found and destroyed before fruits are formed.

With no detections of African olive since 2011 and estimated seed persistence of 2.4 years, it is theoretically possible to declare this species eradicated now. However, given the cryptic nature of this species and seedling persistence, our preference is to wait until at least 2021 before making this claim (if there are no further detections). Yellow guava persists as occasional suckers in just three accessible locations, presumably from relict root systems, so should be eradicable with annual checks of the locations although the timeframe is difficult to estimate. Castor

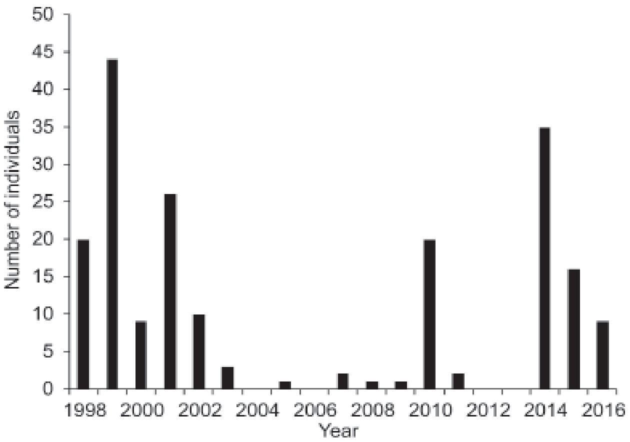

Fig. 5 Number of mature plants of Mysore thorn removed in the period 1998–2016 from Denham Bay, Raoul Island.

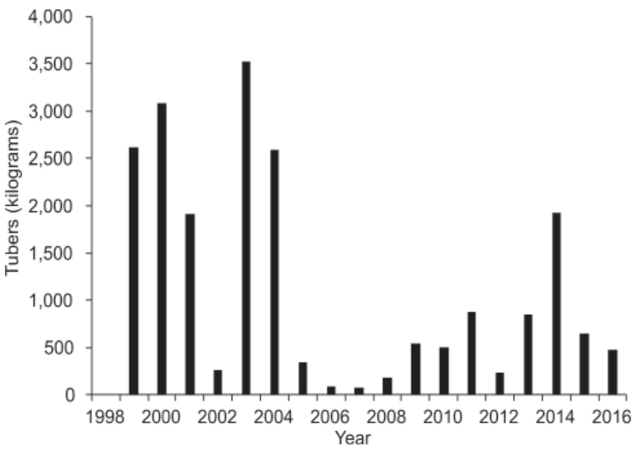

Fig. 6 Quantity (kg) of Madeira vine tubers (aerial and ground) removed in the period 1998–2016 from Raoul Island.

oil plant has seed longevity of >19 years so with the last detection of this species in 2012, and assuming no further detections, it could be deemed to have been eradicated by 2040. All grapes detected on Raoul Island so far have been resprouts from persistent root systems. Given the low feasibility of eradicating this species if it were reproducing by seed (Table 2) it is vital that fruit are never produced. All plots containing grape must be searched a minimum of once a year, ideally twice, given the short juvenile period possible for resprouts.

The feasibility of eradication for purple guava and black passionfruit is relatively high (Table 2), however, both species have been recorded in highest numbers in recent years (Fig. 4) and have a relatively high proportion of random finds (for black passionfruit there have been more random finds than there are active plots). The trend for both species indicates that the current search effort (Fig. 2) of less than one plot search per year, and the current area searched, is not sufficient to prevent seed dispersal. Black passionfruit is known to produce fruit within one year on Raoul Island, and the development of mature plants and purple guava seedlings may have occurred in the dry crater in the three years between 2012 and 2015. At the moment, both species have still been found within their historic range but, since they are bird dispersed, it is quite possible that these two species could be spreading to areas outside the current extent of searched plots (Fig. 1). All of these results indicate that grid searching for both purple guava and black passionfruit must continue until suppression of reproduction in these species is clearly demonstrated. Searching needs to be undertaken within the known range of these species and be extended into surrounding areas to detect any new infestations from bird-dispersed seeds. Uowolo and Denslow (2008) suggest the most effective time for purple guava control is at least three months after the fruiting season when the majority of seeds have germinated or died, given the short seed persistence. Another risk is the potential of large purple guava plants to sucker and reproduce rapidly after long periods of quiescence. A weed detection dog is being trained to focus on black passionfruit, both guava species and grape but is not yet ready for deployment.

Brazilian buttercup is the most widespread of the target species (Table 3). Range extensions of this species, discovered in 2009, plus its occurrence on the Meyer Islets indicates this species has rare long-distance dispersal, possibly by birds (although human dispersal can't be ruled out). In order for this species to be eradicated, detection methods need strong focus (Holloran, 2006). Any opportunities for helicopter surveillance should be taken, particularly when these coincide with the flowering period.

Although Mysore thorn is confined to Denham Bay, too many individuals are being missed in plot searches allowing considerable seed set e.g. the very high numbers of seedlings recorded in 2011 (Fig. 4). Seed germination in 2011 could also have been aided by two cyclones that affected Raoul Island in February (Atu) and March (Bune), with the latter resulting in widespread treefalls and stripping of foliage from trees. Given the rapid growth rate of Mysore thorn (Table 2) and the high proportion of random finds, the plots in Denham Bay need to be searched a minimum of twice each year and possibly with a closer spacing between observers than in the past (we suggest a minimum of 2 m). The short juvenile period plus the long seed persistence time make this species less feasible for eradication. However, because long-distance dispersal is very rare, this species is eradicable if seed banks and fruiting can be eliminated (as demonstrated by the eradication in Ravine 6). Of all the transformer species, Mysore thorn poses the greatest threat to ecosystem recovery, as shown by historic photographs and reports of Mysore thorn smothering the Kermadec pohutukawa canopy in Denham Bay (West, 1996). Landing seabirds can get entangled in vegetation: vines provide greater opportunities for entanglement and thorny vines (like Mysore thorn) less opportunity for safe escape (Arcilla, et al., 2015).

Madeira vine is the most difficult species to control on Raoul Island (West, 2002). It was last detected in its original location in Bell's Ravine in 2015 but because it can grow from tiny aerial tubers and subterranean tubers, it may still occasionally resprout in that location. However, at the main location east of Fishing Rock, this species grows on steep, unstable cliffs so tubers can be removed only from the most accessible sites and places that can be reached by abseiling safely. Herbicide is still used to knock back foliage when necessary to gain access to the herbicide-resistant tubers so they can be removed. Until a control method is developed that can kill tubers on the inaccessible cliffs, this species can only be contained rather than eradicated. Management so far has successfully contained Madeira vine.

It has been stated frequently in the literature that eradications are unlikely to succeed if the area occupied is large (Panetta, 2015). Howell (2012) identified that the only successful eradications of environmental weeds in New Zealand were those where the initial extent was < 1 ha, noting that there were other eradications of similar extent that were unsuccessful. On Raoul Island, four species currently have distributions of < 1 ha: African olive, yellow guava, castor oil plant and grape although the area to be grid-searched in the plots within which they occur ranges from 6 ha (castor oil plant) to 80 ha (grape). The area to be searched for the more abundant species – purple guava, black passionfruit, Brazilian buttercup and Mysore thorn ranges from 60 ha (Mysore thorn) to 550 ha (Brazilian buttercup).

However, Panetta's (2015) model of extirpation and containment indicates that African olive, yellow guava, castor oil plant and grape are all currently being extirpated (the rate of extirpation of managed infestations exceeds the rate of establishment of new ones) and contained, and could be eradicated with the current level of resourcing. As indicated above, breakthroughs in methodology are needed for extirpation of Madeira vine to become a reality. For the other four species, there have been more new infestations leading to greater numbers and, for Brazilian buttercup, range extension in recent years. The current level of resourcing is not sufficient to enable eradication as, based upon GPS track logs, not all known and potential areas are able to be searched within the time it takes for each species to fruit. There is also insufficient resourcing to analyse records of infestation within the Raoul Island weed database, and therefore plan the work more effectively.

There are many factors that have led to this situation. Rats (*Rattus norvegicus* and *R. exulans*) are no longer eating flowers, seeds and seedlings of plants. This effect can be seen in the results for several species, e.g. Mysore thorn, black passionfruit and yellow guava. No access to the crater was permitted for two years after the eruption in 2006 resulting in mature plants of purple guava and Brazilian buttercup, with dispersal of the former and seed added to the seed bank for the latter. Cyclones have been frequent in the past decade, resulting in large areas of windfallen trees within weed control sites that stimulates germination (good for reducing the seed bank) but also impedes access and slows down the rate of grid-searching, making weed removal harder. A formal process for retiring plots is not in place and this may have resulted in a lack of focus on areas that need more regular checking.

Health and safety requirements and biosecurity management are the factors that have most influenced the drop in the mean number of plots grid-searched (Fig. 2). Staff effort has been directed towards biosecurity management as the number of visitors to the island (including organised tour groups) has increased over the years. Health and safety requirements have increased considerably since 2006: permission must be sought to enter the crater to search the plots (the granting authority is the Operations Manager based on advice from the Institute of Geological and Nuclear Sciences which monitors seismic activity on the island); weed plots that must not be searched after heavy rain or during seismically active periods and those that require climbing gear have been identified. Staff numbers have been reduced since 2015 when deployment of volunteers for six-month periods was replaced by seconded staff for three-month periods.

These factors, combined with the lack of assistance from rats, reduced staff resourcing, plus the increased number of cyclones, has led to the increase in weed abundance, particularly for purple guava, black passionfruit, Brazilian buttercup and Mysore thorn. Health and safety standards should never be compromised but need to be compensated for by increased resourcing for the eradication of transformer weeds to be successful. The current budget for the programme is $555,000, reduced from $566,000 in the previous year. This needs to be increased to $850,000 to provide sufficient staff time to check all plots once a year plus an additional check of all plots with known Mysore thorn, grape and black passionfruit infestations (two checks a year, minimum). Madeira vine plots should be checked at the current rate of one day per week. As Panetta (2015) states "Despite the best of intentions and the highest level of professionalism, an eradication effort will not succeed if it is not adequately resourced". Increasing the budget is a wise move given that Cacho, et al., (2007) showed that total cost of weed eradication is high when low search effort was involved, but falls rapidly with increasing search effort because a more intense search effort would reduce the number of reproductive plants. This is currently the sticking point for four of the nine transformer species in the Raoul Island weed eradication programme. Whereas the bulk of the budget should be spent on Raoul Island (staff, infrastructure, materials), the off-island support resourcing is also vital and must be set at the optimal level. Remote island-based programmes require huge logistical support from dedicated teams both on and off the island.

The preceding discussion describes a number of factors that need to be considered when budgeting for success in this programme and the results described are all influenced by these. However, myrtle rust, whose impact is yet to be felt by Raoul Island ecosystems, is the latest agent of change that will need to be considered. Monitoring plots have been established to provide early indications of how this disease will affect Kermadec pohutukawa and what the flow-on effects will be. It is possible that the dynamics of the transformer weeds in this eradication programme will change, just as they have following eradication of rats and cyclones.

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REFERENCES

Arcilla, N., Chang-Yong, C., Kiyoaki, O. and Lepczyk C.A. (2015). 'Invasive species and Pacific island bird conservation: a selective review of recent research featuring case studies of Swinhoe's storm petrel and the Okinawa and Guam rail'. *Journal of Ornithology* 156 (Suppl. 1): S199–207.

Broome, K.G. (2009). 'Beyond Kapiti – a decade of invasive rodent eradications from New Zealand islands'. *Biodiversity* 10: 7–17.

Cacho, O.J., Hester, S.M. and Spring, D. (2007). 'Applying search theory to determine the feasibility of eradicating an invasive population in natural environments'. *Australian Journal of Agricultural and Resource Economics* 51: 425– 433.

Cuneo, P., Offord, C.A. and Leishman, M.R. (2010). 'Seed ecology of the invasive woody plant African olive (*Olea europaea* subsp. *cuspidata*): implications for management and restoration'. *Australian Journal of Botany* 52: 342–348.

Ewart, E.J. (1908). 'Vitality of seeds'. *Proceedings of the Royal Society of Victoria* 2: 1–203.

Harden, G.J., Fox, M.D. and Fox, B.J. (2004). 'Monitoring and assessment of restoration of a rainforest remnant at Wingham Brush, NSW'. *Austral Ecology* 29: 489–507.

Haywood, J.D. (1994). 'Seed viability of selected tree, shrub, and vine species stored in the field'. *New Forests* 8: 143–154.

Holloran, P. (2006) 'Measuring performance of invasive plant eradication efforts in New Zealand'. *New Zealand Plant Protection* 59: 1–7.

Howell, C.J. (2012). 'Progress toward environmental weed eradication in New Zealand'. *Invasive Plant Science and Management* 5: 249–258.

CABI, (2017a). *Passiflora edulis* (passionfruit). In: Invasive Species Compendium. Wallingford, UK: CAB International. <www.cabi.org/isc>.

CABI, (2017b). *Psidium guajava* (guava). In: Invasive Species Compendium. Wallingford, UK: CAB International. <www.cabi.org/isc>.

Kammili, A. and Jatothu, J. (2015). 'Differential longevity of castor (*Ricinus communis* L.) germplasm conserved under uncontrolled storage conditions across extended periods'. *Journal of Crop Improvement* 29: 706–719.

Owen, S-J. (1998) *Department of Conservation Strategic Plan for Managing Invasive Weeds*. Department of Conservation, Wellington.

Panetta, F.D. (2015). 'Weed eradication feasibility: lessons of the 21st century'. *European Weed Research Society* 55: 226–238.

Parkes, J.P. (1984). 'Feral goats on Raoul Island II: Diet and notes on the flora'. *New Zealand Journal of Ecology* 7: 95–101.

Pyšek, P., Richardson, D.M., Rejmánek, M., Webster, G.L., Williamson, M. and Kirschner, J. (2004). 'Alien plants in checklists and floras: towards better communication between taxonomists and ecologists'. *Taxon* 53: 131–143.

Sykes, W.R. and West, C.J. (1996). 'New records and other information on the vascular flora of the Kermadec Islands'. *New Zealand Journal of Botany* 34: 447–462.

Sykes, W.R., West, C.J., Beever, J.E. and Fife, A.J. (2000). *Kermadec Islands Flora - Special Edition: A Compilation of Modern Material about the Flora of the Kermadec Islands*. Lincoln, Canterbury, N.Z.: Manaaki Whenua Press.

Uowolo, A.L. and Denslow, J.S. (2008). 'Characteristics of the *Psidium cattleianum* (Myrtaceae) seed bank in Hawaiian lowland wet forests'. *Pacific Science* 62: 129–135.

Veitch, C.R., Gaskin, C., Baird, K. and Ismar, S.M.H. (2011). 'Changes in bird numbers on Raoul Island, Kermadec Islands, New Zealand, following the eradication of goats, rats and cats'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 372–376. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Veitch, C.R., Miskelly, C.M., Harper, G.A., Taylor, G.A. and Tennyson, A.J.D. (2004). 'Birds of the Kermadec Islands, south-west Pacific'. *Notornis* 51: 61–90.

West, C.J. (1996). Assessment of the weed control programme on Raoul Island, Kermadec Group. *Science & Research Series no 98*. Department of Conservation, Wellington.

West, C.J. (2002). 'Eradication of alien plants on Raoul Island, Kermadec Islands, New Zealand'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 365–373. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

West, C.J. (2011). 'Consideration of rat impacts on weeds prior to rat and cat eradication on Raoul Island, Kermadecs'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 244–247. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

West, C.J. and Havell, D. (2011). 'Plant responses following eradication of goats and rats from Raoul Island, Kermadecs'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* p. 535. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

West, C.J. and Havell, D. (2013). *Raoul Island Review – Phase 2 Part 1: Review of Biodiversity programme.* Unpublished internal report, DOCDM-1157983. Department of Conservation.

West, C.J. and Thompson, A.M. (2013). 'Small, dynamic and recently settled: responding to the impacts of plant invasions in the New Zealand (Aotearoa) archipelago'. In: L.C. Foxcroft, P. Pyšek, D.M. Richardson and P. Genovesi (eds.) *Plant Invasions in Protected Areas: Patterns, Problems and Challenges*, 7: 285–311. Invading Nature – Springer Series in Invasion Ecology.

Successful eradication of signal crayfish (*Pacifastacus leniusculus*) using a non-specific biocide in a small isolated water body in Scotland

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Abstract The North American signal crayfish (*Pacifastacus leniusculus*) has been present in Scotland since at least 1995 and the species is now known to be present in a number of catchments. Once established, few opportunities for containment exist and eradication can often be impossible to achieve. However, in small, isolated water bodies, the application of a non-crayfish-specific biocide has provided the opportunity to remove this species permanently. In July 2011, signal crayfish were discovered in a flooded quarry pond at Ballachulish in the Scottish Highlands. This is an isolated site located ~100 km from the nearest known population and it is likely that the population was established as the result of a deliberate release of these animals 10 years previously. Experience gained from using the eradication technique at other sites in the UK led to the site being treated with a natural pyrethrum biocide (Pyblast®) in June 2012. Post treatment monitoring from 2012–2017 indicates that eradication has been successful. Monitoring of native species affected by the biocide suggests that both invertebrates and amphibians quickly recolonised the quarry pond. Eradication of crayfish using biocide is only feasible in water bodies where the entire population of crayfish can be exposed to a lethal dose and the impact on non-target species can be accepted. The technique is not appropriate for large, connected water bodies, although it may be possible to treat short stretches of canals where biocide exposure can be controlled and isolated populations of crayfish can be effectively treated.

Keywords: invasive species, natural pyrethrum, ponds

INTRODUCTION

Invasive species are the second largest cause of biodiversity loss globally through species extinction and habitat destruction (EEA, 2012). Their impact can be dramatic and often irreversible, so it is important that their spread is contained and that eradication is achieved wherever practicable. As a function of their isolation, islands may offer the best hope of locally eradicating an invasive species. Conventionally we think of islands as areas of land which are surrounded by water. However, for obligate aquatic species the reverse may be true, and it is the land which can form an effective barrier to invasion. In freshwater ecosystems, invasive non-native species can pose a major threat to native species through competition, predation and transmission of diseases (EEA, 2012) and their control in these 'freshwater islands', is therefore of particular importance.

Newly introduced species can establish rapidly and it is important to detect their presence, and take action, as early as possible. This is often not possible because, unlike terrestrial habitats, freshwaters are not easily surveyed (Boon & Bean, 2010). This means that invasive species in these habitats may not be detected until they have become fully established, often making it more expensive to remove them (Simberloff, et al., 2013).

Signal crayfish (*Pacifastacus leniusculus*) have been introduced to over 20 European countries since the 1960s. After escaping from farms in the 1970s they are now widespread across parts of England and Wales (Bean, et al., 2004). The species was first discovered in Scotland in 1995 (Maitland, 1996). In just over 10 years it had been illegally introduced into at least eight river catchments (Gladman, et al., 2009). Signal crayfish are omnivores and, through increased grazing pressure and predation, they can reduce the diversity of aquatic invertebrates and significantly alter food webs (Holdich, et al., 2014). As well as direct predation of eggs and young fish, they compete with Atlantic salmon (*Salmo salar*) and trout (*S. trutta*) for food and space and can mobilise sediment, causing silting of spawning beds (Gladman, et al., 2012; Bean & Yeomans, 2016).

Controlling signal crayfish has proved difficult and, in most situations, impossible to achieve. Several approaches have been attempted, ranging from the physical removal of animals using techniques such as trapping and electrofishing, to the construction of barriers to prevent their spread (Bean & Yeomans, 2016). Of these, trapping is often perceived as being the easiest and most effective option. In reality, however, the removal of crayfish by trapping has proved ineffective at eradicating signal crayfish because it does not remove the entire population (Freeman, et al., 2010). Where trapping has been allowed to take place on a commercial basis, either as a management tool or for the establishment of legal fisheries, it has been associated with the detection of an increased number of illegal introductions (e.g. Alonso, et al., 2000; Diéguez-Uribeondo, 2006; Arce & Alonso, 2011; Bohman, et al., 2011).

The use of biocides to control or eradicate crayfish populations is a relatively recent development. Early attempts to eradicate signal crayfish using chlorinated lime (Kozak & Policar, 2003) were not successful. However, later trials using natural pyrethrum (as Pyblast®) (Peay, et al., 2006) showed more promise in trials in Scottish freshwaters without being totally effective. O'Reilly (2015) provides a comprehensive review of the toxicity of Pyblast® and other organophosphates for signal crayfish control.

There is no single biocide available that is selective for signal crayfish only. This means that any attempted eradication using a biocide treatment would be expected to kill some, or all, of the non-target invertebrate and vertebrate fauna in the area being treated.

Signal crayfish were first detected in north-west Scotland in an artificial waterbody, a flooded slate quarry, near Ballachulish, in 2011. This species is thought to have

In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) (2019). Island invasives: scaling up to meet the challenge, pp. 443–446. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

been present within the pond for approximately 12 years prior to its discovery there (P. Madden, pers. comm.). The nearest signal crayfish population to that discovered at Ballachulish is located over 100 km south in the River Kelvin near Glasgow. This reinforced the initial view that this species was introduced to the Ballachulish quarry pond by people; in addition, it has footpaths and a recreational area adjacent so is readily accessed by the public. The pond, and therefore the crayfish population, was isolated with no source of natural re-infestation. However the proximity of the pond to local rivers, and the potential impact that this species may have on species of conservation and recreational value, such as Atlantic salmon and trout, made it essential that the signal crayfish population was removed as soon as possible.

Study site

Ballachulish quarry pond (Ordnance Survey Great Britain National Grid Reference NN08525824) is located immediately west of the town of Ballachulish on the west coast of Scotland (Fig. 1). The area contains a large pond and a smaller waterbody located 25 m to the north (NGR NN08435835). The affected waterbody has a surface area of 18,776 m^2 and a volume of 46,000 m^3. Whilst it is relatively shallow over much of its surface area (approximately 0–5 m deep), a smaller area of deeper water, extending to a maximum depth of 13 m, is present.

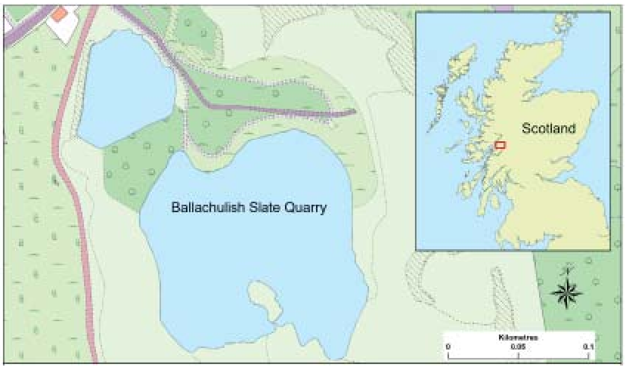

Fig. 1 The location of Ballachulish and the quarry pond relative to western Scotland.

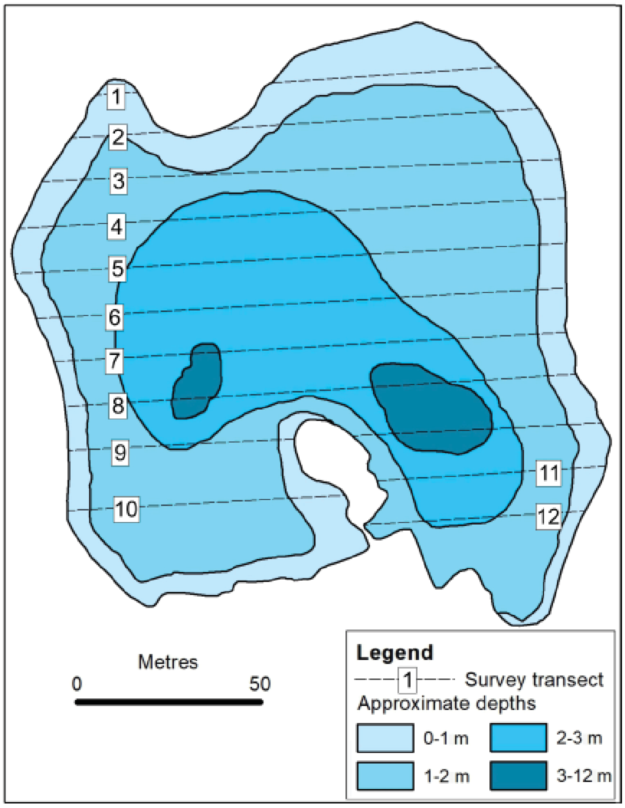

Fig. 2 Ballachulish quarry pond bathymetry.

Survey prior to any management action revealed that signal crayfish were restricted to the larger of the two ponds. The large pond also hosted a number of invertebrate and vertebrate species. Vertebrates found during the survey included fish (trout and European eel (*Anguilla anguilla*)) and amphibian species such as common toad (*Bufo bufo*) and palmate newt (*Lissotriton helveticus*).

METHODS

It was deemed acceptable, given the absence of any conservation priority species, that some mortality of native fauna would occur as a result of biocide application. The risk of inadvertently transferring juvenile crayfish in the act of translocating rescued animals to new locations meant that no attempt was made to rescue non-target species prior to the treatment taking place.

Bathymetric transects of the pond were obtained by the use of a plumb-line at 100 sample points (Fig. 2). These were used to divide the pond into compartments of equal volume. A total volume of 620 l of Pyblast® was applied to the surface of the pond by boat-mounted sprayers (Fig. 3) to achieve a target dose rate of at least 0.3 mg/l, on 12 June 2012. Water pumps and a boat with an outboard motor were used to ensure thorough mixing throughout the entire water column. In addition, backpack sprayers treated a 1 m band around the edge of the pond and the shallow margins of the pond to prevent signal crayfish leaving the water (Fig. 4). The following day, deep water sections of the pond were re-treated by spraying Pyblast® down 6 m-long rigid hoses, increasing the dose rate in the deepest areas of the pond to at least 0.4 mg/l. Mixing was achieved, as far as possible, using an outboard engine and shore-based pumps.

The effectiveness of the treatment was monitored by placing 13 sentinel cages, each containing 10 signal crayfish, of mixed sex, into the pond at different positions and depths and monitoring their mortality once the biocide had been applied. Bioassays using the freshwater shrimp (*Gammarus pulex*) as a test organism, were conducted according to the methodology described by Peay, et al. (2006). These were run on the pond water to monitor its toxicity at the point of treatment and to monitor the breakdown of the Pyblast® over subsequent days and weeks. Natural pyrethroids break down quickly when exposed to sunlight and their toxicity should reduce

Fig. 3 Pyblast® being applied to the surface of the pond from boat-mounted sprayers.

Fig. 4 Using a backpack sprayer to deliver biocide to the quarry pond edge.

Fig. 5 Dead signal crayfish in the margins of the pond following Pyblast® treatment.

rapidly. During this eradication exercise, toxicity levels, sufficient to kill *Gammarus*, persisted for 34 days.

The effectiveness of the signal crayfish removal attempt was monitored through baited Fladen-style traps set in the pond for a total of 195 trap nights in August/September each year for five years post-treatment (975 trap nights in total). Traps were set in a range of habitats and depths throughout the site to maximise the potential of capture. The ability of invertebrates to recover very quickly, in as little as 24 days, was already known from other studies (e.g. Peay, et al., 2006), therefore recovery of the pond ecosystem was assessed through amphibian surveys carried out using sweep netting and kick-sampling in late June, August, September and October 2012. Larval common toad and palmate newts were measured, aged and their general behaviour assessed to determine whether it deviated from that normally expected in undisturbed sites.

RESULTS

During, and immediately after the Pyblast® application, signal crayfish held in sentinel cages were checked intermittently to assess mortality levels and the efficacy of treatment. By the end of the first day (12 June 2011) most of the signal crayfish were dead (Fig. 5), however, those in deep water sections (as determined from the use of sentinel crayfish) were still active. Effort was focused on increasing the concentration of Pyblast® in these areas and by the third morning (14 June 2011) all signal crayfish, even in the deep sections, were dead. The annual post-treatment monitoring found no signal crayfish in the pond for five years after the treatment and in August 2017 the eradication was declared successful.

Bioassay monitoring indicated that after one month the concentration of Pyblast® in the pond was below the lethal limit for *G. pulex* and it was judged safe to re-open the pond to the public. Fig. 6 shows the speed at which the reduction in toxicity of water samples taken from the surface and 5 m depth in the pond took place following Pyblast® treatment. These data show that biocide toxicity in deeper waters took longer to drop below lethal levels than those near the surface, but confirmed that toxicity levels dropped to levels non-lethal to signal crayfish in all areas within 20 days post-treatment.

The amphibian surveys found larval stages of common toad and palmate newt in late June 2012, which strongly suggested they had survived the Pyblast® treatment. There was no difference in size or development stage between tadpoles from the treated pond and a nearby untreated pond. All amphibian larvae behaved normally and showed no physical abnormalities (see O'Brien, et al., 2013). A low level of fish mortality was observed and this included one brown trout plus a very small number of European eels and three-spined sticklebacks (*Gasterosteus aculeatus*).

DISCUSSION

Post-treatment monitoring demonstrated that the application of Pyblast® at a target dose rate of 0.3 mg/l was successful in removing signal crayfish from the pond. Monitoring also showed that a number of non-target species survived the treatment, or were able to recolonise quickly (O'Brien, et al., 2013). The pond is artificial and the presence of signal crayfish would have significantly altered its ecology, meaning that there is no recent, or pre-crayfish, baseline against which to measure ecosystem recovery. However, five years after the biocide treatment an abundant invertebrate and amphibian fauna is present within the pond, and no lasting chemical effect of the treatment is visible.

The risk of signal crayfish being spread to new waterbodies within the local area by natural or anthropogenic means has been reduced as a result of this successful eradication. There are now no known populations in the north-west Highlands which pose a threat of re-introduction to this site. This project has shown that full eradication is achievable in small, isolated waterbodies where the entire signal crayfish population can be exposed to a natural pyrethroid biocide, and the impact on non-target species is deemed an acceptable risk.

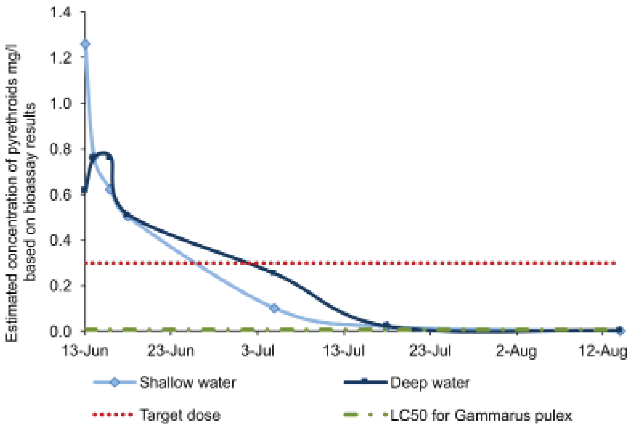

Fig. 6 Graph showing the reduction in toxicity of water samples taken from the surface and 5 m depth in the pond following Pyblast® treatment on 12t June 2012. Toxicity was estimated through a bioassay exposing *Gammarus pulex* to diluted samples and comparing to previous reference data collected on their mortality rates.

A partnership approach to dealing with signal crayfish in this location was a significant component of its success. Buy-in from public agencies and the local fisheries management sector provided the financial and physical resources required to provide adequate materials to carry out this work effectively. Ongoing promotion of a national biosecurity campaign aims to prevent future reintroductions. At an operational level, careful monitoring of sentinel signal crayfish and having a sufficient contingency of Pyblast® to supplement concentrations in the deepest areas of the pond proved crucial. The quarry pond at Ballachulish is the largest water body in the UK to date where signal crayfish have been eradicated using a natural pyrethroid. The main limitations to the wider application of this method to large waterbodies are the financial cost of the biocide (in 2012, Pyblast® cost over £50 per litre), the manpower required, the collateral damage to native biota and connectivity to outflowing rivers and streams. This trial accounted for biocide costs of >£30k alone, and with additional costs in terms of staff time and equipment hire (pumps, etc.) the total estimated figure was £73.1k. Additional costs associated with post-treatment monitoring are not included within this total. In Sweden and Norway, less expensive synthetic pyrethroids have been used (Sandodden & Johnsen, 2010), but these have the disadvantage of being more toxic and persistent in the freshwater environment. O'Reilly (2015) showed, using laboratory-based acute toxicity tests, that signal crayfish were most sensitive to Detamethrin, a synthetic pyrethroid, used in the aquaculture industry, and that juvenile signal crayfish were significantly more sensitive to Pyblast® than adult conspecifics at concentrations far lower than those used in this study (57.95 µg/l versus 0.3 mg/l). It may be possible, therefore, to use alternative biocide approaches in some situations, or lower the costs of treatment in populations which are detected at an earlier stage in their establishment. Recent advances in the detection of invasive species by environmental DNA may allow for earlier, and cheaper, identification of new populations through the expansion of surveillance networks to include a larger number of waterbodies. Environmental DNA assays have already been developed for signal crayfish (Larson, et al., 2017; Harper, et al., 2018) and a wide range of other biota (e.g. Ficetola, et al., 2008) which will allow for cheaper, and possibly more reliable, pre-and post-treatment monitoring of signal crayfish and other species to take place in future years.

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REFERENCES

Alonso, F., Temiño, C. and Diéguez-Uribeondo, J. (2000). 'Status of the white-clawed crayfish, *Austropotamobius pallipes* (Lereboullet, 1858) in Spain: Legislation and conservation'. *Bulletin Francais de la Peche et de la Pisciculture* 356: 31–54.

Arce, J.A. and Alonso, F. (2011). 'Factors related to the presence of the *Austropotamobius pallipes* (Lereboullet, 1858) species complex in calcareous mountain rivers in central Spain'. *Knowledge and Management of Aquatic Ecosystems* 401: 25.

Bean, C.W., Maitland, P.S. and Collen, P. (2004). 'Crayfish in Scotland: A review of current status and legislative control'. *Freshwater Crayfish* 15: 220–228.

Bean, C.W. and Yeomans, W.E. (2016). 'North American Signal Crayfish'. In: M.J. Gaywood, P.J. Boon, D.B.A Thompson and I.M Strachan (eds.) *The Species Action Framework Handbook*, pp. 314–324. Perth, UK: Scottish Natural Heritage.

Bohman, P., Degerman, E., Edsman, L. and Sers, B. (2011). 'Exponential increase of signal crayfish in running waters in Sweden – due to illegal introductions?' *Knowledge and Management of Aquatic Ecosystems* 401: 23.

Boon, P.J. and Bean, C.W. (2010). 'Freshwater Non-Natives – Prevention, Control or Eradication?' In: J.M. Baxter and C.A. Galbraith (eds.) *Species Management: Challenges and Solutions for the 21st Century*, pp. 229–246. London, UK: Her Majesty's Stationery Office.

Diéguez-Uribeondo, J. (2006). 'The dispersion of the *Aphanomyces astaci*-carrier *Pacifastacus leniusculus* by humans represents the main cause of disappearance of the indigenous crayfish *Austropotamobius pallipes* in Navarra'. *Bulletin Francais de la Peche et de la Pisciculture* 380–381: 1303–1312.

EEA (2012). *The Impacts of Alien Species in Europe*. Copenhagen: European Environment Agency.

Ficetola, G.F., Miaud, C., Pompanon, F. and Taberlet, P. (2008). 'Species detection using environmental DNA from water samples'. *Biology Letters* 4: 423–425.

Freeman, M.A., Turnbull, J.F., Yeomans, W.E. and Bean, C.W. (2010). 'Prospects for management strategies of invasive crayfish populations with an emphasis on biological control'. *Aquatic Conservation: Marine and Freshwater Ecosystems* 20: 211–223.

Gladman, Z., Adams, C., Bean, C., Sinclair, C. and Yeomans, W. (2009). 'Signal Crayfish in Scotland'. In: J. Brickland, D.M. Holdich and E.M. Imhoff (eds.) *Crayfish Conservation in the British Isles. Proceedings of a Conference held on 25th March 2009 in Leeds, UK*, pp. 43–48. Bristol, UK: Environment Agency.

Gladman, Z.F., Adams, C.E., Bean, C.W., Long, J. and Yeomans, W.E. (2012). 'Investigating the threat of non-native North American signal crayfish (*Pacifastacus leniusculus*) to salmonid redds'. *Aquatic Conservation: Marine and Freshwater Ecosystems* 22: 134–137.

Harper, K.J., Anucha, P.N., Turnbull, J.F., Bean, C.W. and Leaver, M.J. (2018) Searching for a signal: Environmental DNA (eDNA) for the detection of invasive signal crayfish, *Pacifasticus leniusculus* (Dana, 1852). *Management of Biological Invasions* 9: 137-148.

Holdich, D.M., James, J., Jackson, C. and Peay, S. (2014). 'The North American signal crayfish, with particular reference to its success as an invasive species in Great Britain'. *Ethology, Ecology and Evolution* 26: 232–262.

Kozak, P. and Policar, T. (2003). 'Practical Elimination of Signal Crayfish, *Pacifastacus leniusculus* (Dana), from a Pond'. In: D.M. Holdich and P.J. Sibley (eds.) *Management and Conservation of Crayfish: Proceedings of a conference held on 7 November 2002*, pp. 200–208. Bristol, UK: Environment Agency.

Larson, E.R., Renshaw, M.A., Gantz, C.A., Umek, J., Chandra, S., Lodge, D.M. and Egan, S.P. (2017). 'Environmental DNA (eDNA) detects the invasive crayfishes *Orconectes rusticus* and *Pacifastacus leniusculus* in large lakes of North America'. *Hydrobiologia* 800: 173–185.

Maitland, P.S. (1996). 'The North American signal crayfish, *Pacifastacus leniuculus* (Dana), established in the wild in Scotland'. *Aquatic Conservation: Marine and Freshwater Ecosystems* 6: 107–110.

O'Brien, C.D., Hall, J.E., O'Brien, C.T., Baum, D. and Ballantyne, L. (2013). Impact of a natural pyrethrin biocide on two amphibians, common toad *Bufo bufo* and palmate newt *Lissotriton helveticus*, in Highland, UK. *Conservation Evidence* 10: 70–72.

O'Reilly, S. (2015). 'Assessing the Toxicity of Biocides on the North American Signal Crayfish *Pacifastacus leniusculus* (Dana) to Aid Eradication'. MSc thesis. Glasgow: University of Glasgow.

Peay, S., Hiley, P.D., Collen, P. and Martin, I. (2006). 'Biocide treatment of ponds in Scotland to eradicate signal crayfish'. *Bulletin Francais de la Peche et de la Pisciculture* 380–381: 1363–1379.

Sandodden, R. and Johnsen, S.I. (2010). 'Eradication of introduced signal crayfish *Pacifastacus leniusculus* using the pharmaceutical BETAMAX VET'. *Aquatic Invasions* 5: 75–81.

Simberloff, D., Martin, J.-L., Genovesi, P., Maris, V., Wardle, D.A., Aronson, J., Courchamp, F., Galil, B., Garcia-Berthou, E., Pascal, M., Pysek, P., Sousa, R., Tabacchi, E. and Vila, M. (2013). 'Impacts of biological invasions: What's what and the way forward'. *Trends in Ecology and Evolution* 28: 58–66.

Small- and large-scale eradication of invasive fish and fish parasites in freshwater systems in Norway

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Abstract In July 2016, the European Union adopted a list of invasive alien species of concern, and at present there are two freshwater fish species on the list. Member states are obliged to prevent further spread and to perform rapid eradication when problem species are discovered at new sites, but continental EU member states have limited experience with eradication of fish. Eradications are more likely to succeed if the invasive species is confined to insular habitats. Freshwater invasives can be regarded as island invasives, since their habitats have boundaries against shorelines, saline waters, waterfalls and dams, and these boundaries make eradications possible. CFT Legumine® containing rotenone is the only legal piscicide in the EU, and Norway has used CFT Legumine® in eradication efforts for many years. Species that have been introduced outside their native range and have been successfully eradicated include minnow (*Phoxinus phoxinus*), roach (*Rutilus rutilus*), pike (*Esox lucius*), common whitefish (*Coregonus lavaretus*), and the salmon parasite *Gyrodactylus salaris*. This manuscript summarises the eradication efforts of invasive fish species and fish parasite species during the last two decades in Norway, covering eradications from such diverse habitats as small ponds, lakes, marshlands, small streams and large rivers. An estimated £100 million has been spent in the *Gyrodactylus salaris* eradication programme. Costs of invasive fish eradications are given, ranging from less than £10,000 to more than £200,000. There are no known invasive fish eradication failures in Norway in the last 20 years. A summary of the efforts in Norway can be an aid for planning control and eradication measures of invasive fish species in other countries.

Keywords: CFT Legumine, fish control, *Gyrodactylus salaris*, IAS, rotenone

INTRODUCTION

Alien invasive fish species are a global problem (Gozlan, et al., 2010). Eradication is more likely to succeed if the invasive species is confined to insular habitats. Freshwater invasives can be regarded as island invasives, since their habitats have boundaries against shorelines, saline waters, waterfalls and dams, and these boundaries make eradications possible.

The EU requires member states to rapidly implement measures, including eradications, against invasive alien species. In July 2016 the EU adopted a list of invasive alien species of European Union concern that requires control or eradication (<http://data.europa.eu/eli/reg_impl/2016/1141/oj>). This list includes two fish species, topmouth gudgeon (*Pseudorasboras parva*) and Amur sleeper (*Perccottus glenii*), and new species can be added. Transfer of knowledge is therefore essential as many EU countries have little experience with such operations. In Europe, successful eradications against invasive freshwater fish have been done in Spain (Fernandez-Delgado, 2009), England (Britton, et al., 2010) and Norway.

The Norwegian Veterinary Institute has extensive knowledge of fish eradication through their work on behalf of the Norwegian Environment Agency. A simplified way to look at historic immigration routes for freshwater species to Norway is that all indigenous freshwater fish species can be found in south-eastern Norway, while the rest of the country has very few indigenous species, making most south-eastern species, e.g. all cyprinids and pike (*Esox lucius*), domestic exotic in other parts of the country (Huitfeldt-Kaas,1918). Exotic invasive fish are North-American salmonids, imported for aquaculture and improvements of wild stocks, and cyprinids from the European continent (Hesthagen & Sandlund, 2016). One of the most severe threats to an indigenous fish species was the introduction of the salmon fluke *Gyrodactylus salaris*, which in a worst-case scenario could lead to local extinction of Atlantic salmon (*Salmo salar)* populations. Norwegian authorities have committed to eradicate the salmon fluke from Norwegian rivers, and the Norwegian Veterinary Institute is in charge of the project planning and eradication campaigns. The experiences from these campaigns against *G. salaris* are used in other operations against invasive freshwater fish species.

The piscicide rotenone has been used for fish control and eradication for more than 70 years (McClay, 2000). Rotenone is a natural product isolated from roots of tropical plants in the pea family *Leguminosae*, and it is highly toxic to fish (Ling, 2003). The rotenone product used in Europe is CFT Legumine® (CFT L). It is the only piscicide currently under assessment of the Biocidal Product Regulation (BPR, Regulation (EU) 528/2012), and thus the only piscicide legal for use in Europe. The effect of rotenone on non-target organisms has been extensively studied (Ling, 2003; Vinson, et al., 2010; Finlayson, et al., 2010a; Dalu, et al., 2015) and, even if some invertebrate taxa are very sensitive, the general findings from Norway are that most taxa recolonise treated areas within a year (Fjellheim, 2004; Kjærstad, et al., 2015).

Two different solutions of CFT L have been used in the described treatments. The first CFT L formula used contained 2.5% rotenone and the synergist piperonylbutoxid (PBO). Since PBO did not have the desired synergic effect (Finlayson, et al., 2010a), the manufacturer made a change in the product in 2012. The new product omitted PBO and increased rotenone content to 3.3%. As of 2013, all treatments described have used the 3.3% solution.

The objective of this manuscript is to present all rotenone treatments against invasive freshwater fish species in Norway the last 20 years, and a short summary of the still ongoing eradication campaign against the salmon fluke. None of the invasive fish species eradications are previously published. Only lake volumes are described in detail, but the treatment area also includes adjacent streams, pools and marshlands, to ensure that no fish survives in temporary locations. The work of treating these surroundings varies depending on the site, but the amount of CFT L used in these areas is small compared to the amount used in lakes. A map is included for the geographical location of the invasive fish species eradications (Fig. 1). Costs are included to give an idea about the cost of invasive fish eradications (Table 1). The following descriptions can be an aid for planning control and eradication measures of invasive fish species in the EU and for other stakeholders.

In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) (2019). Island invasives: scaling up to meet the challenge, pp. 447–451. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

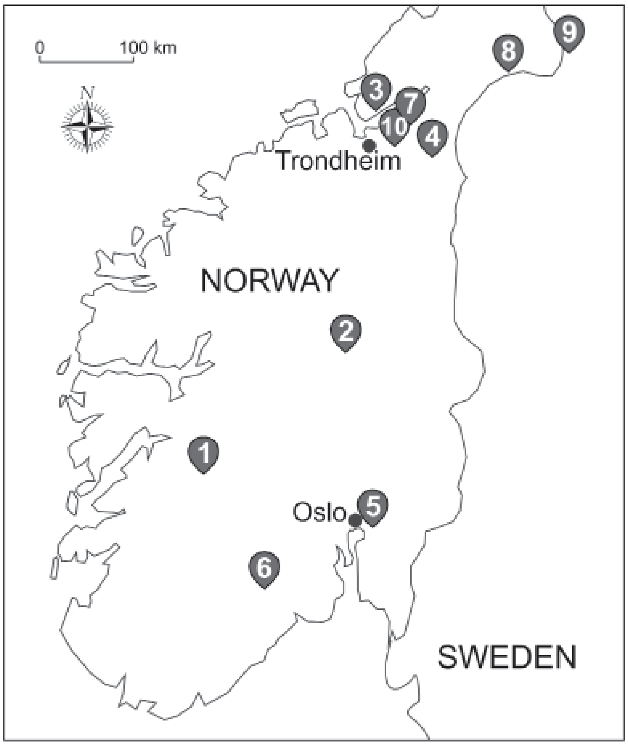

Fig. 1 Geographical location of invasive fish eradications. 1) Hardangervidda, 2) Sør-Fron, 3) Lake Ålmotjønna, 4) Lake Alsettjønna, 5) Lake Lille Mortetjern, 6) Telemark Canal, 7) Lake Vikerauntjønna, 8) Lake Klokkartjønna, 9) Gäddede (in Sweden) 10) Bymarka.

OVERVIEW OF ERADICATION OF INVASIVE FISH AND FISH PARASITES IN FRESHWATER SYSTEMS IN NORWAY

Salmon fluke (*Gyrodactylus salaris)* in rivers and lakes in Norway

The salmon fluke *G. salaris* is a freshwater salmon ectoparasite indigenous to the Baltic region, and an exotic invasive species in Norway that was first detected in the 1970s. *G. salaris* is one of the most severe threats against Norwegian Atlantic salmon (Anon., 2016), It has been introduced via fish transports from Sweden, and distributed in Norwegian rivers through stocking from salmon hatcheries. Norwegian Atlantic salmon populations are highly susceptible to the parasite, with up to 95 % mortality for salmon fry and parr (Johnsen & Jensen, 1991). Rainbow trout (*Oncorhynchus mykiss*) and Arctic char (*Salvelinus alpinus*) can also host the salmon fluke. It has been found in 50 rivers in Norway, and rotenone treatments aim to eradicate the Atlantic salmon from infected river systems since the parasite cannot survive without its host. The salinity of the fjords acts as a barrier for dispersion of the salmon fluke, providing defined borders for the treatment area (Soleng & Bakke, 1997). Extensive operations to preserve and re-establish local strains of Atlantic salmon and sea trout (*Salmo trutta*) are performed before, during and after eradications (O'Reilly & Doyle, 2007). Forty-three rivers throughout Norway have been treated and the salmon fluke has so far been successfully eradicated from 31 rivers. Meanwhile, 12 rivers are still under post-treatment surveillance awaiting confirmation of eradication. The rivers differ in size but eradication campaigns have included 42 km long rivers in rugged terrain (Sandodden, et al., 2018) and 10 rivers in the same fjord system in the Vefsna region, where River Vefsna had a discharge of 200 m^3/s on the day of treatment (Stensli & Bardal, 2014). Also, in the Vefsna region, the salmon fluke infested Arctic char in three lakes. In total these lakes covered more than 18 km^2, two of them 65 m deep. The Vefsna region was one of the largest rotenone treatments ever performed, for both lake and riverine systems (Stensli & Bardal, 2014). Confirmation of eradication was attained in the rivers of the Vefsna region in 2017. The lakes in the same region still await eradication confirmation because of a different procedure for eradication confirmation.

Minnow (*Phoxinus phoxinus*) in Hardangervidda National Park

The minnow is indigenous to south-eastern Norway, but has been introduced to most parts of the country. It is believed that minnow has been spread through the use of live bait and also accidentally released, mistaken as brown trout (*Salmo trutta*) fry. Minnow have been present at Hardangervidda for decades. Minnows can multiply to high numbers and food competition has had a negative impact on local fish stocks and birdlife. Minnows are not present in western watercourses at Hardangervidda National Park, a high altitude, tree-less plateau in southwest Norway, but were found up to the water divide in several places. The risk of further dispersion across

Table 1 Year of treatment, location and target species, with approximate volume of lakes, litres of CFT L used and approximate cost of treatment.

Year	Location	Target species	Volume (1,000 m^3)	CFT L volume used (l)[a]	Cost (£1,000)
1999	Hardangervidda	Minnow		137	30–50
2005	Sør-Fron	Rainbow trout	5.6	7.5	<10
2008	Lake Ålmotjønna	Roach	120	180	10
2008	Lake Alsettjønna	Common whitefish	84	90	10
2009	Lake Lille Mortetjern	Roach	2.9	4.5	10
2012	Telemark Canal	Pike		100	30–50
2013	Hardangervidda	Minnow	640	805	>100
2014	Telemark Canal	Pike		22	10
2014	Lake Vikerauntjønna	Roach	188	293	10–30
2015	Lake Klokkartjønna	Lake trout	675	670	30–50
2016	Gäddede	Several species		8	10
2016	Bymarka	Roach	2,500	4,000	>200

a) CFT L used also includes CFT L in streams, pools and marshlands surrounding the lakes.

the water divide was regarded as imminent. A successful treatment was conducted in 1999–2000 in the area around Stigstuv (Tønset & Bakkeli, 2000), which is set at the east-west water divide at Hardangervidda National Park. Fish barriers were built to create a buffer zone towards the water divide. Minnows were found in the buffer zone again 10 years later, most likely because the barriers were not working properly. In flooding periods the water level downstream from the barrier could rise and thus level out the height difference. The barriers were adjusted in 2013 prior to a new treatment of the Stigstuv area. The treatment area comprised 40 ponds/small lakes, streams and marshes within an area of 2 km^2. Total water coverage of the ponds/small lakes was 140,000 m^2, with depths of up to 4 m and average depths of 0.5–1 m. The treatment was performed during four days in August 2013 by 16 people. Target dose was 1 ppm CFT L, and a total of 225 l of CFT L was used.

In addition, Lake Hætjørna, another site on Hardangervidda, also was treated due to minnow invasion. Two new barriers had been built, one in the inlet and one in the outlet, making the lake a buffer zone without minnows. Twelve people treated the lake, ponds and marshes in the surrounding area in two days, just prior to the treatment at Stigstuv. Lake Hætjørna covers an area of 0.2 km^2, with an estimated mean depth of 2.5 m and maximum depth 8 m. The target dose was 1 ppm CFT L, and a total of 580 l of CFT L was used. No minnows were caught in hoop net surveys in 2014 and 2016. An environmental DNA survey found no traces of minnows at either site in 2016 (Fossøy, et al., 2017).

Rainbow trout (*Oncorhynchus mykiss*) in Sør-Fron

Sør-Fron is a municipality in Oppland County. Rainbow trout were found in four artificial ponds at a farm in Sør-Fron. Rainbow trout are indigenous to North-America, and have been exported worldwide for angling and fish farming. In Sør-Fron it had been released in these ponds, and netting in October 2000 confirmed their existence. Rainbow trout can host the salmon fluke and thus act as a vector for the parasite. There was risk of escape from the ponds during large floods in the nearby River Lågen. The ponds were treated with rotenone in October 2005. The volumes ranged from 350 to 3,750 m^3, and two people completed the job in one day. The target dose was 1 ppm CFT L, and a total of 7.5 l CFT L was used. On arrival, all ponds were covered with ice. Ice cover was broken before adding CFT L, and the dosage was increased slightly to compensate for the low water temperatures. Prior to treatment the number of fish was reduced through netting. There has been no programme for eradication confirmation, but there have been no later reports of rainbow trout in the ponds.

Roach (*Rutilus rutilus*) in Lake Ålmotjønna

Lake Ålmotjønna is situated in Rissa municipality in Sør-Trøndelag County. The roach is indigenous to south-eastern Norway, but alien to the Trøndelag region. It is believed that roach were released by anglers. Roach were discovered in Lake Ålmotjønna in the summer of 2007. The purpose of the treatment was to prevent further spread downstream to the large Lake Storvatnet, which could lead to a permanent foothold for roach in the region. The volume of Lake Ålmotjønna was 120,000 m^3, and average depth was 5 m. A rotenone treatment was conducted in August 2008. Two people worked for one day. Target dose was 1.5 ppm CFT L, and a total of 180 l CFT L was used. Only two dead roach were found post-treatment. Fish scale analysis revealed that the roach had been introduced at least five years prior to treatment, and apparently had not reproduced. There has been no programme for eradication confirmation, but there have been no later reports of roach in the lake.

Common whitefish (*Coregonus lavaretus*) in Lake Alsettjønna

Lake Alsettjønna is situated in Selbu municipality in Sør-Trøndelag County. The common whitefish is indigenous in south-eastern Norway and eastern watersheds, but alien to the region. Common whitefish was released in Lake Alsettjønna around 1875 as part of a wedding gift. The purpose of the treatment was to prevent further spread to the larger Lake Selbusjøen, which could lead to a permanent foothold for common whitefish in the region. Common whitefish is a food competitor of the indigenous brown trout and Arctic char and can be more effective in exploitation of food sources. It also has a large capacity for propagation. Increased rainfall in the catchment could, in the future, make the stream from Lake Alsettjønna habitable for common whitefish in flooding periods, leading it to spread to Lake Selbusjøen. A rotenone treatment was conducted in Lake Alsettjønna in August 2008. The volume was 84,000 m^3, and average depth was 4 m. Two people worked for one day. Target dose was 1 ppm CFT L, and a total of 90 l CFT L was used. There has been no programme for eradication confirmation, but there have been no later reports of common whitefish in the lake.

Roach (*Rutilus rutilus*) in Lake Lille Mortetjern

Lake Lille Mortetjern is situated in Nittedal municipality in Akershus County. The roach is indigenous to south-eastern Norway, where Lake Lille Mortetjern is situated, but fish had not been recorded in this lake before, making it ideal for amphibians. Roach are present in a neighbouring lake in walking distance, so suspicion is that it has been carried from there and released into Lille Mortetjern. Roach were discovered here in 2007. The lake is known for its rich population of amphibians. The endangered smooth newt (*Lissotriton vulgaris*), great crested newt (*Triturus cristatus*), and moor frog (*Rana arvalis*) can be found here. Since the discovery of roach in 2007, the population of amphibian larvae dwindled to a minimum due to roach predation (Kooij & Redford, 2012), and lack of recruitment threatened the long-term survival of the amphibians. For the first time in Norway, a rotenone treatment was conducted to benefit endangered amphibians. The lake is small, only 2,880 m^3, and rotenone treatment was performed in September 2009. Two people completed the treatment in one day. The target dose was 1.5 ppm CFT L, and a total of 4.5 l of CFT L was used. No mortality of amphibians was observed during treatment. The following spring, newts and frogs reproduced in high numbers (Kooij & Redford, 2012). Eradication of introduced fish in amphibian habitats can be done effectively with rotenone with apparently few negative effects on the amphibian population. It is recommended that treatments be carried out in the autumn when most adult amphibians and metamorphosed larvae have left their breeding habitat and water temperatures are still high enough for rapid rotenone degradation. No roach were detected in biodiversity surveys after the treatment.

Pike (*Esox lucius*) in the Telemark Canal

The Telemark Canal connects the coast of Telemark County with the inland by means of eight sluice stations on a stretch of 105 km. The pike is indigenous to south-eastern Norway but is alien to the Telemark region. Pike were released in the lower parts of the watercourse about 200 years ago. Pike are a voracious predator with the potential to severely decimate indigenous populations of fish. Over the last century, pike have spread upstream. Further upstream are large lakes with populations of large brown trout and river pearl mussel (*Margaritafera margaritafera*) that could be severely affected by invasive pike. Pike were found between Kjeldal and Hogga sluice, Hogga being the critical last sluice before entering the large lake system. This led to permission for rotenone treatment between Hogga and Kjeldal sluice, and the building of an electric fish barrier in the side canal leading up to Kjeldal sluice to prevent pike from being sluiced upstream together with boat traffic. The goal was to stop the pike at Kjeldal sluice, creating a pike free buffer zone up to Hogga sluice. A rotenone treatment was carried out between Hogga and Kjeldal sluice in October 2011, a stretch of about 1 km, to eradicate pike and restore the buffer zone. The river segment between the sluices was drained, and five people treated the remaining pools in one day. The target dose was 1 ppm CFT L, and 100 l of CFT L was used. An electric fish barrier was established in 2012, at the side canal leading up to Kjeldal sluice, to stop further spread.

The Norwegian Veterinary Institute conducted a new treatment at Kjeldal sluice in April 2014, this time only in the side sluice canal downstream of the area treated in 2011. The electric fish barrier in the side canal had been shut down during the winter season in 2013 due to maintenance work. Therefore, it was necessary to prevent the pike that had passed the non-functional electric barrier during winter from entering the previously treated area upstream of Kjeldal sluice before the boat sluices were opened for the season start. The side canal was 220 m long, 3 m deep and 17 m wide. One person did the job in one day, and a total of 22 l of CFT L was used. The target dose was increased to 1.5 ppm CFT L to compensate for water leaking through the sluice gates. Netting has been conducted in the rotenone-treated areas over several years, and no pike have been found.

Roach in Lake Vikerauntjønna

Lake Vikerauntjønna is situated in Trondheim municipality in Sør-Trøndelag County. The roach is indigenous to south-eastern Norway, but alien to this region. It is believed that roach had been released, and the source was other lakes with an alien population of roach in the same municipality, in Bymarka. A dense population of roach was detected in Lake Vikerauntjønna in 2013. It is located only 250 m from Trondheim municipality's main potable water source, Lake Jonsvatnet. There was a concern that roach could adversely affect water quality. The two lakes belong to separate catchments, but the risk of further spread to Lake Jonsvatnet was considered to be high due to the small distance between the lakes. Rotenone treatment was considered as the only measure that would eradicate the roach. Lake Vikerauntjønna covers an area of 0.04 km^2 with a water volume of 188,000 m^3 and maximum depth of 17 m. Treatment was conducted in September 2014 by five people in one day. The target dose was 1.5 ppm CFT L, and a total of 293 l of CFT L was used. Fish scale analysis revealed that the roach had been introduced for the first time around 2007, and possibly new introductions in following years too. No roach were detected in biodiversity surveys after the treatment.

Lake trout (*Salvelinus namaycush*) in Lake Klokkartjønna

Lake Klokkartjønna is situated in Blåfjella-Skjækerfjella National Park in Nord-Trøndelag County. The lake trout is indigenous to North-America but has been imported to Scandinavia for fish farming and angling purposes. The first records of release in Norway are from the 1970s (Hesthagen & Sandlund, 2016). In Lake Klokkartjønna the introduction could have come from source populations in Sweden, since lake trout are more common across the border, but no one knows for sure. A first finding of lake trout in Lake Klokkartjønna was recorded in the autumn of 2010. Lake trout are considered to be a threat to natural habitats, ecosystems, and indigenous fish populations. The risk of spread downstream to adjacent lakes was considered high, and permission for rotenone treatment was granted. Lake Klokkartjønna covers an area of 0.14 km^2 with an estimated volume of 675,000 m^3. Eradication was performed in July 2015, and eight people participated over two days. The target dose was 1 ppm CFT L, and a total of 670 l of CFT L was used. No lake trout have been found through post-treatment netting.

Several species at hydroelectric power plant in Gäddede

Gäddede hydroelectric power plant is situated in Sweden in Strømsund municipality in Jämtland County close to the Norwegian border. The power plant separates two lakes with different fish species due to a natural fish barrier. The upstream lake contained only indigenous brown trout and Arctic char whilst the lower lake also hosted pike, common whitefish, burbot (*Lota lota*), and rainbow trout. It was unwanted for any of these fish species to be spread upwards in the waterway. It is not possible for fish to pass upstream through the power plant, but a planned maintenance shutdown in 2016 could enable fish to pass the turbines and later rise up above the dam into the upper lake. As a precautionary measure, permission for rotenone treatment in the stagnant ponds on both sides of hydro power turbines was given. The volumes of the ponds ranged from 150 to 2,500 m^3, and eradication was performed in June 2016. Two people did the job in one day. A high dose of 3 ppm CFT L was used to compensate for fresh water leaking into the isolated ponds. A total of 8 l CFT L was used. There has been no programme for eradication confirmation. Dead fish were found during treatment in the ponds.

Roach in seven lakes in Bymarka

Bymarka is on the outskirts of Trondheim city in Sør-Trøndelag County. The roach is an invasive species in the region and was released in the 1880s to three small lakes in the same watercourse. From the 1960s to the 1980s roach were spread to three neighbouring lakes, and were found in another four lakes from 1998 to 2013. It was suspected that the roach population in Bymarka was the source of spread. When roach were found in Lake Vikerauntjønna, close to the Trondheim municipality's main potable water source, plans for treatment of the lakes in Bymarka were put forward. The main reasons were a concern for the roach to adversely affect potable water quality, a wish to permanently eradicate this blacklisted species from the region, and to contribute to conservation of natural fish stocks and biodiversity. Rotenone treatment was considered to be the only measure that could eradicate roach from these lakes. Several of the lakes have a dam, and an attempt to eradicate roach through dewatering in 2004 failed. One lake was 17,000 m^3 and 10 m deep, while the six other lakes ranged from 412,000 to 615,000 m^3, with maximum depths of 10–17 m. In September 2016, treatment was performed by a crew of 14 people for four days. A total of 4,000 l of CFT L was used and, as before, the target dose was 1.5 ppm CFT L. Populations of invasive pike in the lakes were eradicated simultaneously. No roach were detected in a biodiversity survey after the treatment.

DISCUSSION

Rotenone treatments are not without controversy, but most times invasive fish eradications are welcomed. The general public's acceptance of rotenone treatments in Norway might be a result of the absence of failure, thus strengthening the understanding for rotenone as a necessary and effective tool in the fight against invasive freshwater fish. The description of rotenone treatments does not include method, but relevant method can be found in Sandodden, et al. (2018). A standard operating procedure for the use of rotenone in fish management is given by Finlayson, et al. (2010b). For rotenone analysis, an on-site determination of rotenone has been developed by the Norwegian Veterinary Institute (Sandvik, et al., 2018).

The Norwegian Environment Agency is in the process of writing an action plan, which will identify and prioritise measures against invasive freshwater fish. This will lift the coordination of possible eradication measures from county level to national level, making top prioritised measures easier to identify. Forthcoming eradication projects are mostly for domestic invasive pike. The *G. salaris* eradication campaign will continue, and is now at an intermediate planning stage with the next eradication, at the earliest, in 2022.

Costs of treatments

Costs are not easy to describe uniformly. Eradication projects have had different levels of participation from the County Governor, and work hours are usually the main expense in these smaller eradications. The cost consists of the Norwegian Veterinary Institute's planning and preparations and expenses with treatments, including hired crew. The cost does not include pre- and post-treatment biodiversity surveys, cost of CFT L, and County Manager expenses. Eradication of *G. salaris* is not included in the table, but the cost of the eradication campaign so far is estimated to be about £100 million.

Eradication confirmation

The *G. salaris* eradication campaign includes a surveillance programme for eradication confirmation, but no parallel surveillance programme exists for invasive freshwater fish. Eradication confirmations are based on the absence of new detections by biodiversity surveys, local netting and angling. Successful restocking of indigenous fish also indicates the absence of the introduced species. Net trapping and environmental DNA surveys are other possible ways to document the outcome of a treatment but there is, at present, no national set of rules for eradication confirmation. However, there are no examples from Norway, during the past 20 years, of failed rotenone-based eradication attempts against invasive freshwater fish.

This may be because all eradications are assigned for planning and execution to a national competence group for rotenone treatment, which gives continuity-based experience and knowledge. Secondly, smaller lentic waters are less complicated treatments due to longer time for adequate mixing of rotenone and thus ensuring lethal concentration in all parts of the lake, which should leave the target fish no opportunities to accidently avoid lethal exposure. Large-scale lotic waters systems are also possible to succeed in, proven by the *G. salaris* eradication campaign.

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REFERENCES

Anon. (2016). *Status for Norske Laksebestander i 2016*. (Status of Norwegian Salmon Stocks in 2016). Rapport fra Vitenskapelig råd for lakseforvaltning no. 9. Trondheim: Vitenskapelig Råd for Lakseforvaltning.

Britton, J.R., Davies, G.D. and Brazier, M. (2010). 'Towards the successful control of the invasive *Pseudorasbora parva* in the UK'. *Biological Invasions* 12: 125.

Dalu, T., Wasserman, R.J., Jordaan, M., Froneman, W.P. and Weyl, O.L.F. (2015). 'An assessment of the effect of rotenone on selected non-target aquatic fauna'. *PLOS ONE* 10(11): e0142140.

Fernandez-Delgado, C. (2009). *Restoration of a Waterfowl Community by Eradication of the Invasive Fish Species Cyprinus carpio, with Rotenone*. Aphanius, investigacion peces. Cordoba (Spain): University of Cordoba.

Fjellheim, A. (2004). *Effekt av Rotenonbehandling på Bunndyrsamfunnene i et Område ved Stigstu, Hardangervidda*. (Effect of rotenone treatment on benthic invertebrates in an area at Stigstu, Hardangervidda). Rapport no. 122. Bergen: LFI-Unifob, Universitetet i Bergen.

Finlayson, B., Somer, W.L. and Vinson, M.R. (2010a). 'Rotenone toxicity to rainbow trout and several mountain stream insects'. *North American Journal of Fisheries Management* 30: 102–111.

Finlayson, B., Schnick, R., Skaar, D., Anderson, J., Demong, L., Duffield, D., Horton, W. and Steinkjer, J. (2010b). *Planning and Standard Operating Procedures for the Use of Rotenone in Fish Management—Rotenone SOP Manual*. Bethesda, Maryland: American Fisheries Society.

Fossøy, F., Dahle, S., Birkeland Eriksen, L., Hagen Spets, M., Karlsson, S. and Hesthagen, T. (2017). *Bruk av Miljø-DNA for Overvåking av Fremmede Fiskearter – Utvikling av Artsspesifikke Markører for Gjedde, Mort og Ørekyt*. (Use of environmental DNA for monitoring invasive species – development of species-specific markers for pike, roach and minnow). Rapport 1299. Trondheim: Norsk Institutt for Naturforskning.

Gozlan, R.E., Britton, J.R., Cowx, I. and Copp, G.H. (2010). 'Current knowledge on non-native freshwater fish introductions'. *Journal of Fish Biology* 76: 751–786.

Kooij, J van der and Redford K., (2012). *Rotenonbehandling av Amfibietjern på Holumskog. Overvåking av Dyrelivet*. (Rotenone treatment of an amphibian tarn at Holumskog. Monitoring of fauna). Nittedal: Nittedal Kommune.

Hesthagen, H. and Sandlund, O.T. (2016). *Tiltaksrettet Kartlegging og Overvåking av Fremmed Ferskvannsfisk – en Tilstandsvurdering av Spredningen pr. 2016*. (Targeted mapping and surveillance of foreign freshwater fish – an assessment of the spread per 2016). Rapport 1302. Trondheim: Norsk Institutt for Naturforskning.

Huitfeldt-Kaas, H. (1918). *Ferskvannsfiskenes Utbredelse og Indvandring i Norge med et Tillæg om Krebsen*. (Spread and immigration of freshwater fishes in Norway with an addition on crayfish). Kristiania: Centraltrykkeriet.

Johnsen, B. O. and Jensen, A. J. (1991). 'The *Gyrodactylus* story in Norway'. *Aquaculture* 98: 289–302.

Kjærstad, G., Arnekleiv, J. V. and Speed, J. D. M. (2015). 'Effects of three consecutive rotenone treatments on the benthic macroinvertebrate fauna of the River Ogna, Central Norway'. *River Research and Applications* 32: 572–582.

Ling, N. (2003). 'Rotenone: A Review of its Toxicity for Fisheries management'., *Science for Conservation* 211, Wellington, New Zealand: Department of Conservation.

McClay, W. (2000). 'Rotenone use in North America (1988–1997)'. *Fisheries* 25: 15–21.

O'Reilly, P. and Doyle, R. (2007). 'Live Gene Banking of Endangered Populations of Atlantic Salmon'. In: E. Verspoor, L. Stradmeyer and J.L. Neilsen (eds.) *The Atlantic Salmon. Genetics, Conservation and Management*, pp. 425–469. Oxford, UK: Blackwell Publishing.

Sandodden, R., Brazier, M., Sandvik, M., Moen, A., Wist, A.N. and Adolfsen, P. (2018). 'Eradication of *Gyrodactylus salaris* infested Atlantic salmon (*Salmo salar*) in the Rauma River, Norway, using rotenone'. *Management of Biological Invasions* 9(1): 67–77.

Sandvik, M., Waaler, T., Rundberget, T., Adolfsen, P., Bardal, H. and Sandodden, R. (2018). 'Fast and accurate on-site determination of rotenone in water during fish control treatments using liquid chromatography'. *Management of Biological Invasions* 9(1): 59–65.

Soleng, A. and Bakke, T.A. (1997). 'Salinity tolerance of *Gyrodactylus salaris* (Platyhelminthes, Monogenea): Laboratory studies'. *Canadian Journal of Fisheries and Aquatic Sciences* 54(8): 1837–1864.

Stensli, J.H. and Bardal, H. (2014). 'Bekjempelse av *Gyrodactylus salaris* i Vefsnaregionen'. (Eradication of Gyrodactylus salaris in the Vefsna region). *Veterinærinstituttets rapportserie* 2-2014. Oslo: Veterinærinstituttet.

Tønset, K. and Bakkeli, G. (2000). *Bekjempelse av ørekyte (Phoxinus phoxinus) i Stigstuvområdet i perioden 19.09–25.09.2000*. (Eradication of minnow (*Phoxinus phoxinus*) in the Stigstuv area in the period 19.09–25.09.2000 ') Rapport Desember 2000. Trondheim: Veterinærmedisinsk Oppdragssenter.

Vinson, M.R., Dinger, E.C. and Vinson, D.K. (2010). 'Piscicides and invertebrates: After 70 years, does anyone really know?' *Fisheries* 35: 61–71

First report of marine alien species in mainland Ecuador: threats of invasion in rocky shores

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Abstract Invasive species are of significant concern, especially in mega-diverse countries, because they cause negative effects such as loss of native biodiversity, ecological alterations, disease spread, and impacts on economic development and human health. In mainland Ecuador, information on invasive invertebrates in marine ecosystems is scarce. The objective of this study was to describe and locate the invasive species present in the rocky shores of the intertidal and subtidal zones along 10 areas (83 sites) covering most of the Ecuadorian coast during 2015–2016. Benthic macroinvertebrates communities were measured over quadrats located randomly on a 50 m transect positioned parallel to the coast in the intertidal and subtidal zone, covering an area of 1,860 km^2. Six invasive species were recorded: Arthropoda *(Amphibalanus amphitrite),* Cnidaria *(Pennaria disticha, Carijoa riisei*), Bryozoa (*Bugula neritin*a), Rhodophyta (*Asparagopsis taxiformis*) and Chlorophyta (*Caulerpa racemosa*). The areas with highest abundance of invasive species were in Jama (not a protected area), Marine and Costal Wildlife Reserve Puntilla of Santa Elena and Santa Clara Island Wildlife Refuge (protected areas). The most abundant species was *Carijoa riisei* with a relative abundance of up to 80%. It was the most aggressive of the invasive species registered in the subtidal zone, mainly in northern centre of the Ecuadorian coast. *C. riisei* is growing on native coral (*Pocillopora* spp.*)* and on sessile macroinvertebrate communities (*Pinctada mazatlanica*, *Muricea appresa* and *Aplysina* sp.) that are being affected by its invasion. This study must be taken into account by local and regional government authorities to create public policy programmes of monitoring for surveillance and control of invasive species. These programmes should focus on integration of socio-economic and ecological effects. They should be complemented by experimental design and analysis of environmental variables to provide technical information for a baseline of bio-invasion analysis along the Ecuadorian coast and Galápagos, to avoid the expansion of invasive species negatively affecting the marine biodiversity of mega-diverse countries such as Ecuador and other countries of South America.

Keywords: *Carijoa riisei*, continental Ecuador, Galapagos Islands, invasive species, macroinvertebrates, marine ecosystems, marine protected areas

INTRODUCTION

Invasive species are a cause of worldwide concern especially in mega-diverse countries because they can cause loss of native diversity, ecological alterations, increases in pests, diseases (Prenter, et al., 2004), impacts on benthic communities, impacts to the water column (Darrigran & Damborenea, 2011). Additionally, they can affect economic development and human health (Lowe, et al., 2000; Pimentel, et al., 2005). Many species are transported accidentally through anthropogenic means breaking geographic barriers that once restricted their range of expansion (Schüttler & Karez, 2008); they invade new areas, where they can settle, reproduce, spread and compete with native species.

Biological invasions, along with climate change, are key processes that feedback and affect global biodiversity. Climate change facilitates the dispersal and establishment of species which aggravates their impacts and makes their control more difficult, while invasive species can influence the magnitude of the environmental impacts by altering the structure and function of ecosystems (Mendoza, et al., 2014).

At present, there are numerous global and regional initiatives dedicated to optimising information and management of invasive alien species, including the Global Invasive Species Program (GISP), the IUCN-ISSG Invasive Species Global Information Network on Invasive Alien Species (GISIN), the Global Invasive Species programme of The Nature Conservancy (TNC-GISI) and the Inter-American Invasive Species Information Network (IABIN-I3N) (Schüttler & Karez, 2008).

On mainland Ecuador, information on invasive invertebrate species on intertidal rocky shores and subtidal zones is limited, fragmented and scattered. However, research on non-native species conducted in the Galapagos Islands (1,000 km off the coast of mainland Ecuador) has increased in the last decade, both in the terrestrial and marine environments (Campbell, et al., 2015). In 2012, the Charles Darwin Foundation (CDF), in collaboration with the Galapagos National Park Directory (GNPD), the Galapagos Biosecurity Agency (ABG), the Ecuadorian Navy and the Ecuadorian Navy Oceanographic Institute (INOCAR), initiated the Marine Invasive Species Project in the Galapagos Marine Reserve (Keith, et al., 2015).

The study of non-native species in Ecuador has mainly been done in the Galápagos Islands, due to the importance of this unique ecosystem in the world and the relative lack of scientific funding on the mainland. In the Galápagos Marine Reserve (GMR), an initial baseline study produced a list of seven non-native species in the GMR (Keith, et al., 2016). The marine invasive species team of the CDF have continued the research and applied different methodologies to learn more about non-native species in the GMR and

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the Eastern Tropical Pacific (ETP) region (I. Keith, pers. comm.). The objective of this study was to identify invasive species located in rocky shore habitats of the intertidal and subtidal zones covering 1,860 km^2 of the Ecuadorian coast during 2015–2016, that could be considered as threats for Ecuadorian mainland as well as Galapagos vulnerable ecosystems.

MATERIALS AND METHODS

Study area

Fieldwork was carried out in 10 areas along the Ecuadorian coast in six protected coastal marine areas (acronym in Spanish: AMCP) and four non-protected areas. The study areas ranged from Playa Escondida (0°49'8.05" N, 80° 0'22.66" W) in the north of Ecuador in Esmeralda province to Santa Clara Island in the south of Ecuador (3°11'21.11" S, 80°27'10.21" W) in El Oro province, covering 1,478 km^2 of protected areas and 382 km^2 of additional areas on the mainland coast. This survey included the protected areas (Fig. 1) from the north of Ecuador in the Galeras San Francisco Marine Reserve (acronym in Spanish: RMGSF) (Esmeralda province); Wildlife Refuge and Marine Coastal Pacoche (Pacoche) and Machalilla National Park (acronym in Spanish: PNM) (Manabí province); El Pelado Marine Reserve (acronym in Spanish: REMAPE); and Wildlife Coastal Marine Reserve Puntilla of Santa Elena (acronym in Spanish: REMACOPSE) (Santa Elena province) to Santa Clara Island Wildlife Refuge (Santa Clara) (El Oro province). The non-protected areas (Fig. 1) were: Jama, Canoa (Manabí province), Ayampe-La Entrada (between Manabí and Santa Elena provinces) and Copé (Santa Elena province). The Ecuadorian coast has an extension of 2,900 km corresponding to 45% of open coastal and 55% of inner coastal waters (Ayón, 1988). There is a wide range of geological features along the coast, including bluffs, barriers and sandplains, estuaries and lagoons, and engineered shoreline structures (Boothroyd, et al., 1994).

The climate on the coast varies seasonally from dry season (May to November) to the rainy season (December to April). The average annual temperature is above 22°C, with maxima fluctuating between 32–38°C and minima fluctuating around 15°C (Sonnenholzner, et al., 2013). Ecuador belongs to the Tropical East Pacific (TEP) region, with two sub-regions known as Panama Bight Ecoregion and Guayaquil Ecoregion (Sullivan & Bustamante, 1999; Miloslavich, et al., 2011). The northern half of the Ecuadorian mainland coast corresponding to the Panama Bight Ecoregion extends from Azuero Peninsula of Panamá to Caráquez Bay. It is characterised as a tropical zone, covered mostly by mangroves and dense rainforest vegetation (Miloslavich, et al., 2011), with >2,000 mm/yr of rainfall and without ecologically dry months through the year (Sonnenholzner, et al., 2013). The southern Ecuadorian coast, falling within the Guayaquil Ecoregion, extends from Caráquez Bay to Illescas Peninsula in the north of Perú and is characterised by a drier climate with <100 mm/yr of rainfall (Miloslavich, et al., 2011).

Survey

A total of 83 sites were sampled from February 2015 to February 2016 along the four coastal provinces of Ecuador (Esmeraldas, Manabí, Santa Elena and El Oro). These sites were established considering aspects such as representativeness of ecosystems; areas with greater and lesser anthropogenic intervention, biological processes (reproduction hotspots, feeding areas, seabirds and sea turtle nesting sites); and sensitive habitats or areas of great ecological importance according to the requirements established in the terms of reference of the Environmental Ministry (Ministry of Environment, 2014). The composition and abundance of the macroinvertebrates present in the rocky shore in the intertidal and subtidal zones were quantified using a band-transect system parallel to the coastline.

Data were collected in the intertidal zone following the standardised protocol from the South American Research Group on Coastal Ecosystems (SARCE) (SARCE, 2012). At each site, 10 quadrats (50 × 50 cm each) were randomly placed and sampled along a 50 m transect positioned parallel to the waterline in the mid, low and high intertidal level. A total of thirty replicates was sampled for each site. The abundance of colonial organisms was estimated by percent cover and all mobile individuals (>2 cm long) were counted. Most identification of biota was done in the field, although occasional problematic specimens were collected for reference and sent to specialists for identification. For the subtidal zone, at each site the organisms were separately estimated in two transect blocks by a diver, one on each side of 50 m transect line set along a shallow depth (normally 6–8 m). Every transect block encompassed a total reef area of 50 m × 5 m. The next diver scanned the nearby transect block by swimming back parallel to the initial transect at a distance of 5 m from the transect line (Edgard, et al., 2011). This up and back procedure for two adjacent blocks was

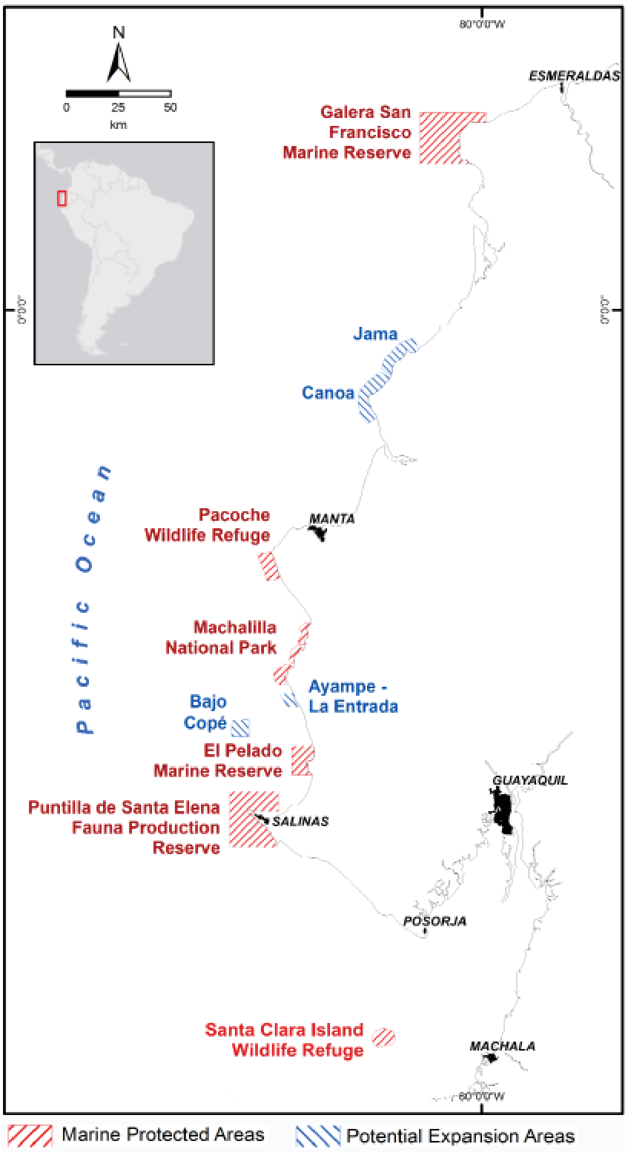

Fig. 1 Study area and location of the sampling sites on the Ecuadorian coast.

repeated along the shallower depth contour, generating a duplicate transect block data at each site. Sessile organisms were estimated by percent coverage of different taxa and grouped in substratum classes (crustaceans, cnidarians, sponges, ascidians, bryozoans, hydroids) within the transect lines. The cover was generally recorded by divers within 10 quadrats (0.5 × 0.5 m), placed sequentially every 5 m along the 50 m transect, and mobile organisms were counted along each quadrat. Digital photo quadrats were taken during the field work. We summed counts across all quadrats to create site totals.

Data analysis

To explain the biological assemblage, an X sites by Y species matrix of abundances was built to perform a non-metric multidimensional scaling (n-MDS) and cluster analysis to visualise the similarity of studied areas. For both analyses a similarity matrix was generated using the Bray-Curtis index on the fourth root transformed data to remove the weight of the dominant species (Clarke & Gorley, 2006). Further bubbled MDS analyses were performed to visually establish the differences among the abundances of *Carijoa riisei* between zones, using the statistical package Plymouth Routines in Multivariate Ecological Research (Primer). In order to determine the difference of organism abundance by province, a nonparametric ANOVA was performed, after the assumptions were not fulfilled, using the Kruskal Wallis test. In addition, to determine the difference in abundances between protected and unprotected zones, we applied the ANOSIM test using PRIMER V6 (Clarke & Gorley, 2006).

Distribution maps of species were prepared using the information collected from the fieldwork. These maps represent the relative abundance with a percent of coverage/m^2 on every site for inter-tidal and subtidal zones, with scales ranging between 0.1–50%, indicating a spatial approximation of alien invasive species location and coverage. Besides the status, invasiveness of each species was established using international databases such as: the IUCN list of 100 most harmful invasive alien species in the world (Lowe, et al., 2000); Global Invasive Species Database, ISSG (IUCN/SSC, 2014); and Invasive Species Compendium (<www.cabi.org/isc>).

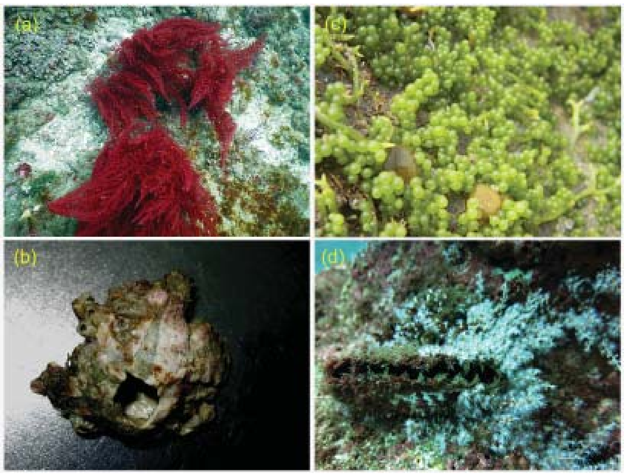

Fig. 2 Alien species found in the survey: a) *Asparagopsis taxiformis,* b) *Amphibalanus amphitrite* c) *Caulerpa racemosa*, and d) *Carijoa riisei* growing on the bivalve *Pinctada mazatlanica*.

RESULTS

A total of six alien invasive species from five phyla were recorded: Cnidaria (*Pennaria disticha*, *Carijoa riisei*), Bryozoa (*Bugula neritina)*, Arthropoda (*Amphibalanus amphitrite),* Rhodophyta (*Asparagopsis taxiformis*) and Chlorophyta (*Caulerpa racemosa*). Assemblages were numerically dominated by cnidarians. The most abundant species was *Carijoa riisei* (Table 1; Fig. 2). Invasive species were recorded at 24 sites (14 sites in the subtidal zone and ten in the intertidal zone). In the subtidal zone, the area with the highest presence of invasive species was the RMGSF in the north of Ecuador (Esmeralda province) while in the intertidal zone it was Punta Carnero site in the REMACOPSE, south-central part of the coast in Santa Elena province· (Table 1; Fig. 3).

The n-MDS of invertebrate invasive species abundance showed four groups with major similarity (60%), one group formed by Jama, REMAPE and RMGSF, the second group clustered the sites of REMAPE (south-central coast); the third group formed REMACOPSE, Ayampe, Copé and Santa Clara (central and south-central coast) and the last one grouped by REMACOPSE, Pacoche, Santa Clara, Ayampe and Canoa (Fig. 4).

Amphibalanus amphitrite, Pennaria distincha and *Carijoa riisei* were the invasive species with greatest

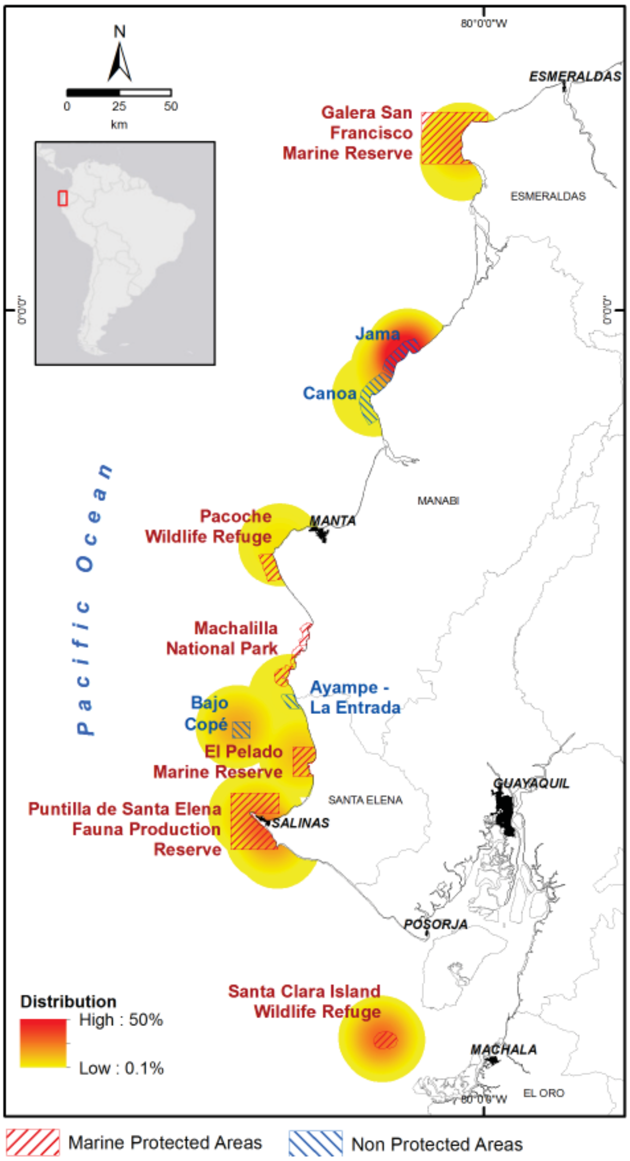

Fig. 3 Relative abundance and distribution of invasive species along the Ecuadorian coast during 2015–2016.

occurrence. Of these three, C. *riisei* was most abundant (Fig. 5) in the non-protected area located in the central coast of Ecuador (Jama) (Table 1). However, it was also recorded in the north zone of the Galera San Francisco Marine Reserve (Punta Alta, Piedra de Quingue) and in the south-central coast at El Pelado Marine Reserve (La Pared).

Statistically, no significant differences were found between the abundance of invasive species by provinces (global R=0.08, p>0.001) or by protected and unprotected zones (global R=-0.06, p>0.001).

The MAP's that presented the greatest number of invasive species were REMACOPSE (four species) and Ayampe (three species), followed by Jama and REMAPE (less than three species. Galeras San Francisco, Canoa, Pacoche and Copé recorded low benthic numbers of invasive species (Table 1).

DISCUSSION

This is the first report investigating the presence of invasive species along the Ecuadorian coast, including marine protected areas and unprotected areas, covering the coast from north to south of the country and two ecoregions in four distinct provinces. There are four species classified as macroinvertebrate invasive species worldwide, of which the majority are the cnidarians, mainly the Anthozoa class. Although the invasive species recorded are not listed in the 100 world's worst invasive alien species according to IUCN (Lowe, et al., 2000), two species (*Carijoa riisei* and *Bugula neritina,*) are listed in the Global Invasive Species Database (ISSG) and four species (*Carijoa riisei*, *Bugula neritina, Pennaria disticha* and *Amphibalanus amphitrite)* are registered by the Global Register of Introduced and Invasive Species (GRIIS).

Table 1 Invasive species recorded by provinces, areas and sites on the Ecuadorian coast, including abundance (coverage percentage) in the subtidal and intertidal zones.

PROVINCE / Area	Sites	*Amphibalanus amphitrite*	*Caulerpa racemosa*	*Aspargopsis taxiformis*	*Pennaria distincha*	*Carijoa riisei*	*Bugula neritina*
ESMERALDAS							
Galeras San Francisco Marine Reserve	Punta Alta	-	-	-	-	1.37	-
	Piedra de Quingue	-	-	-	-	11.5	-
MANABÍ							
Jama	Vaca Brava 1	-	-	-	-	20.25	-
	Punta Ballena*	0.11	-	-	-	-	-
	Bajo Londres	-	-	-	-	44.57	-
Canoa	Cabo Pasado*	-	0.96	-	-	-	-
Wildlife Refuge and Marine Coastal Pacoche	Liguiqui*	3.28	-	-	-	-	-
Ayampe – La Entrada	Los Ahorcados 1	-	-	-	0.49	-	0.12
	La Entrada*	0.03	-	-	-	-	-
Bajo Copé	Seco Manta	-	-	-	4.68	-	-
	Bajo Fer 3	-	-	-	8.86	-	-
SANTA ELENA							
El Pelado Marine Reserve (REMAPE)	La Pared	-	-	-	-	4.44	-
	Bajo 40	-	-	0.37	-	-	-
	Corales	-	-	4.82	-	-	-
Puntilla de Santa Elena Marine and Coastal Wildlife Reserve (REMACOPSE)	Guarro	-	-	-	1.12	-	-
	Bajo Ballena	-	-	-	0.25	-	5.31
	Chocolatera*	0.11	0.03	-	-	-	0.17
	Loberia*	7.8	0.22	-	-	-	-
	Punta Carnero*	16.01	-	-	-	-	-
	Anconcito*	0.5	-	-	-	-	-
EL ORO							
Santa Clara Island Wildlife Refuge	Sur*	-	0.6	-	-	-	-
	Norte*	-	1.68	-	-	-	-
	Sitio 2	-	-	-	18.64	-	-
	Sitio 3	-	-	-	3.67	-	-

* Sites with results of intertidal zones.

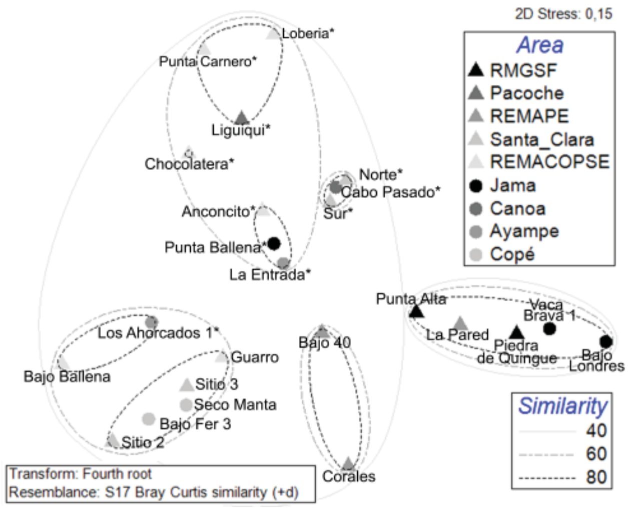

Fig. 4 Non-metric multidimensional scaling ordination, showing relative abundance of marine invasive species registered along the Ecuadorian coast during the period 2015–2016.

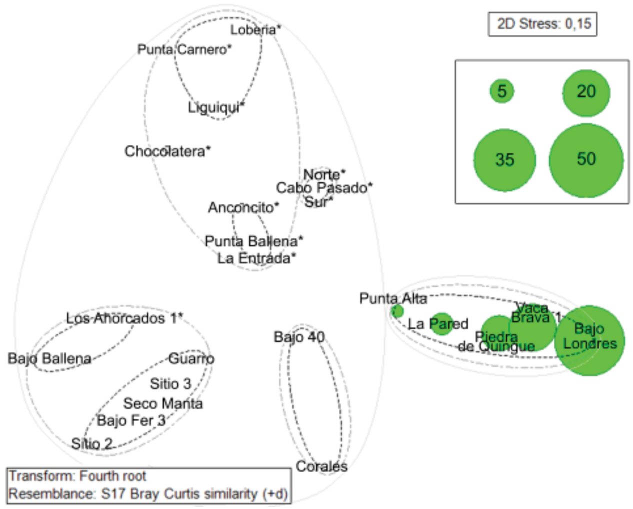

Fig. 5 2D bubble MDS configuration showing relative abundance of *Carijoa riisei*.

Carijoa riisei showed a greater abundance in the central zone of the Ecuadorian coast, mainly in Jama. This species has increased its colonisation in some areas of the El Pelado Marine Reserve in two years (2013–2015) (Cárdenas-Calle & Triviño, 2014). The invasion of *C. riisei* to new sites is probably caused by marine currents and maritime traffic. The invasive growth of *C. riisei* was noted among colonies of Pocilloporidae corals, *Pinctada mazatlanica, Muricea appresa* and *Aplysina* sp., confirming the imminent threat of this species to the sessile biota of the marine protected areas (Martínez, 2013). This species has an extensive geographic distribution in the Pacific from the Philippines, Indonesia, Australia, and Thailand, South Atlantic (Silva, et al., 2011) and Caribbean region (Kahng & Grigg, 2005; Kahng, et al., 2008;) with a variety of reproductive strategies (Barbosa, et al., 2014) including sexual and asexual reproduction, growing in different habitats, but preferring shallow areas.

Carijoa riisei has caused great impacts and damage to coral areas in Hawaii (Barbosa, et al., 2014) where it is currently considered a pest and has affected over 70% of the colonies of black corals *Antipathes dichotoma* and *A. grandis* (Global Invasive Species Database, 2017). It is considered a common invasive species from Florida (USA) to Santa Catarina (Brazil), displacing native species. It is now known to monopolise benthic surfaces under optimal conditions for its growth, from the intertidal zone to depths of >100 m (Venkataraman, et al., 2016). *C. riisei* competes successfully over black coral and invertebrates (Kahng & Grigg, 2005) and is dispersed through marine vectors (Grigg, 2003), and it is reported as a major biofouler in the Atlantic region (Concepcion, et al., 2010).

The rapid growth of the *C. riisei* colonies and their widespread dispersion in coral ecosystems has begun to generate great concern worldwide for being considered a threat to the diversity of sessile corals and invertebrates. For this reason, it is listed in the database of invasive species of IUCN (Global Invasive Species Database) and there is evidence of ecological impacts of this species in some countries of the South Pacific, as in Colombia, where high mortality of corals and octocoral coating has been reported on the island of Malpelo (Sánchez, et al., 2011).

Orensanz, et al. (2002) detected more than 40 invasive species in the Southern Atlantic Ocean, where poor knowledge of the regional biota makes it difficult to track invasions. For these reasons it is necessary to begin an alliance between national and international academics and environmental authorities (Ministry of Environment) in Ecuador to develop a strategy for surveillance and research on the ecological effects of invasive species in the coastal zones. With *Carijoa riisei* it is necessary to quantify mortality and replacement of existing coral communities in Ecuador, because this information is currently unknown, as is the habitat and biota preferences for colonisation. It is important to know its distribution, its ecological effects on native fauna, and its preferences (habitat, substrates, depths and environmental variables) to allow the establishment of substantial management actions to avoid its dispersion to other sensitive areas, such as the Galapagos Islands where it is still absent.

We found that the greatest abundance of invasive species was in the Ecuadorian central coast (Manabí), belonging mainly to the cnidarians. However, the largest diversity of species was in the south-central coast (Santa Elena). The presence of these invasive species is possibly due to the currents, ballast water and encrustations of invaders on ships. We can speculate that factors such as marine currents, rise of temperature, increase of maritime traffic, global warming and invasive breeding strategies will accelerate the augmentation of invasive alien species and the loss of diversity of corals, octocorals, sponges and other marine sessile invertebrates on the Ecuadorian coast. Four of the six non-native species found on the mainland of Ecuador (*Pennaria disticha, Bugula neritina, Asparagopsis taxiformis* and *Caulerpa racemosa*) from Table 1, are already present in the Galapagos Islands (Danulat & Edgard, 2002; Keith et al, 2016).

This study must be taken into consideration by local and regional government authorities to create public policies and programmes to monitor for surveillance and control of invasive species. These programmes have to be integrated with socio-economic and ecological effects and complemented by experimental design and analysis of environmental variables to provide technical information and a baseline of bio-invasions along the Ecuadorian coast and Galápagos. It is important to avoid or limit the expansion of invasive species that negatively affect the marine biodiversity of mega-diverse countries such as Ecuador and other countries of South America.

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REFERENCES

Ayón, H. (1988). *Grandes Rasgos Geomorfológicos de la Costa Ecuatoriana*. Ecuador: PMRC-Proyecto de Manejo de Recursos Costeros.

Barbosa, T.M., Gomes, P.B., Bergeron, A.S., Santos, A.M., Chagas, C., Freitas, E.M.S. and Perez, C.D. (2014). 'Comparisons of sexual reproduction in *Carijoa riisei* (Cnidaria, Alcyonacea) in South Atlantic, Caribbean, and Pacific areas'. *Hydrobiologia* 734: 201–212.

Boothroyd, J., Ayón, H., Robadue, D., Vásconez, J. and Noboa, R. (1994). *Características de la Línea Costera del Ecuador y Recomendaciones para su Manejo*. Technical Report 2076. Ecuador: PMRC-Proyecto de Manejo de Recursos Costeros (Atlas). <http://simce.ambiente.gob.ec/sites/default/files/documentos/anny/Caracter%C3%ADsticas%20de%20la%20l%C3%ADnea%20costera%20del%20Ecuador%20y%20recomendaciones%20para%20su%20manejo.pdf>.

Campbell, M.L., Keith I., Hewitt C.L., Dawson T.P. and Collins, K. (2015). 'Evolving marine biosecurity in the Galápagos Islands'. *Management of Biological Invasions* 6(3): 227–230.

Cárdenas, M. and Triviño, M. (2014). *Caracterización Bio-ecológica de Siete Sitios de Buceo Situados Alrededor del Islote El Pelado en Ayangue*. Guayaquil-Ecuador: Reporte Técnico. Corpei.

Clarke, K.R. and Gorley, R.N. (2006). *Primer v6: User Manual / Tutorial*. UK: Plymouth.

Concepcion, G.T., Kahng, S.E., Crepeau, M.W., Franklin, E.C., Coles, S.L. and Toonen, R.J. (2010). 'Resolving natural ranges and marine invasions in a globally distributed octocoral (genus *Carijoa*)'. *Marine Ecology Progress Series* 40: 113–127.

Danulat, E., and Edgar, G.J. (eds.) (2002). *Reserva Marina de Galápagos. Línea Base de la Biodiversidad*. Santa Cruz, Galápagos, Ecuador: Fundación Charles Darwin. Fundación Charles Darwin/Servicio Parque Nacional Galápagos. <http://www.darwinfo undation.org/files/library/pdf/RMG-Linea-Base-Bio. pdf>.

Darrigran, G. and C. Damborenea (2011). 'Ecosystem engineering impacts of *Limnoperna fortunei* in South America'. *Zoological Science* 28: 1–7.

Edgard, G.J., Banks, S., Bessudo, S., Guzman, H.M., Henderson, S., Martinez, P., Rivera, F., Soler, G., Ruiz, D. and Zapata, F. (2011). 'Variation in reef fish and invertebrate communities with level of protection from fishing across the Eastern Tropical Pacific seascape'. *Global Ecology and Biogeography* 20: 730–743.

Global Invasive Species Database. (2017). 'Species Profile: *Carijoa riisei*'. <http://www.iucngisd.org/gisd/speciesname/carijoa+riisei>. Accessed 12 March 2017.

Grigg, R.W. (2003). 'Invasion of a deep-water coral bed by an alien species, *Carijoa riisei' Coral Reef* 22: 121–122.

IUCN. (2017). 'The IUCN Red list of Threatened Species'. <www.iucnredlist.org>. Accessed 15 June 2017.

IUCN/SSC. (2014). 'Invasive Species Specialist Group. Global Invasive Species Database'. <http://www.iucngisd.org/gisd/>. Accessed 15 June 2017.

Kahng, S.E., Benayahu, Y. and Wagner, D.R.N. (2008). 'Sexual reproduction in the invasive octocoral *Carijoa riisei* in Hawaii'. *Bulletin of Marine Science* 82: 1–17.

Kahng, S.E. and Grigg, R.W. (2005). 'Impact of an alien octocoral, *Carijoa riisei*, on black corals in Hawaii'. *Coral Reef* 24: 556–562.

Keith, I., Dawson, T., Collins, K. and Bank, S. (2015). 'Especies Marinas Invasoras en la Reserva Marina de Galápagos: Un Caso para Investigación Adicional, Mejoramiento del Manejo y Revisión de Políticas'. In: *Informe Galápagos 2013–2014*, pp. 83–88. Puerto Ayora, Galápagos. Ecuador: DPNG, CGREG, FCD and GC.

Keith, I., Dawson, T., Collins, K. and Campbell, M.L. (2016). 'Marine invasive species: Establishing pathways, their presence and potential threats in the Galápagos Marine Reserve'. *Pacific Conservation Biology* 22: 377–385.

Lowe, S., Browne, M., Boudjelas, S. and De Poorter, M. (2000). '100 of the World's Worst Invasive Alien Species: A Selection from the Global Invasive Species Database'. The Invasive Species Specialist Group (ISSG) a specialist group of the Species Survival Commission (SSC) of the World Conservation Union (IUCN). <http://www.issg.org/pdf/publications/worst_100/english_100_worst.pdf>. Accessed 15 June 2017.

Martínez, P. (2013). *Potencial Amenaza de la Especie Invasora Carijoa riisei en Áreas Marinas Protegidas del Ecuador* (*Potential Threat of the Invasive Species Carijoa riisei in Ecuadorian Marine Protected Areas*). Unpublished report. Quito, Ecuador: Instituto Nazca de Investigaciones Marinas.

Mendoza, R., Born-Schmidt, G., March, I.J. and Álvarez, P. (2014). 'Especies Invasoras Acuáticas y Cambio climático'. In: R. Mendoza and P. Koleff (eds.) *Especies acuáticas invasoras en México*, pp. 469-495. México: Comisión Nacional para el Conocimiento y Uso de la Biodiversidad.

Miloslavich, P., Klein, E., Díaz, M.J., Hernández, C., Bigatti, G., Campos, L., Artigas, F., Castillo, J., Penchaszadeh, P.E., Neill, P.E., Carranza, A., Retana, M., Díaz de Astarloa, J.M., Lewis, M., Yorio, P., Piriz, M.L., Rodríguez, D., Yoneshigue-Valetin, Y., Gamboa, L. and Martín, A. (2011). 'Marine biodiversity in the Atlantic and Pacific coasts of South America: Knowledge and gaps'. *PLOS ONE* 6: 1–43.

Ministry of Environment. (2014). *Términos de Referencia de la Consultoría Inventarios Cuantitativos Submareales e Intermareales de Biodiversidad Marina en Seis Áreas Marino Costeras Protegidas y cuatro Zonas de Posible Expansión* (*Reference Terms of the Consultancy Submarine and Intertidal Quantitative Inventory of Marine Biodiversity in Six Marine Protected Areas and Four Zones of Possible Expansion*). Guayaquil: Ministry of Environment.

Orensanz, J., Schwindt, E., Pastorino, G., Bortulus, A., Casas, G., Darrigran, G., Elías, R., López Gappa, J.J., Obenat, S., Pascual, M., Penchaszadeh, P., Piriz, M.L., Scarabino, F., Spivak, E.D. and Vallarino, E.A. (2002). 'No longer a pristine confine of the world ocean–a survey of exotic marine species in the southwestern Atlantic'. *Biological Invasions* 4: 115–143.

Pimentel, D., Zúñiga, R. and Morrison, D. (2005). 'Update on the environmental and economic costs associated with alien invasive species in the United States'. *Ecological Economics* 52: 273–288.

Prenter, J., MacNeil, C., Dick, J.T.Z. and Dunn, A.M. (2004). 'Roles of parasites in animal invasions'. *Trends in Ecology and Evolution* 19: 385–390.

Sánchez, J.A., Gómez, C.E., Escobar, D. and Dueñas, L.F. (2011). 'Diversidad, abundancia y amenazas de los octocorales de la isla Malpelo, Pacífico Oriental Tropical, Colombia'. ('Diversity, abundance and threats to Malpelo Island´s octocorals, Tropical West Pacific, Colombia'). *Boletín de Investigaciones Marinas y Costeras – INVEMAR* 40: 139–154.

SARCE. (2012). *Protocolo y Diseño de Muestreo para la Evaluación de la Biodiversidad Marina en Suramérica* (*Protocol and Sampling Desing for the Evaluation of Marine Biodiversity in South America*). Caracas, Venezuela: South American Research Group on Coastal Ecosystems. https://www.oceanbestpractices.net/handle/11329/412?show=full

Schüttler, E. and Karez, C.S. (eds.) (2008). *Especies Exóticas Invasoras en las Reservas de la Biosfera de América Latina y el Caribe. Un Informe Técnico para Fomentar el Intercambio de Experiencias entre las Reservas de Biosfera y Promover el Manejo Efectivo de las Invasiones Biológicas*. (*Exotic Invasive Species of Biosphere Reserves of Latin American and the Caribbean. A Technical Report to Promote the Effective Management between Biosphere Reserves and to Promote the Effective Management of Biological Invasions*). Montevideo, Uruguay: UNESCO.

Silva, A.G., Lima, R.P., Gomes, A.N., Fleury, B.G. and Creed, J.C. (2011). 'Expansion of the invasive corals *Tubastraea coccinea* and *Tubastraea tagusensis* into the Tamoios Ecological Station Marine Protected Area, Brazil'. *Aquatic Invasions* 6, supplement 1: S105–S110.

Sonnenholzner, J., Brandt, M., Francisco, V., Hern, A., Luzuriaga, M., Guarderas, P. and Navarro, J.C. (2013). 'Echinoderms of Ecuador'. In: J.J. Alvarado and F.A. Solís-Marín (eds.) *Echinoderm Research and Diversity in Latin America*, pp. 183–233. Berlin Heidelberg: Springer-Verlag.

Sullivan, K.K. and Bustamante, G. (1999). *Setting Geographic Priorities for marine Conservation in Latin América and the Caribbean*. Arlington, Virginia: The Nature Conservancy.

Venkataraman, K., Raghunathan, C., Satyanarayana, C. and Rajkumar, R. (2016). 'Invasion of snowflake coral, *Carijoa riisei* (Duchassaing and Michelotti, 1860), in Indian seas: Threats to coral reef ecosystems'. *Indian Journal of Geo-Marine Sciences* 45: 1403–1408.

Lessons on effectiveness and long-term prevention from broad-scale control of invasive alien species in Scotland's rivers and lochs

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Abstract Prior to 2008 there were few invasive alien species (IAS) initiatives operating in Scotland on a scale required for effective control. The establishment of the Biosecurity and Invasive Non-Native Species Programme by the Rivers and Fisheries Trusts of Scotland was the first attempt to link local efforts with national IAS strategy on scales appropriate to the effective control of target species. The programme worked with 26 local fisheries trusts to produce biosecurity plans that covered over 90% of Scotland's rivers and lochs. The programme implemented a range of prevention measures, including promoting awareness of invasive species issues and the need for biosecurity among water users. Projects were established for invasive plants on most major river systems, and for American mink (*Neovison vison*) in the north of Scotland. These projects involved public/private partnerships, using a mix of professional staff and volunteers. Interactive data management systems were developed to manage input from a large number of individuals and to inform an adaptive-management approach. These control projects demonstrated that it is feasible to reduce the size and density of target populations of invasive species across large geographic areas. The key to maintaining the momentum of this control effort in the future will be to demonstrate sustainable IAS management in the longer term. This challenge led to the formulation of the Scottish Invasive Species Initiative (SISI) whose overall aim is the development of a long-term, cost-effective strategy for IAS management throughout the north of Scotland. SISI will test strategies derived from experience and information from previous control projects. Important areas that the initiative will seek to address include defining outcomes, integrating IAS management into other management initiatives, and maintaining partnership interest and cohesiveness in a challenging funding environment.

Keywords: adaptive management, biosecurity, community-based effort, giant hogweed, Himalayan balsam, IAS, Japanese knotweed, mink, rivers

INTRODUCTION

Scotland depends on the quality of its iconic natural environment for economic and recreational wealth. The high number of protected sites (1,868) and designated natural features (5,376) reflects the importance placed on natural heritage.

Watercourses are integral, defining features of Scotland's landscapes and culture. Historically, Scottish society relied on healthy rivers and lochs for food, recreation, transport and industry. Art, folklore and traditional activities have long drawn inspiration from them. Today, the economic reliance extends to whisky-distilling, salmon-farming, tourism and many new forms of recreation. In 2010, Scottish residents generated £2.3 billion from their visits outdoors (SNH, 2011). Recreational freshwater fishing is estimated to support around 4,300 jobs, contributing £79.9 million to the economy (Marine Scotland, 2017).

Within increasingly fragmented landscapes, water courses also function as corridors between habitats for biodiversity. This vital function is compromised by invasive alien species (IAS) (Also known as Invasive Non-Native Species (INNS) in the United Kingdom) for which rivers and lochs are excellent pathways into the broader natural environment. The margins and shorelines of watercourses themselves are among the most exposed to the risk of IAS spread and damage. Climate change, pollution and habitat disturbance accelerate rates of invasion, with corresponding costs for socio-economic, human and ecological well being (Forest Research, 2008; Williams, et al., 2010).

The UK and Scottish Governments have recognised the IAS threat. The Great Britain Invasive Non-Native Species Strategy (GBNNSS, 2008) is a policy and strategic response. The Scottish Environment Protection Agency (SEPA) addresses the threat through the INNS supplementary plan to the Scotland and Solway-Tweed River Basin management plans (SEPA, 2009a; SEPA, 2009b). Scottish Natural Heritage (SNH), a Scottish Government agency, has included IAS in its Species Action Framework (Raynor, et al., 2016).

Prior to high level recognition of this sort, the vast majority of reponses to IAS were small-scale and localised. Management on larger scales was confined to catchment-based control of invasive alien plant species (IAPS) on the River Tweed (Tweed Forum, 2006) and to control of American mink (*Neovison vison*) in the Cairngorms National Park and rural Aberdeenshire (Bryce, et al., 2011).

The scale of the threat, the likely severity of ecological, social and economic impacts and the prospect of rises in control and eradication costs have constituted a case for better, more strategic and systematic approaches to managing IAS. This paper reports on the results and lessons learnt between 2008–2017 from the work of 26 member organisations of the Rivers and Fisheries Trusts of Scotland (RAFTS) in partnership with government agencies and universities to address the IAS threat to Scotland's rivers and lochs. We will also refer to an ambitious project in which those lessons are incorporated to manage multiple IAS cost effectively in the long term over 29,500 km^2 of northern Scotland.

METHODS

Biosecurity planning

At the northern invasion front for high-impact IAS of the United Kingdom and Europe, Scotland was well placed to manage the threats strategically at national and local scales. On the national scale there was an opportunity to defend the IAS-free region to the north of the front, control IAS in the lightly infested catchments in northern and southern Scotland, before addressing the more impacted areas of central Scotland.

RAFTS and its 26 local Trust members created area-specific biosecurity plans in three phases between 2008 and 2010 (Fig. 1). All plans used a template designed by RAFTS in consultation with the Great Britain Non-Native Species Secretariat, Scottish Government, SNH and

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SEPA. The template linked key elements of IAS policy and strategy to local action and acted as a framework for universal consistency. Plan objectives reflected the three key elements of the Great Britain INNS Strategy (GBNNSS, 2008): (1) prevention, early detection, and surveillance; (2) monitoring and rapid response; and (3) mitigation, control and eradication. Objectives and actions were also linked to related plans and initiatives such as River Basin Management Planning (SEPA, 2009a; SEPA, 2009b). This approach translated the key elements of national policy and strategy into action across relevant sectors in ways which emphasised coordination and partnership.

Funding secured for a series of projects from 2009 to 2017 enabled local organisations to coordinate and monitor the control of invasive alien plant species and American mink by professionals and volunteers (Table 1). RAFTS provided the overall coordination, strategic direction and evaluation of the activities. SNH, SEPA, the University of Aberdeen (mink) and Queens University Belfast (plants) provided specific technical support. Principal target species were the IAPS, giant hogweed (*Heracleum mantegazzianum*), Japanese knotweed (*Fallopia japonica*) and Himalayan balsam (*Impatiens glandulifera*) and the alien invasive mustelid American mink – all recognised as high-impact species for waterbodies and/or biodiversity (UKTAG, 2015).

Engagement

Engagement of key stakeholders was critical given the scale of the work, the need to obtain permissions for access and the recruitment and maintenance of the volunteer workforce. Awareness campaigns, mailshots, presentations, meetings with local environment/community groups, schools and individuals, newsletters, the websites and media were means to initiate contact with potential volunteers. Working in public areas and approaching landowners for permission also proved effective in engaging local communities.

Once engaged, stakeholders were kept informed through websites, newsletters, media and meetings and later through interactive reporting systems. Participating organisations and individuals received skills-training for efficient, effective, legally compliant surveillance and control of IAS. Formal training courses were tailored to roles in the control strategy. Volunteers were also offered informal training if they were unavailable for formal courses.

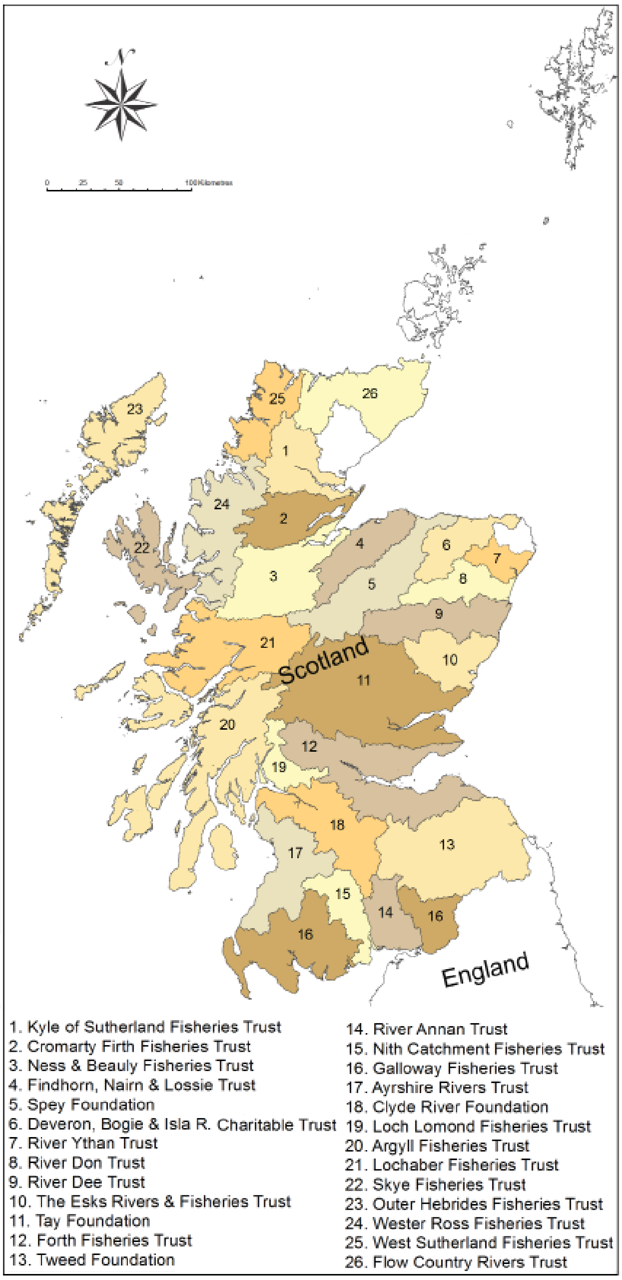

Fig. 1 Map of Trust areas.

Table 1 Summary of projects implemented through the RAFTS Biosecurity and Invasive Non-Native (Alien) Species Programme with duration, description of activities and geographic scope and participating local partner organisations.

Name of Project	Duration	Description	Participating Trusts
Pan Scotland Invasive Non Native Plant Species Control	2009–2016	A series of projects for the control and eradication of invasive alien riparian plant species in northern, southern and central Scotland. Included biosecurity, awareness and training of professional staff and volunteers.	Annan; Argyll, Ayrshire, Cromarty Firth; Deveron, Isla and Bogie; Don; Dee; Esk River; Forth; Findhorn, Nairn and Lossie; Galloway; Lochaber; Kyle of Sutherland; Tweed; West Sutherland
Scottish Mink Initiative (SMI)	2010–2015	Aiming to eradicate breeding mink from 20,000 km^2 (later extended to 28,000 km^2) of north and north-eastern Scotland. The Initiative also supports awareness and local capacity building activities as well as the development of local management models for future mink control.	Cromarty Firth; Deveron, Isla and Bogie; Dee; Don; Esk; Findhorn, Nairn and Lossie; Spey, Tay, Ythan
Controlling priority invasive non-native riparian plants and restoring native biodiversity (CIRB)	2010–2014	Control and eradication of invasive alien riparian plant species in 12 catchments in southern Scotland, piloting biosecurity, awareness activities, training of Trust staff and volunteers, best practice identification and dissemination.	Argyll; Ayrshire; Galloway; Tweed Forum

IAPS densities, distribution and control

Surveys of river and loch catchments identified the location, extent and abundance of IAPS. The distribution of IAPS populations were entered into a geo-database along with estimates of abundance based on the DAFOR scale (Kent & Coker, 1992). The impact of treatment was monitored by recording distribution and abundance post-treatment. Treatments varied by species but were primarily foliar leaf spray (Japanese knotweed and Himalayan balsam), stem injection (Japanese knotweed) and physical removal (Himalayan balsam).

Initially the majority of local Trusts took a 'top down' approach to control, starting at the upstream extent of IAPS distribution and working downstream. The rationale was the reduction of potential reinfestation of treated sites from upstream populations. Later, working from the lower to the upper catchment was adopted by some Trusts when treating whole catchments. This tactic recognised that plants lower in the catchment developed earlier than those in the upper regions.

Mink control

Volunteers and paid staff relied mainly on mink rafts to detect and trap American mink. Originally conceived by the Game and Wildlife Conservation Trust (GWCT) (Reynolds, et al., 2004), the mink raft is a floating platform on which a tunnel covers a clay pad. The raft is anchored to the bank of a waterway. American mink are predominantly active within 10 m of waterways (Yamaguchi, et al., 2003), are naturally attracted to tunnels and leave footprints in the clay when investigating them. Once a mink is detected, a live-capture cage-trap is inserted in the tunnel. Captured mink were despatched humanely. Carcasses were tagged and sent to Aberdeen University to determine sex, age and provenance based on genetic profile (Fraser, et al., 2013; Melero, et al., 2015; Ruiz-Suarez, et al., 2016).

Evaluation

Stakeholder engagement and impacts on IAS populations were evaluated in 2015 as measures of success. Data recorded for stakeholder engagement included contacts, background, and time spent. Assessment of mink control recorded raft locations and status, raft checks, mink sightings and captures. The locations and extent of target IAPS were recorded using geographic positioning systems and abundance by percentage cover or the DAFOR scale.

From 2012 data recording by volunteers and professional staff used specifically designed digital tools that not only managed data but also fed back information to users. The web- and map-based interactive geo-database for IAPS management made it easy to acquire survey and monitoring data and to translate changes in IAPS treatment status and abundance to maps presented on the website. An online platform, the MinkApp was developed in collaboration with Aberdeen University's dot.rural initiative (http://www.dotrural.ac.uk/) for the recording, management and presentation of data derived from American mink control. The MinkApp used natural-language-generation (NLG) to inform volunteers by email of mink captures and sightings in their area.

Trends in mink detections and captures were used to determine whether large-scale coordinated control efforts had had an impact on mink populations. The best (least biased) impact data were derived from the checking records for mink rafts. Detection rates could be calculated from the percentage of raft checks where mink footprints were observed. Further analysis through a generalised linear mixed effects model (GLMM) was carried out on long-term mink detection data from three test catchments: the Dee, Spey and Ythan where control had been ongoing since 2006 / 2007 (Bryce, et al,. 2011; Lambin, et al., 2019). The GLMM model accounted for differences among catchments, which was fitted as a fixed effect and as an interaction with time of mink control (i.e. the effect of mink control was allowed to vary by catchment). Non-independence between multiple records from the same raft(s) was accounted for by fitting raft as a random effect.

The effectiveness of IAPS treatment was assessed by the area cleared of infestation (i.e. no regrowth occurred for a year or more), the percentage decrease in coverage and the number of sites in a low maintenance state (DAFOR ≤ 1 (Rare) = 1–10% coverage) before and after treatment. Where coverage was recorded using the DAFOR scale, the mid value for each category of the index was used. Use of DAFOR categories, although simpler to record, encompasses score ranges of 10–25% and therefore more subtle changes in IAPS coverage may not be apparent with this index.

RESULTS

Stakeholder engagement

Throughout the reporting period a total of 1,000 volunteers serviced 2,020 surveillance points for mink control (Fig. 2) and at least 391 volunteers participated in IAPS control, contributing ≥ 2,587 hours of work. Actual numbers at any given time varied, being dependent on the size of area being managed and funding availability. In 2015 there were approximately 800 volunteers participating in mink and IAPS control. Continual recruitment was necessary to offset loss of volunteers. Volunteers left because of a number of reasons. A small but significant number decided it was not really something they wanted to do shortly after recruitment. Other reasons were moving from the area, changed employment and boredom.

Volunteers participating in mink control were from a broad range of backgrounds. Residents of the area with no

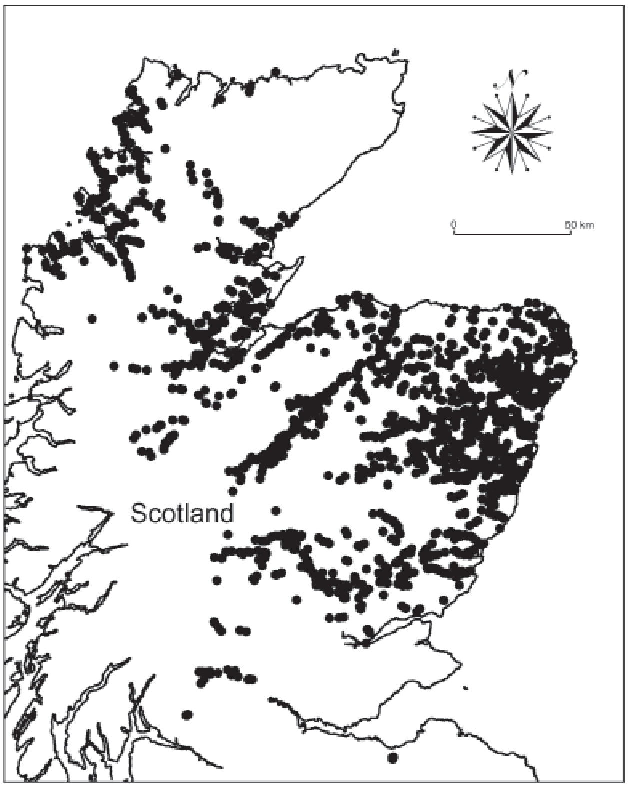

Fig. 2 Location of the 2,020 surveillance points (rafts, tunnels, traps) monitored for American mink between 2006 and 2015.

connection to the local environment constituted the largest proportion, followed by two professional groups – estate workers (game keepers and land managers) and fisheries personnel (managers, owners, guides and anglers). These three groups provided 78% of volunteers. The remaining 22% came from conservation organisations, government agencies and local councils, the tourism and leisure industry, farmers, fish farmers and University staff.

The degree to which individual volunteers engaged with control activities varied greatly, with most content with participating in surveillance e.g. checking mink rafts. However, a relatively small but significant proportion of volunteers, in terms of their contribution, received instruction for skilled activities e.g. humane despatch, stem injection and foliar spray near watercourses. These latter tasks required informal training and/or certification and increased commitment from the volunteer and host organisation.

There was only one landowner where there was issue with gaining access to land despite the large geographic area and the number of landowners involved. Access permissions were initially given verbally but insurance requirements meant that written permissions were increasingly required.

American mink

Across the entire control area, and considering all raft-check records in a calendar year, there was a steady decline, from a positive check rate of around 0.14 in 2011, to a low of around 0.02 in July 2015 (Fig. 3). The majority of the 86 positive raft checks towards the end of the study period (from a total of 2,776 recorded in the period July 2014 to July 2015) were concentrated along the frontier of the project area, which was consistent with frontier catchments receiving an influx of dispersing mink from outside of the control area and the coast.

Trends in mink captures followed those of the detection rate, with a decrease from over 280 in 2012, to only 98 mink captured in the 12 months prior to July 2015. Although mink were captured across the raft network, the areas with the highest numbers of captures reflected the optimum habitat for mink and the history of control effort. In agreement with the mink raft detection data, nearly all of the captures in 2015 were from lowland or coastal areas, indicating an overall contraction of the mink population both in range and population size (see also Lambin, et al., 2019).

The GLMM analysing how mink detection rate changes with year of mink control, showed a clear and statistically significant ($P < 0.0001$) negative relationship (Fig. 4; Table 2). Based on the fitted curves, the model predicts that

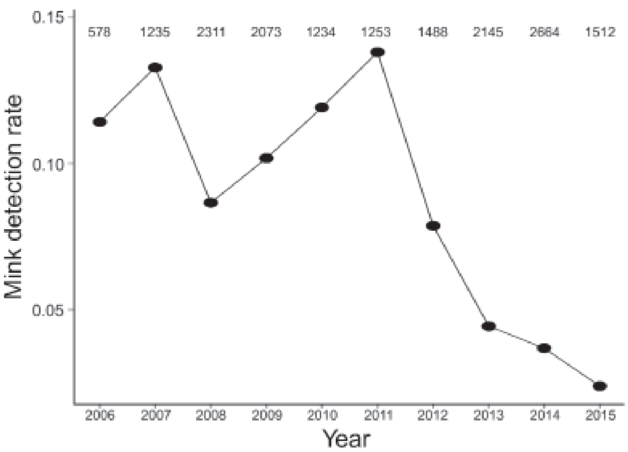

Fig. 3 Changes in the mink detection rate (number of positive raft checks / total number of raft checks) per year of coordinated mink control. Numbers above the points show the total number of checks from which the rates are estimated.

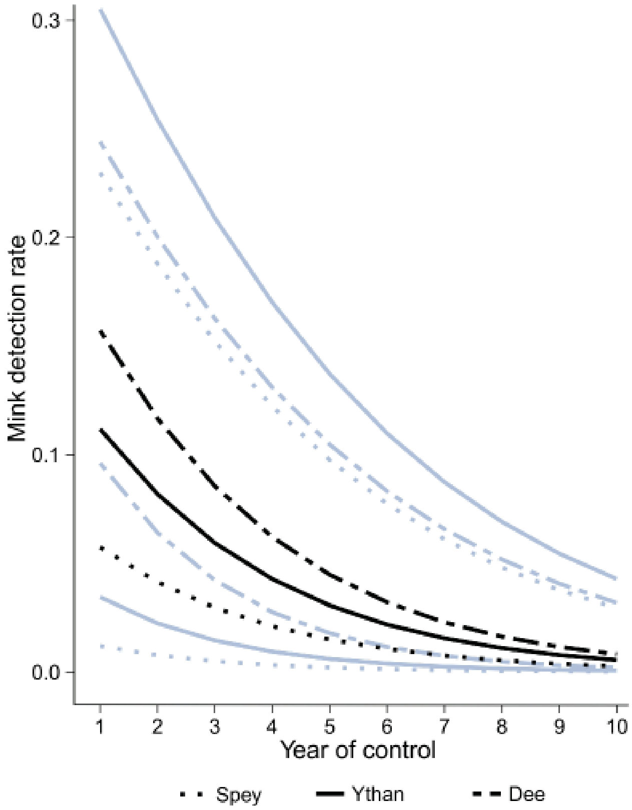

Fig. 4 The effect of control on mink detection rates (abundance) calculated by a generalised linear mixed effects model (GLMM). The black lines are fitted curves for the Dee, Spey and Ythan river catchments The grey lines areas are 95% profile confidence intervals.

Table 2 Summary table for a generalised linear mixed model (GLMM) analysing the relationship between mink detection rate (per raft check) and the year of mink control (by river catchment). Data are for the rivers Dee, Spey and Ythan. Observations is the number of raft checks. Groups refers to the number of rafts.

Observations: 9086	**Groups: 399**	**Residual d.f. 9079**	**Variance: 1.57**	**St. dev: 1.25**
	Estimate	S.E.	Z value	P value
Intercept	-1.33	0.24	-5.67	< 0.0001
Year of control	-0.35	0.05	-7.27	< 0.0001
Catchment (Spey)	-1.12	0.44	-2.58	0.01
Catchment (Ythan)	-0.55	0.30	-1.85	0.06
Year of control: Spey	0.00	0.10	0.02	0.99
Year of control: Ythan	0.15	0.06	2.54	0.01

mink abundance will be reduced to ca. 40% of the starting abundance in four years and further to around 6% of initial levels after nine years. A large amount of the uncertainty in the model's predictions (illustrated by the 95% confidence intervals [grey lines] in Fig. 5) is attributable to differences between the catchments, rather than the overall estimate of the effect of mink control (Table 2). This was particularly true of one catchment where the mink population remained high before dropping abruptly after control in the adjacent catchment.

A small number of rafts influenced trends significantly with a majority of rafts never detecting any mink footprints. All information on mink presence came from 36% of rafts (n = 357) checked at least once. In fact, only 6% of checked rafts (a mere 59) accounted for 637 (53%) of the 1,307 detections. Whilst factors such as duration of raft placement and checking frequency may influence this result, the take home message is that a small portion of the raft network does most of the work in detecting, and vis a vis removing, mink.

Invasive alien plants

The 10 river Trusts that supplied information surveyed a minimum of 2,403 km of waterways (Table 3). Their surveys revealed that IAPS were widespread (extending over ca. 1,603,821 m^2) and had become a serious threat to riparian biodiversity and activities along Scottish river corridors.

Japanese knotweed was the most frequently encountered IAPS. Trusts recorded it in all survey areas though the extent varied significantly among them (Table 3). Giant hogweed was least prevalent and abundant. Three Trusts reported it absent and a fourth discovered only one small stand. But in all other areas infestations averaged > 4,000 m^2. In Ayrshire giant hogweed had invaded 188,000 m^2 . Himalayan balsam infestations proved to be the most challenging. This IAPS had reached 699,233 m^2 of river corridor. Stands in two catchments extended over tens of kilometres.

Table 3 Summary of the area surveyed (in metres) and area recorded as infested by each IAPS for each of the 10 trusts (reported as m^2).

	Area surveyed	GH	HB	JK	Total
Annan	197,000	20	200,000	11,364	211,384
Argyll	195,000	-	-	9,198	9,198
Ayrshire	739,000	188,000	204,000	257,000	649,000
Cromarty	300,000	27,000	128,500	54,500	210,000
Dee	170,000	4,176	32,938	41,768	78,882
FNLT	103,500	72,000	62,700	88,500	223,200
Galloway	114,000	4,196	75	21,663	25,934
Lochaber	42,400	0	0	43,500	43,500
Nith	160,450	41,955	70,430	39,718	152,103
WSFT	30,000	0	590	30	620
Total	2,403,834	337,347	699,233	567,241	1,603,821

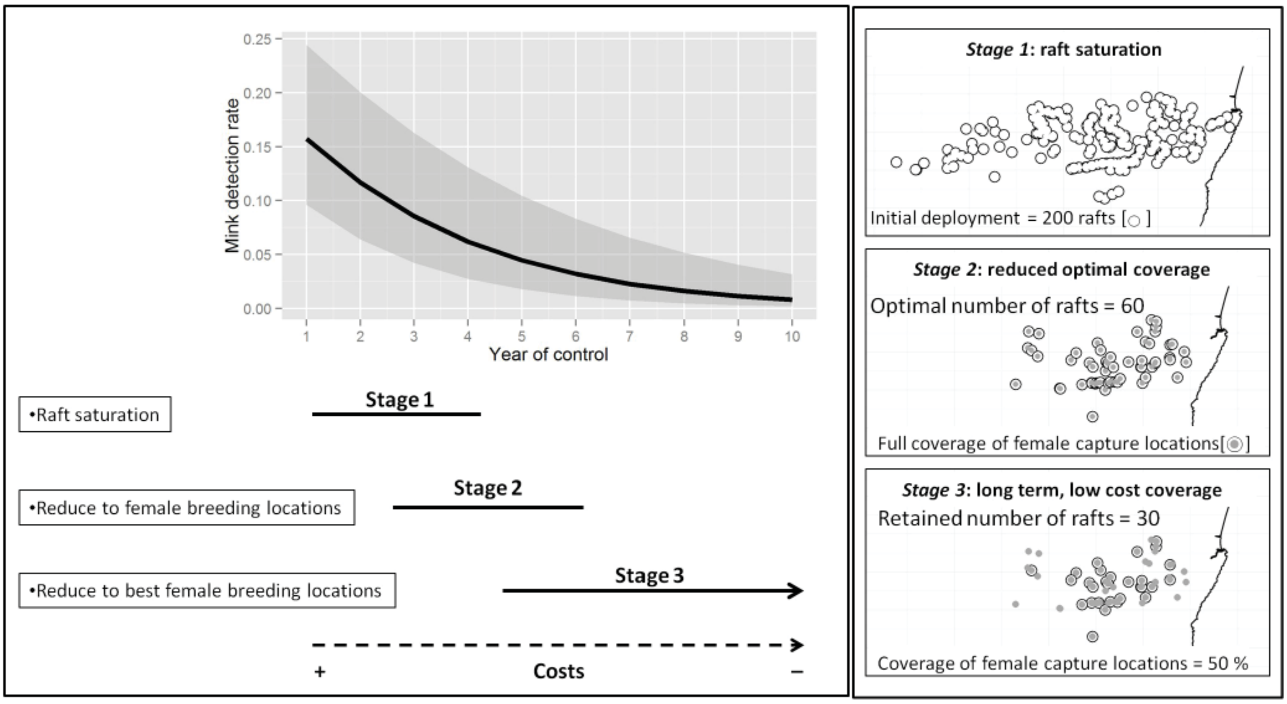

Fig. 5 Schematic of a graduated three-phase strategy for mink control (based on capture data from the River Dee, NE Scotland). In Stage 1 (years 1–4) mink abundance is at its initial maximum. The box on the right illustrates how the strategy moves from a saturated raft network in Stage 1, to cover all female capture locations in Stage 2, and only a subset of these in Stage 3.

Success in clearing areas of infestations was limited with 16%, 11% and 10% of the original area of infestation cleared for giant hogweed, Japanese knotweed and Himalayan balsam, respectively (Table 4). However, decreases in coverage between 50% and 80% were common for all three target IAPS.

The greatest decrease in coverage was for Japanese knotweed, with five areas achieving >85% decrease. Despite the reduced coverage, shoots from the sub-surface rhizome prevented sites from being categorised as cleared. Cover of giant hogweed fell by 53%–75%. However, there was mixed success in controlling Himalayan balsam (Table 4). Trusts reported that effective control of this IAPS was problematic as it is easy to miss individual plants hidden among native vegetation, or in areas of limited access. In four areas, Himlayan balsam was also anecdotally observed to quickly colonise sites that had recently been cleared of giant hogweed or Japanese knotweed. Of note, however, is that both Nith and Cromarty Trusts, using a targeted approach and a larger coordinated workforce, decreased coverage of Himlayan balsam by >82% (as well as clearing >29,000 m^2) across large areas.

Standardised percentage coverage decreased from a median of 38% (mean 33%) to 5% (mean 14%) in 447 pre-treatment sites following control. The majority of sites (327; 73%) showed improvement, 103 (23%) were recorded as having no change, and infestation levels at 17 (4%) had got worse. Around half the sites infested by giant hogweed and Japanese knotweed, and 38% of those by Himalyan balsam, were in a low maintenance state after treatment (Table 5). This was despite the reported increase of infestations of giant hogweed after the large floods of the winter of 2013/14.

Costs

The work reported in this paper was undertaken through the sequential securing of short-term (1–4 year duration) funding. Consequently funding was cyclical with periods of higher funding alternating with those of low or no funding (Fig. 6). Using northern Scotland as an example, the amount of funding secured for IAS work has increased in each subsequent funding phase, from £124,000 (1996–2005), £639,000 (2006–2009) to over £1.95 million provided in the period 2010–2015. The increased funding reflected the expanding geographic reach (from 5,000 km^2 to almost 30,000 km^2) and complexity of the work undertaken. This included the addition of IAPS control in 2009 and biosecurity, awareness, education and capacity building activities after 2010.

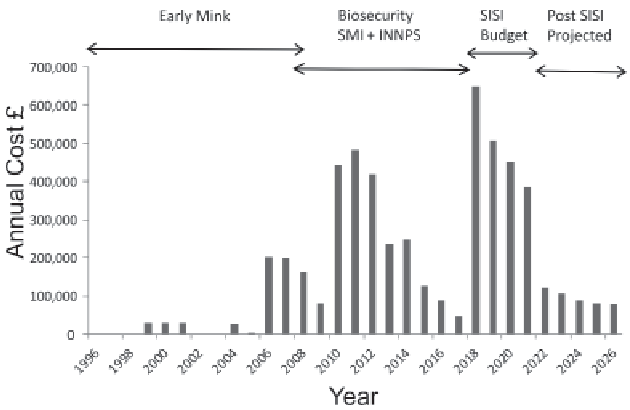

Fig. 6 Funding for IAS work in northern Scotland.

Table 4 Area cleared (no growth detected in post-treatment survey) in m2 and relative percentage decrease in mean coverage at infested sites for each of the INNPS and each Trust. A dash (-) indicates that no data were available.

	GH		HB		JK	
	Area cleared	**% decrease**	**Area cleared**	**% decrease**	**Area cleared**	**% decrease**
Annan	0	53	0	19	0	63
Argyll	-	-	-	-	8,070	88
Ayrshire	53,452	60	>6,684	25	29,722	47
Cromarty	750	57	38,000	82	7,750	81
Dee	0	0	0	0	0	65
FNL	-	75	-	-	-	-
Galloway	-	-	0	50	2,840	84
Lochaber	-	-	-	-	16,268	42
Nith	40*	-	29,871	94**	0	99
WSFT	-	-	0	90	30	100
Total	54,242		>74,555		64,680	

* The Nith group recorded the number of plants treated, rather than area cleared.
** Percentage calculated as change in the *number of plants* treated between initial (maximum) levels and final treatment in 2014.

Table 5 Number and percentage of the total number of sites that were in a low maintenance state before and after treatment for each target species.

		Before treatment		After treatment	
	Total no. of sites	**No. of sites**	**% of sites**	**No. of sites**	**% of sites**
Giant hogweed	468	82	17	243	63
Japanese knotweed	598	41	7	295	88
Himalayan balsam	293	40	14	111	100

DISCUSSION

Findings and lessons for future work

The control strategies and measures for both IAPS and American mink have had a demonstrable, although variable, impact in suppressing target populations in terms of coverage and population density over large geographic areas (see also Bryce, et al., 2011; Melero, et al., 2015; Oliver, et al., 2016). The variation in results suggests there is room for improvement in strategy and local implementation.

The use of an evidence-based approach, derived from evaluation of activities and research associated with the project, provided the central core of the adaptive-management strategy. The findings were utilised to improve control strategies (e.g. concentration of surveillance in lowland areas and along migration routes for American mink, control methods for IAPS, engagement and retention of volunteers, and implementation of management efforts at an appropriate geographic scale in defensible areas for all IAS). An example of the latter is that the GLMM analysis highlighted the importance of taking a coordinated multi-catchment approach to mink control as the number of mink in a catchment depends on control both within that catchment and in neighbouring catchments.

Working over such a large geographic scale, including urban areas, with limited secured funding was made possible by the use of a large trained volunteer workforce supported by professional staff. Staff were either employed by the project or from local organisations. The latter arrangement allowed the building of capacity for volunteer management and IAPS control within the organisations. Although this approach helped to build longer term management sustainability, it sometimes resulted in competing priorities between the project and the organisation. Employing dedicated project staff avoided this conflict but did not effectively address long-term sustainability, as employment ended with the cessation of project funding.

The use of large volunteer networks rather than increased numbers of staff reduced employment costs, a significant cost. However it did not reduce liability risk for the organisation(s) that supported the network. To mitigate risk as the project developed, RAFTS increasingly used written rather than verbal permission for volunteer participation and access agreements. The information and training given to volunteers increased, particularly regarding health and safety. Organisation policies and public liability insurance was also regularly reviewed in light of volunteer numbers and their work. Changes in project management structure required revision of all agreements. One outcome of these changes was that significant numbers of volunteers expressed concern and dissatisfaction with perceived increased bureaucracy, with a small number withdrawing their participation.

Management over such a large area required the building and maintenance of coordinated partnerships with defined roles for individual partners at both local and national level (Table 6). At the local level, non-government/non-profit organisations (Trusts) provided the hub of the partnerships and collaboration. The Trusts have close ties to sectors of the local communities, particularly landowners. At the national level RAFTS was the main contact point for government agencies and universities, and coordinated the work of the local organisations. Partnership arrangements were not pre-determined but rather developed over the course of the work and in response to the varied demands of the management strategies employed. Partnerships and collaboration involved over 70 organisations, including the Scottish Government, state agencies, local authorities, universities, >50 local non-government organisations and businesses and over 800 volunteers at any one time.

Coordination was generally effective but there were instances of inconsistency of approach and in data collection among local organisations (Arts, et al., 2013). Although consistency of data collection improved with the advent of the on-line reporting systems, ensuring consistency of approach and data collection among large numbers of local organisations remained a significant challenge.

Common interest formed the basis for collaboration. Differing characteristics of communities (individuals and community organisations) within and among geographic areas of Scotland meant approaches to engagement varied. The diverse composition of the volunteer base demonstrated that IAS control, particularly of American mink, provided a common base for a wide range of community groups, some of which had a history of conflicting interest (e.g. gamekeepers and bird conservationists). Motivational factors included professional or commercial interest and a concern for the local environment – as expressed by residents who made up a large proportion of the volunteers.

Taking action and demonstrating results were important factors in retaining participating volunteers and organisations. Demotivating factors included the breaks in project activities caused by short term funding cycles and perceived increased bureaucracy. The use of on-line reporting systems provided a means to disseminate progress and results through a limited functionality for data interrogation (mink) (Beirne & Lambin, 2013) and a map interface for IAPS. These reporting mechanisms became part of an overall volunteer and organisational recruitment and retention strategy that combined a variety of awareness activities with training and legal empowerment. Successful control also influenced volunteer retention with the lack of detection of IAS leading to boredom. Maintaining interest and motivation remains a critical long-term challenge for future management (Beirne & Lambin, 2013).

Despite repeated efforts to obtain long term funding, IAS control in Scotland has relied on short term, or project specific, funding. The resultant funding cycles occur as one project has to finish before funding for the next stage can be secured. Start-stop cycles result in a loss of staff, volunteers, equipment and, as a consequence, momentum, capacity and credibility (see also Lambin, et al., 2019). Furthermore, overall costs increase as start up costs (staff and volunteer recruitment, training, control) exceed recurrent costs of established projects.

Funders' regulations also influence the work that can be undertaken. The majority of short-term funders require

Table 6 Contributions by participating institutions.

Level of collaboration	Partnership organisations
Strategy	RAFTS, GB Non Native Species Secretariat, Scottish Environment Protection Agency, Scottish Natural Heritage, national park authorities (Cairngorms and Loch Lomond),
Management	RAFTS and 18 member Trusts
Implementation	18 local trusts, other non-government organisations e.g. (Scottish Wildlife Trust, Royal Society for the Protection of Birds), local authorities (Highland, Moray, Rural Aberdeenshire, Angus, North Tayside, Argyll and Bute, Ayrshire, Dumfries and Galloway.
Evaluation	RAFTS, University of Aberdeen, Queens University Belfast.

tangible benefits for their support. These benefits are more easily expressed in terms of IAS reductions than prevention (biosecurity), where no occurrence or a 'negative' result defines success. Regulations have also prevented funding being used for rapid-response, another key element of successful IAS management. Funding for IAS management should recognise that 'negative' results indicate success both in prevention and control, have flexibility to allow for rapid response and changes in approach required by adaptive-management and be available for work over appropriate geographic- and time-scales.

Although there is still no long-term funding of IAS control in Scotland, project funding has been secured for the Scottish Invasive Species Intiative (SISI) (2018–2022). SISI aims to develop a long-term, cost-effective management system for multiple IAS across 29,500 km^2 of northern Scotland. The project builds on the experiences of its predecessors and tests more focused strategies for IAS management.

One such approach to mink control derives from the variation in the relative contribution of individual rafts to overall detection rates, coupled with the analysis from the GLMM. The model predicts abundance will be more than halved following four years of control and reduced to < 10 % after ten years. Accordingly, capture data will be used to reduce raft coverage in three stages over the same timeframe (Fig. 5). If patterns of mink dispersal and settlement are influenced by habitat quality, despite the species's mobility and generalist habits, reductions would track capture rates for females. This assumes that populations under control pressure will reoccupy optimum habitat preferentially, and that concentrations of female mink will indicate where that is. Reactive redeployment may be required in response to localised increases in mink activity. If successful, the strategy will use the best available evidence and scientific understanding to substantially reduce costs.

Protecting non-invaded areas through awareness targeted to user groups (e.g anglers and boaters) and the use of biosecurity stations and individual biosecurity kits is a key component of the project. Habitat restoration using resilient native communities will be tested as a means to reduce reinvasion risk of areas cleared of IAS.

Emphasis is placed on strengthening the capacity of local organisations, so IAS management becomes part of normal working practices. SISI will also develop means to maintain volunteer participation over the timeframes required to manage IAS. Evidenced-based adaptive management is central to the strategic approach of SISI and the project will develop interactive and map-based data-recording systems.

SISI faces some significant challenges in balancing costs with outcomes, particularly in regard to reducing introductions and spread over such large geographic areas and defining what reduction in IAS can be sustained. Effective coordination, and quality assurance, of the work undertaken by multiple local organisations is not to be underestimated.

Despite the challenges, it is envisaged that by the end of SISI the more focused control will have suppressed target populations to levels where that suppression can be affordably maintained by motivated local organisations and their volunteer networks (Fig. 6, from 2022–2026). However, post-project IAS management in Scotland will still require additional funding to that provided by local organisations and at present it is not clear how that will be provided.

ACKNOWLEDGEMENTS

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REFERENCES

Arts, K., Webster, G., Sharma, N., Melero, Y., Mellish, C.S., Lambin, X. and van der Wal, R. (2013). 'Capturing mink and data: Interacting with a small and dispersed environmental initiative over the introduction of digital innovation'. *Framework for Responsible Research and Innovation in ICT FRRIICT.* <http://responsible-innovation.org.uk/resource-detail/1059/>. Accessed August 16, 2015.

Beirne, C. and Lambin, X. (2013). 'Understanding the determinants of volunteer retention through capture-recapture analysis: Answering social science questions using a wildlife ecology toolkit'. *Conservation Letters* 6: 391–401.

Bryce, R., Oliver, M.K., Davies, L., Gray, H., Urquhart, J. and Lambin, X. (2011). 'Turning back the tide of American mink invasion at an unprecedented scale through community participation and adaptive management'. *Biological Conservation* 144: 575–583.

Forest Research. (2008). *A Survey and Appraisal of Rhododendron Invasion in Argyll and Bute*. Roslin, Midlothian, UK:Forest Management Division, Northern Research Station.

Fraser, E.J., Macdonald, D.W., Oliver, M.K., Piertney, S.B. and Lambin, X. (2013). 'Using population genetic structure of an invasive mammal to target control efforts – an example of the American mink in Scotland'. *Biological Conservation* 167: 35–42.

GBNNSS. (2008). *The Great Britain Invasive Non-native Species Strategy*. Sand Hutton, York, UK: The Great Britain Non-native Species Secretariat, Animal and Plant Health Agency.

Kent, M. and Coker, P. (1992). *Vegetation Description and Analysis: A Practical Approach.* New York: John Wiley and Sons..

Lambin, X., Horrill, J.C. and Raynor, R. (2019). 'Achieving large scale, long-term invasive American mink control in Northern Scotland despite short-term funding'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 651–657. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Marine Scotland. (2017). *An Analysis of the Value of Wild Fisheries in Scotland*.Edinburgh, UK: Marine Scotland.

Melero, Y., Robinson, E. and Lambin, X. (2015). 'Density-and age-dependent reproduction partially compensates culling efforts of invasive non-native American mink'. *Biological Invasions* 17: 1–13.

Oliver, M.K., Piertney, S.B., Zalewski, A., Melero, Y. and Lambin, X. (2016). 'The compensatory potential of increased immigration following intensive American mink population control is diluted by male-biased dispersal'. *Biological Invasions* 18: 3047–3061.

Raynor, R., Horrill, C., Taylor, M. and Taylor, K. (2016). 'American Mink. Version 1.0.'. In: M.J. Gaywood, P.J. Boon, D.B.A Thompson and I. Strachan (eds.) *The Species Action Framework Handbook*, p. 27. Perth, Scotland: Scottish Natural Heritage.

Reynolds, J.C., Short, M.J. and Leigh, R.J. (2004). 'Development of population control strategies for mink *Mustela vison*, using floating rafts as monitors and trap sites'. *Biological Conservation* 120: 533–543.

Ruiz-Suarez, Melero, Y., Giela, A., Henriquez-Hernandez, L.A., Sharp, E., Boada, L.D., Taylor, M.J., Camacho, M., Lambin, X., Luzardo, O.P. and Hartley, G. (2016). 'Rate of exposure of a sentinel species, invasive American mink (*Neovison vison*) in Scotland, to anticoagulant rodenticides'. *Science of the Total Environ*ment 569–570: 1013–1021.

SEPA. (2009a). *Scotland River Basin Management Plan 2009–2015*. Scottish Environment Protection Agency. <www.sepa.org.uk>. Accessed 13 August 2017.

SEPA. (2009b). *Solway Tweed River Basin Management Plan 2009–2015*. Scottish Environment Protection Agency. <www.sepa.org.uk>. Accessed 13 August 2017.

SNH. (2011). *Scottish Recreation Survey, Annual Summary Report*. SNH Commissioned Report 535. <www.snh.gov.uk>. Accessed 13 August 2017.

Tweed Forum. (2006). *The Tweed Invasives Project*. Melrose, UK: Tweed Forum.<http://www.tweedforum.org/publications/tweed-invasives>. Accessed 13 August 2017.

UKTAG. (2015). 'Revised classification of aquatic alien species according to their level of impact'. *Aquatic alien species and the WFD: proposed amendments to the impact classification in the UKTAG alien species guidance*., UK: Environment Agency. <www.wfduk.uk>. Accessed 15 August 2017.

Williams, F., Eschen, R., Harris, A., Djeddour, D., Pratt, C., Shaw, R.S., Varia, S., Lamontagne-Godwin, J., Thomas, S.E. and Murphy, S.T. (2010). *The Economic Cost of Invasive Non-Native Species on Great Britain*. CAB/001/09. UK: Centre for Agriculture and BioSciences International. <www.nonnativespecies.org>. Accessed 14 August 2017.

Yamaguchi, N., Rushton, S. P. and Macdonald, D. W. (2003). 'Habitat preferences of feral American mink in the Upper Thames'. *Journal of Mammalogy* 84: 1356–1373.

Chapter 3: Strategy

With Sections: A Biosecurity
B Collaboration
C Outcomes
D Scaling up

Biosecurity on St Helena Island – a socially inclusive model for protecting small island nations from invasive species

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Abstract St. Helena Island, 122 km^2 (47 sq. miles) is a UK Overseas Territory in the South Atlantic. It is a remote volcanic island situated in the sub-tropics 1,127 km (700 miles) from Ascension Island and 2,736 km (1,700 miles) from South Africa. Its resident population of ca. 4,500 is serviced by a single supply ship which visits up to 25 times a year. Isolation has acted historically as a natural barrier to pest arrival and border control has followed the conventional practice of protecting agricultural interests through restrictions on fresh produce, plant materials, livestock and pets. The benefits of isolation were compromised in 2016 when the first airport opened. Private jets arrive now from Africa, Europe and South America, and commercial flights started at the end of 2017. A programme of biosecurity capacity building and strengthening was established in anticipation of this air traffic. St Helena authorities introduced a national biosecurity framework and associated policy (entitled *Biosecurity St Helena*), the latter constructed through multi-sectoral consultation, and key stakeholders participated throughout in policy development. *Biosecurity St Helena* applies international standards set by the International Plant Protection Convention across the biosecurity continuum. As is typical in small island nations, human and financial resources are limited, so that the biosecurity strategy addresses mainly higher risks. Compliance is heavily reliant on public awareness. Active communication engages all community sectors in biosecurity work through education, information, advocacy and feedback. Authorities use key performance indicators to measure the effectiveness of this approach. *Biosecurity St Helena* is a model of actively socialised biosecurity for other small island nations.

Keywords: biosecurity strategy, capacity building, community engagement, inclusive planning, small island nation

INTRODUCTION

St Helena Island, a United Kingdom Overseas Territory in the South Atlantic Ocean, is a volcanic island with an area of 122 km^2 (47 square miles) and total population of 4,534 (St Helena Government, 2016). St Helena is remote and isolated, lying 1,127 km (700 miles) southeast of Ascension Island and 2,736 km (1,700 miles) from South Africa, with a sub-tropical, maritime climate. A total of 502 endemic species are currently known, comprising around one third of the total endemic biodiversity of the UK Overseas Territories and making a significant contribution to global biodiversity (Churchyard, et al., 2014). The economy is based mainly on agriculture, fishing, a small but growing volume of tourism and income from offshore employment.

Until 2016 the only regular access to St Helena was via the Royal Mail Ship (RMS) *St Helena*, calling around 25 times a year in passage from Ascension Island and South Africa. Most commodities are imported. More than 69% of the island's annual requirements for agriculture and food are sourced from South Africa, including almost all fruit, and significant quantities of vegetables.

In 2010 the UK Government announced its intention to build an airport on St Helena, conditional upon the St Helena Government's commitment to internal investment and increased tourism. Air access and expansion of the tourism sector augmented biosecurity risks to St Helena. To meet the challenge, the St Helena Government launched a programme in 2013 to upgrade biosecurity arrangements. The 2009 South Atlantic Invasive Species Strategy had already defined dedicated biosecurity capacity for St Helena as a strategic priority for the prevention of invasive species and unwanted organisms in the region (Shine & Stringer, 2010).

St Helena has a limited range of existing cosmopolitan pests and is very vulnerable to new introductions harmful to the economy, community health, environment and the new investments in tourism development (Pryce, 2015). Until now, biosecurity has relied heavily on its isolation as an oceanic island and limited modes of entry to minimise exposure to new pest threats.

The Government's Agricultural and Natural Resources Division (ANRD) reviewed biosecurity practices in in 2013 (Key, 2013) and concluded that capacity was inadequate to address the new biosecurity pressures associated with air access. Lack of biosecurity-specific legislation or overall operational framework severely compromised post-border controls and enforcement. There were no fumigation or other specialist facilities for local treatment of contaminated goods. Tellingly, the common interests of different sectors, particularly agriculture, public health and the environment were not harmonised for biosecurity purposes.

PREPARING FOR NEW BIOSECURITY MEASURES

The St Helena Government's programme to upgrade biosecurity in 2013 departed from the existing emphasis on managing agricultural and animal imports at the border. It moved biosecurity to a risk-based approach across the broader continuum of invasive pest organisms in marine and terrestrial environments. Interception measures pre-border, at-border and post-border were to be more closely integrated. Resource limitations in the small-island context argued for greater investments in pre-border controls and post-border surveillance.

Approaches to building the new biosecurity framework

The ANRD led a new policy team comprising agency representatives of Environment, Customs and Public Health; the first time this multi-sector team had been brought to the same table. Their purpose was to establish the architecture of the new biosecurity system through an overarching policy statement. The policy team recognised that understanding of biosecurity issues was essential for community buy-in and compliance. Accordingly, the team developed the new biosecurity policy in full consultation with all sectors in the community, from farmers to politicians, coupled with close participation throughout the reform process. Stakeholders were given multiple opportunities to discuss new ideas and to object to them if warranted. The policy team intended biosecurity

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awareness in the community, as a whole, to benefit from these approaches.

Consultations, commencing with twelve focal groups of stakeholders in 2013, explored attitudes to the current biosecurity procedures. Despite some criticisms, all 54 stakeholders that were consulted supported the current arrangements and the need to strengthen them in anticipation of air access.

Participants in a subsequent workshop agreed on the vision for biosecurity policy and then defined strategic objectives and expected outcomes. The broader public were invited to consider the resulting policy statement. The St Helena Government endorsed the policy after it had been revised to incorporate feedback. Now entitled *Biosecurity St Helena*, the policy was launched officially in the biosecurity facility at the seaport in November 2014 (St Helena Government, 2014).

Today, ANRD is the agency lead for biosecurity. It holds the authority to approve import licences and has the principal duty to launch responses to incursions. The community relies on ANRD for government leadership in matters of compliance and enforcement of the new biosecurity legislation.

Biosecurity St Helena defined

The policy vision and principles govern the new biosecurity arrangements and are supported by the island's legal and institutional structures. The policy is the blueprint for "an effective biosecurity system of shared responsibility that protects the sustainable future of our island environment, allowing a vibrant economy, safe movement of people and goods, and enhanced livelihoods and health" (St Helena Government, 2014, page 3). Overarching outcomes are:

- Effective management of biosecurity risks to St Helena's environment, agriculture, amenities, public health and well-being, including safety;
- Effective governance of St Helena's biosecurity system through shared responsibility and roles

Biosecurity St Helena recognises that a zero-risk approach is not practical and works to reduce the risk to an acceptably low level. The policy endorses a white list and licencing approach, whereby all high-risk goods are prohibited except those for which import health standards have been developed. Import health standards specify the conditions under which goods can be imported and the treatments required in response to pest organisms intercepted or simply suspected pre-border, at the border or post-border.

Six crucial principles guide biosecurity work:

1. Leadership for effectiveness throughout the biosecurity apparatus
2. Clear communication of stakeholder roles, responsibilities, and the 'what, why and how' of biosecurity investments.
3. Shared responsibility across all sectors and interests for mutual benefit
4. Risk-based responsiveness to the probability of border challenges, potential harm and changes in the nature of threats.
5. Evidence-based decision-making supported by quality systematic research
6. Co-operation between sectors to minimise the probability of new incursions and manage existing ones.

BIOSECURITY ON THE GROUND

A multi-sector plan was developed alongside the biosecurity policy to put the new structures in place. The policy team supervised developments for the first year, and thereafter improvements were mainstreamed into the work plans of ANRD, Environment, Customs and Public Health, taking effect in the 2016/2017 financial year.

At the border

The St Helena Government recruited two full-time biosecurity officers in 2015, the first in the island's history. They work closely at the border with Customs whose warrants they also hold. Customs and Immigration officials received the same biosecurity training to ensure harmonised border security.

Import health standards apply for a range of commodities, and the island's main traders assisted the development of these standards. Inspection procedures now align with international standards set by the International Plant Protection Convention (IPPC) for phytosanitary (plant health) risks and the World Organisation for Animal Health (OIE) for zoosanitary (animal health) imports. Inspection practices are codified for consistency and transparency. Import Health Standards, application forms and general guidance are now available on-line at <http://www.sainthelena.gov.sh/st-helena-biosecurity-service/>.

The officers employ a dog trained to detect honey, bananas and citrus, to protect St Helena's disease-free bees and bananas. Likewise, citrus (commonly intercepted on incoming visitors) may introduce newly emerging diseases such as huanglongbing citrus greening. Dog handling at the border is governed by a Standard Operating Procedure written together with Customs who run their own detection dogs.

The full-time team has extended biosecurity operations on the wharf, beyond the former pre-occupation with fresh produce. Customs help with passenger and cargo profiling so that higher risks can be ranked for quarantine inspections. Profiling relies on interception data for visitors and surveys of imported cargo arriving by sea, and will be refined as data accumulates for both visitors and freight arriving by air.

Personal goods in shipping containers and vehicles shipped in break-bulk were predicted to be high risk freight. Between January 2016 and March 2017, 99 (40.4%) of 245 imported vehicles (mostly cars) were found to be contaminated with soil. Inspectors intercepted 75 live spiders in 16 (6.5%) of the vehicles. Over the same period, 23 live spiders were intercepted in four (11.8%) of the 34 incoming containers of personal goods. The spiders belonged to seven species known from the UK and South Africa. They were found mainly in the space behind vehicle wing mirrors, on the windscreen wipers, and behind the rear-mounted spare wheel on SUVs. Most spiders in shipping containers were discovered immediately inside the doors.

Soil samples collected from vehicles (typically from rear wheel-arches) were weighed, then placed in seed trays for up to two months to check for seed germination. A mixture of grasses and small dicotyledons germinated successfully from nine (9.1%) of the 99 samples but none survived long enough to identify species.

Building and operating St Helena airport

Construction of St Helena airport commenced in January 2012. Three new biosecurity pressures had to be managed

A second supply vessel now visited every six weeks or so until October 2015. The ship departed from a new port of origin and was the first vessel able to moor alongside the island at a specially constructed wharf. The normal supply vessel RMS *St Helena* barged freight ashore from an anchorage in the bay.

The new vessel discharged large quantities of construction materials, including river and dune sand.

Several hundred off-shore workers arrived (mainly from Africa and Thailand), for whom biosecurity awareness was low to zero.

ANRD negotiated quarantine agreements with the South African construction company. Consignments of sand were fumigated in Namibia and inspected on arrival in the port area. The team inspected break-bulk consignments before disembarkation from the vessel. Compliance improved to a good standard after some initial teething troubles. Only two pests were intercepted during the construction phase - flattened giant dung beetles (*Pachylomera femoralis* Coleoptera: Scarabaeidae) on open metal gantries; and ice plant, (*Galenia papulosa* Aizoaceae) in river sand. Construction staff were quick to report biosecurity issues and responded appropriately.

Border and biosecurity officials meet all inbound flights. Airport biosecurity is guided by a Standard Operating Procedure refined through preliminary test-runs with flights and arriving passengers. An x-ray scanner screens all in-bound baggage. Fresh produce is examined in a small, sole-purpose biosecurity room in the airport's cargo compound.

The Public Health Committee obliges 'disinsection' of all inbound flights, recognising known risks of introducing aerial insect vectors such as mosquitoes (Gratz, et al., 2000). Eighteen private jet and three medevac flights had been treated by March 2016. Commercial flights had not yet commenced.

Post-border surveillance

The 2013 review of biosecurity (Key, 2013) revealed serious weaknesses in post-border surveillance for pest species by-passing earlier lines of defence. Today, monitoring and surveillance behind the island's borders are structured to detect and eradicate pest intruders before they can establish. Biosecurity staff direct their attention to surveillance at the airport construction site and all other ports of entry; targeted surveillance for introduced tephritid fruit flies; and readiness to respond to pest detections.

Surveillance at the construction site has mapped every location at which shipping containers were landed or opened with the participation of the construction company. At each location, the biosecurity team installed a monitoring point comprising a covered breeze block in which crumpled newspaper and a sticky trap attract and contain unwanted invertebrates. A monitoring protocol, identification guide and reference collection assist surveillance. As construction wound down in 2016, monitoring was migrated to new sites around the two seaports and the airport. Each station will include mosquito traps in the future.

Surveillance operates pheromone-baited sticky traps for five species of economically harmful tephritid fruit flies at ten pivotal fruit-growing sites across the island.

The biosecurity team have engaged relevant stakeholders in the preparation of nine response plans for incursion emergencies. The plans address terrestrial and marine risks from a range of phytosanitary, zoosanitary and invasive non-native species. They were refined through a simulation exercise, and further exercises are planned for the future.

Engaging the community

The principle of responsibility shared universally by the St Helena community and visitors is central to biosecurity arrangements. But policy consultations with stakeholders revealed poor understanding of what biosecurity is and what the biosecurity team does. In response, a multi-sector communication strategy targeted key audiences with biosecurity messaging. The strategy adopted *Border Security* (a popular TV programme on Australian border security services) as its brand but switched later to *Biosecurity St Helena* to align messages with the new biosecurity policy. *Biosecurity St Helena* branded pens, shopping bags and mugs were a popular means of reinforcing the messages. Outreach comprises a programme of press releases, articles in the local print and radio media, activities with local primary and secondary schools, and visits by groups to observe biosecurity inspections at the wharf. Councillors, government officials and airport officials were among the first groups invited. The outreach programme continues as a core element in the biosecurity team's work plan. New stakeholders involved in air access readily embraced the messaging, which focused on collective responsibility for protecting the island for the future.

Site visits were very productive; feedback was positive from visitors who were not previously aware of the wharf facility or only generally familiar with the biosecurity team's functions.

The public are actively encouraged to be vigilant for new invasive non-native species. In March 2015, a public awareness campaign comprising press announcements and leaflets invited the public to report unusual tracks, signs, weeds or invertebrates. Reporters are rewarded with a gift of branded promotional goods.

Sustaining external support for *Biosecurity St Helena* is a priority. Biosecurity reaches well back into the supply chains through visits to overseas agents and suppliers who are expected to comply with stringent, time-consuming or costly quarantine requirements often for commodity quantities small relative to their normal trade volumes. Face-to-face contact with suppliers and South African Cape Inspection Service aims to translate their goodwill into co-operation, especially for frequently imported high-risk goods such as South African, produce and plant propagation materials.

Measuring success

A comprehensive database records imports and interceptions. Another holds baseline data for all taxa of native and introduced species, together with a reference collection of pest species known on the island. ANRD uses these data to measure biosecurity outcomes and assess threats based on empirical evidence.

Even so, establishing meaningful indicators to measure biosecurity effectiveness is a challenge. The number of interceptions is a commonly used metric, but one open to confounding interpretation: does an increase in the number of interceptions indicate (i) a decrease in effectiveness (i.e. more introductions arriving owing to poorer pre-border measures) or (ii) an increase (more interceptions owing to better inspection practices)?

To resolve this ambiguity, ANRD uses five key performance indicators based on the notion of tolerance thresholds for interceptions. Once a threshold is exceeded, the biosecurity team investigates likely causes and applies appropriate remedies.

The indicators relate to passenger, fresh produce and cargo pathways arriving by sea (Table 1). Table 2 shows the biosecurity performance results for the 2016 calendar year. None of the thresholds was exceeded in any indicator.

Table 1 Tolerance thresholds used as measures of biosecurity performance for three main risk-pathways on St Helena Island.

Pathway	Threshold	Notes
Percentage of passengers arriving without a quarantine-risk item in their baggage	No more than five in every 100 passengers arrive with prohibited goods such as honey, fruit, nuts	Includes passengers and crew on RMS *St Helena* and private yachts, but excludes day-visitors on cruise ships
Percentage of fresh-produce lots[1] inspected which do not conceal a quarantine pest	No more than five in every 100 lots inspected have a quarantine pest (dead or alive)	
Number of quarantine pests detected at the border as a percentage of the total number of imported shipping containers and uncontainerised vehicle of any type	No more than three quarantine pests detected for every 100 units	
Number of quarantine pests detected post-border as a percentage of the total number of imported shipping containers and uncontainerised vehicle of any type	No more than two quarantine pests detected post-border for every 100 units	
Number of animals breaching border biosecurity requirements as a percentage of total animal imports Breaches include identity issues, disease, or incorrect paperwork	No more than five in 100 animal imports fail to satisfy requirements	

[1]A "lot" is defined as the total amount of any one type of produce which are clearly from the same source.

Table 3 lists commodity types by passengers' reasons for visiting. The 'other' class of passenger was most likely to arrive with prohibited goods. This class includes construction workers possessing few or no English language skills and therefore less likely to have understood the biosecurity arrival information provided on the ship. Fresh produce was most frequently seized, typically apples, pears, oranges and other citrus picked off the breakfast table in the ship before disembarkation.

DISCUSSION

Biosecurity compliance and enforcement can be challenging in small, isolated communities if stakeholders are not willingly engaged through knowledge of need and benefit. Socialising the processes of building a strong biosecurity system through active participation is particularly important. *Biosecurity St Helena* is a relatively short and succinct blueprint which could have been constructed quite quickly. Instead, the St Helena Government chose purposefully to pursue a process of intensive consultation which extended preparation over a period of nearly a year. Thus, the drafting process was considered as important for social acceptance as the resulting document. Local priorities and concerns are now reflected in the language and layout of the plan.

Most importantly, *Biosecurity St Helena* demonstrates the benefits of political will to integrate agricultural and environmental interests for biosecurity purposes. Limitations on human resources common to small island states have been largely overcome on St Helena by close co-operation between biosecurity and customs officials.

Table 2 Results for five key performance indicators for the 2016 calendar year in which the RMS St Helena, 186 yachts and eight cruise ships visited.

Indicator	Threshold	2016	Data
Percentage of passengers arriving without a quarantine-risk item in their baggage	95%	98%	3,930 passengers arrived 469 items confiscated from 60 passengers, of which 76% were fresh produce, 1% honey, and 24% other items
Percentage of fresh-produce lots inspected which do not carry a quarantine pest	95%	97.4%	62 phytosanitary import licences issued" 366,085 kg fresh produce and 16,050 kg seed potatoes imported 536 lots inspected
Number of quarantine pests detected at the border as a percentage of the total number of units imported	3	1.1	1,023 containers and 250 vehicles imported 14 interceptions, of which 4 were tephritid larvae, 8 Lepidoptera larvae and 2 other taxa
Number of quarantine pests detected post-border as a percentage of the total number of units imported	2	0.1	1 interception: a chafer beetle
Number of animals that breach border biosecurity requirements as a percentage of total animal imports	5%	0%	42 animal import licences issued

Table 3 Goods seized from passengers (n = 60) by purpose of visit in 2016. Some passengers imported more than one type of risk item.

Purpose of visit	Type of items seized		
	Honey	Fresh produce	Other produce
Returning resident	0	12	7
Tourist	0	5	3
Government worker	1	4	3
Other	1	18	10

Extending sea-port biosecurity vigilance to high-risk shipping containers and vehicles revealed their prominence as vectors for harmful hitch-hikers such as spiders. This had not been known before.

Two main weaknesses remain in the island's biosecurity framework. First, new biosecurity legislation has been delayed by other priorities in the Attorney General's Office. In the meantime, existing statutes and regulations are neither harmonised nor aligned with international biosecurity expectations, so that *Biosecurity St Helena* lacks explicit legal mandates for compliance and enforcement. Warnings must substitute for fines and other legal sanctions, a shortfall which is disadvantageous to the new system.

Second, import risks are not yet assessed systematically or comprehensively. The new biosecurity policy requires all produce or risk material not on the white list (i.e. not subject to agreed import health standards) to be submitted for import risk assessment (IRA) but, in common with many small island nations, St Helena lacks the domestic technical expertise to apply the international guidelines on pest risk analysis (IPPC, 2017). Biosecurity officers cannot refer to specialist networks for advice on risk likelihood and impact, assessments of which are required at each level of the IRA process. They are often too busy to attempt these formal assessments themselves. Yet, under pressure of requests to import new commodities, St Helena's biosecurity officers regularly have to make such decisions.

Pragmatic guidelines for IRA are being applied in the interim. Risk evaluation for familiar commodities can rely on levels of confidence acquired through practical experience and knowledge of their points of origin. For example, fresh produce from South Africa, vehicles from Ascension Island, UK or South Africa, and selected plant propagating materials from the UK or South Africa are relatively well known and already have import conditions defined for them. These conditions must be revised if the risk profile alters through, for example, a change in pathway or reports of a new pest or disease in the country of origin. For commodities of these sorts, the biosecurity team assess new risks using simple web-based resources such as the CABI Invasive Species Compendium (<https://www.cabi.org/isc/>) and CABI Crop Protection Compendium (<https://www.cabi.org/cpc/>).

Biosecurity St Helena does not have risk assessment measures in place for unknown pathways or commodities such as novel plant or animal species imported for propagation or breeding. These are highly concerning and challenging to address.

The key factor for success in socialising biosecurity is considered to be the amount of time and effort committed to listening, talking and responding to the community, from farmer to government official, and utilising a range of communication media. No attempt was made to directly tackle the few more resistant individuals with arguments. It was found that time and peer pressure were in most cases sufficient to bring them round, and compliance was high.

In conclusion, St Helena Island faces increasing pressures from invasive species and is typical of small island nations in having too few resources to cope. Despite this, it has risen to the challenge and has in place a model for autonomous biosecurity by a small island nation. What has made this possible – and what compensates so significantly for chronic resource stresses – is the decision to engage business and local communities in developing *Biosecurity St Helena* and sustaining it day-by-day. Harmonising of public services has been highly effective. Recognising that a solely official approach to *Biosecurity St Helena* would lack necessary resilience and buy-in, the socialising of biosecurity is what makes *Biosecurity St Helena* a model for other small island nations.

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REFERENCES

Churchyard, T., Eaton, M., Hall, J., Millett, J., Farr, A., Cuthbert, R. and Stringer, C. (2014). *The UK's Wildlife Overseas: A Stocktake of Nature in our Overseas Territories*. Sandy, UK: RSPB.

Gratz, N.G., Steffen, R. and Cocksedge, W. (2000). 'Why aircraft disinsection?' *Bulletin of the World Health Organization* 78 (8): 995–1004.

IPPC (2017). 'International Standards for Phytosanitary Measures, ISPM No. 11'. *Pest Risk Analysis for Quarantine Pests, Including Analysis of Environmental Risks and Living Modified Organism*s. Produced by the Secretariat of the International Plant Protection Convention Adopted 2013. <https://www.ippc.int/static/media/files/publication/en/2017/05/ISPM_11_2013_En_2017-05-25_PostCPM12_InkAm.pdf >.

Key, G.E. (2013). *Biosecurity Status Report*. St Helena: Agriculture and Natural Resources Division, St Helena Government.

Pryce, D. (2015). 'St Helena Pest and Biocontrol Species List'. <http://www.sainthelena.gov.sh/integrated-pest-management/ >.

Shine, C. and Stringer, C. (2010) *South Atlantic Invasive Species Strategy and Action Plan. Based on the Contributions by Participants to a Regional Meeting, Ascension Island, 14–19 May 2009*. Unpublished report.

St Helena Government (2014). *Biosecurity St Helena. A Policy for All Members of the St Helena Community*. St Helena: Agriculture and Natural Resources Division, St Helena Government.

St Helena Government (2016). *St Helena 2016 Population & Housing Census. Summary Report*. St Helena: St Helena Statistics Office, St Helena Government.

Proactive planning and compliance for a high-priority invasive species rapid response programme

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Abstract The California Channel Islands (USA) are home to numerous endemic taxa and provide refugia for many Californian coastal species. Over recent decades, land managers have made considerable progress in eliminating alien invasive mammals and non-native invasive plants and ants. The California Islands Biosecurity Group aims to protect that investment, by coordinating and finding economies of scale in efforts to prevent invasive species introductions, detect incursions, and to respond rapidly and successfully to incursions that do occur. The highest priority is the prevention or early detection and elimination of *Rattus* introductions to any of the rat-free islands or islets in the archipelago. The rapidity of a response to a suspected incursion of rats could be hindered not only by the time required to detect it and mobilise the field effort, but also by the time required to secure regulatory permissions to implement the pre-planned actions. To address these issues, we are proactively obtaining necessary permits and developing strategy that could allow response action within 36 hours of a known or suspected introduction of *Rattus*. Our process includes developing a flowchart to aid managers in determining the appropriate management response, which considers seasonality, location, and potential non-target taxa at the incursion site. Completing the necessary compliance process proactively, and maintaining a ready stock of the materials needed to equip a management response, reduces the time between detecting an incursion and being able to respond to it and thereby increases our ability to protect the conservation value of – and investment in – these islands.

Keywords: biosecurity, California Channel Islands, eradication, planning, rapid response, rat introduction

INTRODUCTION

The extraordinary ecological, economic, and cultural damage invasive alien species can cause on islands, and the high costs and challenges of controlling or eradicating these species, has led many island managers to place greater attention on efforts to prevent biological invasions in the first place (Ruiz & Carlton, 2003; Broome, 2007; Rout, et al., 2011; Bassett, et al., 2016). Biosecurity programmes are designed to proactively prevent invasive species from arriving on an island, detect introductions quickly, and prevent non-native species from establishing on islands via rapid response actions (Broome, 2007; Russell, et al., 2008a; Russell, et al., 2008b). The ability to rapidly detect and respond to incursions is important for a variety of reasons, including that small populations are less expensive and more tractable to eradicate than large ones; some invasive species may not be eradicable at all with existing methods once even a small population has become established. The question of "how rapid does rapid need to be?" is to some degree determined by the biology of the invader. The urgency of detecting and responding to an invasive plant versus an insect or a rodent may vary, for example, because their population growth and patterns of spread may be different. Understanding those differences is important for allocating scarce management resources, as costs of surveillance are likely to increase if finer temporal scale data are needed to provide adequate detection. Costs of interventions may also increase as infestations increase in area and become more established.

This paper focuses on one critical window of time in a biosecurity strategy: the period between detecting a potential incursion and responding to it with management actions. Specifically, we provide a biosecurity programme case study from an archipelago of islands in the United States which has identified priority invaders and invasion scenarios, proactively planned responses, and sought environmental compliance permits in advance so that planned actions can be carried out quickly if an incursion is detected. Risks of rat invasion on islands are often high, because rats are common stowaways on large boats. If they successfully invade and establish populations on some islands, then they may not be eradicable with existing technologies for a variety of reasons, including the potential that available treatment options may pose unacceptable risks to populations of non-target native species. Meanwhile, animals at very low abundance can be difficult to detect (Morrison, et al., 2007; Russell, et al., 2008a). Further compounding that challenge in the context of biosecurity is that some species (e.g. rats) may be unusually mobile and wide-ranging and exhibit other unexpected behaviours in novel environments, and when their numbers are low (Russell, et al., 2008a). Thus, if a rat is detected or suspected (e.g. if there was a shipwreck of a vessel known to be infested), there may be little time to respond in a localised area with relatively high confidence that the animal remains within the project area. A variety of factors can limit response time, ranging from technical (e.g. determining the specific methods which would be most effective under the particular circumstances of the incident), to operational (e.g. getting necessary materials to the incident location), to administrative (e.g. who would make decisions within the institutions with jurisdictions over the proposed response). Environmental review and compliance processes are time consuming for land managers and, combined with required public comment periods in the national permitting process, limit the ability to respond rapidly. Here, we describe how the California Channel Islands Biosecurity Working Group has taken steps to improve the ability of conservation managers to respond quickly to a rat incursion and improve their chances of eliminating it. The most important of these steps are the advance completion of required environmental review and permitting, and the staging of materials necessary for a response to a potential rat incursion.

SITE DESCRIPTION

The Channel Islands encompass eight islands ranging in size from 250 ha to 25,000 ha, as well as numerous islets around them, all of which are within 100 km of the southern California mainland (USA). Five of the islands are included in Channel Islands National Park (Park). The US Navy owns one of the islands in the Park (San Miguel);

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The Nature Conservancy owns 76% of the largest Park island (Santa Cruz); and the National Park Service owns the remainder. Four of the five Park islands are rat-free, as is an islet separated from the fifth island by just 700 m. The islands provide important nesting habitat for seabirds, including five with either state or federal designations, and one federally threatened shorebird (McChesney & Tershey, 1998; Howald, et al., 2005; CDFW, 2017a). They are home to 14 federally threatened plant species and 23 endemic animal species. The three largest islands in the Park are home to the island fox (*Urocyon littoralis*), which was recently removed from the endangered species list (USFWS 2016). The islands have been the focus of significant conservation and restoration efforts over recent decades to remove the most destructive introduced invasive species, including pigs, goats, sheep, horses, donkeys, rabbits, rats, cats, and ongoing efforts are underway to eradicate Argentine ants (*Linepithema humile*) and 32 weed species from Santa Cruz Island (Morrison, 2007, Morrison, 2011; Cory & Knapp, 2014; McEachern, et al. 2016; Boser, et al., 2017).

In order to protect the investments made and the significant biological and cultural values of these islands – and the broader archipelago in which they sit – the California Islands Biosecurity Working Group (Group) was established in 2012 (Boser, et al., 2014). The Group is composed of biologists and managers from the federal agencies, non-profit institutions, and partners that own and or have management responsibility or investment in the island resources. The Group meets quarterly in person or by phone to share updates on obstacles, technological and logistical advances to improve biosecurity, and to suggest improvements to current biosecurity practices and education programmes for professionals, visitors and resident populations. A central organising principle of the Group is that even though there are discrete islands with various jurisdictions across the archipelago, a coordinated and collaborative approach to biosecurity enhances effectiveness and achieves a myriad of efficiencies (Boser, et al., 2014; Matos, et al., in press). To accomplish an economy of scale on biosecurity priorities and to ensure that group objectives are met, the landowning entities jointly fund a full-time position to lead the implementation of group objectives. Staff biologists for the U.S. Navy, the National Park Service and The Nature Conservancy form sub-working groups to deliver results to jointly relevant projects including camera traps to detect invasive species, checks of boats and planes departing for the islands, and development of educational materials and messaging. A top priority of the Group is preventing the establishment of rats on rat-free islands, because of the well documented threats rats pose to island ecosystems, including on the California Channel Islands (Atkinson, 1985; Campbell & Atkinson, 2002; Jouventin, et al., 2003; *Howald, et al., 2005;* Towns, et al., 2006; Jones, et al., 2008; Banks & Hughes, 2012). However, given current available technologies, significant advance preparation would be required if we were to mount a timely and effective response to a rat incursion.

THE URGENCY OF RESPONSE

A comprehensive biosecurity programme will include measures to prevent incursion, as well as plans to respond to an incursion should one occur. The rapidity of the response action needs to be tailored to the species of concern. For rats, field work conducted by Russell, et al. (2008a) assessing the behaviour of collared individual rats introduced to a rat-free island indicated that most stayed within the introduction area for 2–3 days, but after that time could rapidly move away from that site. We interpreted their data to indicate that if a pregnant rat or a small population of rats were introduced to an island, there would be a very narrow window of time to take action in a localised area, and expect with some certainty that the rat(s) would be near the introduction point (Russell, et al., 2008a; Russell, et al., 2008b). For incursions on large islands (e.g. an island like 250,000 ha Santa Cruz Island), once rats have dispersed from the introduction point, we may consider them not possible to remove with current technologies, or consider that impacts to non-target species on an island-wide scale may be too great. Only a few islands larger than 10,000 ha have successfully completed a rat eradication (Howald, et al., 2007), the largest effort being recently conducted on South Georgia Island at approximately 390,000 ha (pers. comm, T. Martin 2017), and each of these used broadcast rodenticides. While rodenticides are currently the most effective tool to eradicate rat infestations (Kaudeinen & Rampaud, 1986; Tershy, et al., 1997; Howald, et al., 2007), they can have substantial non-target impacts on native species (Kaukeinen & Rampaud, 1986; Brown, et al., 1988; Eason & Spurr, 1995; Eason, et al., 2002; Howald, et al., 2010). This underscores the importance of having confidence that a suspected rat infestation remains contained within an area small enough so that risks to non-target native species are acceptably low if rodenticides are utilized. Quick deployment and response following an introduction increases that confidence. Efforts to collect data if rodenticides are used would include genetic analysis of target carcasses collected during post-treatment actions, collection and toxicology analyses of non-target carcasses discovered for at least one year, and a comparison of non-target population data pre- and post-treatment for at least five years.

One programme planning goal for the Group is to have the ability to react to an incursion within 36 hours. In the United States, however, environmental review and compliance documents permitting action that may impact non-target species, such as rodenticide use in conservation areas with sensitive or endangered species, can take months to years to complete. Given the importance of both the environmental review process and the need for a rapid response after an incursion, we sought to undertake the planning and permitting processes in advance, so that if an incursion does occur we are prepared to react appropriately and quickly.

ENVIRONMENTAL REVIEW AND COMPLIANCE

The United States' 1969 National Environmental Policy Act (NEPA) established a framework for protecting the environment from ill-considered actions by ensuring that federal agencies of the United States integrate environmental values into their decision-making processes before taking federal action. The Channel Islands National Park and the U.S. Navy islands are federal properties and therefore an environmental review must be completed before actions that could impact their environments, such as rodenticide use, could be taken. A NEPA document contains assessments of alternative actions (referred to as Alternatives) which could be implemented to achieve a stated objective that are created in consultation with subject matter experts. Further, the analysis chapter reviews the impacts of alternatives on the natural and cultural resources within the action area. A "preferred alternative" is selected by the lead agency after the document is released for public review and comment, revised if necessary, and one alternative is selected for implementation by the lead agency.

Prior to undertaking the environmental review process and structuring an environmental compliance document, we reviewed other scenarios that require proactive planning such as emergency scenarios where human life or property is at stake. In California, the Department of

Fish and Wildlife has published a California State Oil Spill Contingency plan (CDFW, 2017b) that outlines and permits actions required for rapid responses to oil spills. Similar to a rat introduction on an island, the location of the oil spill is not known during the planning process, so planning efforts must include and address a variety of contingencies that may be utilised depending on the timing and location of the spill. In rat response planning and compliance documents, we similarly need to evaluate locational- and seasonally-dependent scenarios. This includes a thorough assessment of how proposed actions may impact specific biological and cultural resources on the five islands included in the action area.

Although this proactive document must plan for the introduction event in an unknown location and time, it must nonetheless provide enough detail that the proposed response actions can be thoroughly assessed by the public and subject matter experts. Due to the ambiguity around the time or location of an introduction, and the biological or cultural resources that may be present at the site at any given time, the alternatives in a proactive compliance document must be structured differently than is typical in NEPA documents. The preferred alternative must encompass all feasible response actions, from the most minimal actions such as deploying remotely triggered cameras, to setting rodent traps, and/or broadcasting rodenticide. Proactively permitting each of these actions would allow managers to appropriately scale their response and utilise the tools that are appropriate under the specific circumstances at that time and place. The creation and use of a flowchart which directs decision-makers to recommended actions based on the known resources in a proposed project area and season allows the managers of the incident to quickly identify a recommended response. If agreed to by all consulting agencies prior to the emergency, the flowchart could be used to rapidly recommend response actions so they can be approved and quickly enacted. A contact list, similar to those used in incident response plans, must be created prior to an incident to maximise the likelihood of rapid action.

Depending on the proposed action and the resources in the affected area, additional federal and state laws may apply. In the California Channel Islands, the protected status of resident bald eagles (*Haliaeetus leucocephalus*) and numerous protected migratory birds require managers to adhere to regulations in the Bald and Golden Eagle Protection Act (1940) and the Migratory Bird Treaty Act (1918). The status of Channel Islands National Park as proposed wilderness under the Wilderness Act (1964) requires agency staff to complete an assessment of impacts to "wilderness character" in a process structured similarly to a NEPA review. The California Environmental Quality Act (1970) is similar in scope to NEPA, and a California Environmental Quality Act compliance document is required if the project is conducted on state lands or if a project uses state money. The federal Endangered Species Act (1973) and the California Endangered Species Act (1970) lists endangered and threatened species and additional permitting may be required if these species are present in the affected area. The sheer scope of the assessments and review that must occur to adhere to federal and state laws designed to protect natural resources illustrates the need to develop a functional tool to proactively gain consensus on the need to take emergency actions to protect the resources these laws were designed to protect.

CONSIDERATIONS IN PRE-STAGING MATERIALS

The compliance document must describe and accommodate assumptions about how tools needed to respond to a rat incursion could be staged for rapid deployment. For instance, if a potential action described in the environmental review document calls for the use of a conservation rodenticide, we could consider a brodifacoum-based conservation pellet designed for use on islands by Bell Laboratory which has been shown to be effective at eradicating rats from islands (Kaudeinen & Rampaud, 1986; Tershy, et al., 1997; Howald, et al., 2007). This specialised bait must be ordered months before it could be used, because it is only manufactured every few months. The rodenticide loses palatability after one year, so the bait must be properly disposed of and reordered annually. The type of bait packaging we might use would depend on the planned staging and deployment method approved in the compliance document, whether it be broadcast deployment and thus must be loaded into a hopper on the mainland and slung out to the island preloaded or packaged for transport by boat and loaded into a hopper on the affected island. For implementation to go smoothly, contingency contracts for services need to be in place prior to any incursion. A rapid deployment of rodenticide to an incursion site is dependent on all technical and logistical parts of an operation being pre-approved and permitted.

ASSESSMENT OF NON-TARGET IMPACTS

The permitting documents required in the United States include a description of projected impacts on non-target species, services such as transportation, and systems such as air and water quality. These anticipated impacts can range from "none" to "major" (the latter typically defined as population-level impacts). Monitoring programmes for bald eagles, peregrine falcons (*Falco peregrinus*), island foxes, island scrub-jays (*Aphelocoma insularis*), island spotted skunks (*Spilogale gracilis amphiala*), and seabirds are implemented annually on these islands, providing us with data on the distribution and abundance of these species on each island. Based on this detailed information, we can annually estimate how many individuals and what percentage of the population we could expect to be impacted by response actions and then use this information to recommend appropriate minimisation measures. The estimates can assist managers in assessing the risk of taking a specific action rather than taking no action at a site. For instance, the 2016 fox density estimate on Santa Cruz Island is approximately 10 foxes per square kilometre (unpublished data, A. Dillon, Colorado State University). If an area of 60 ha is treated, with the possibility of an additional 40 ha within a 200 m buffer zone, the impacted area may be as large as one square kilometre and approximately 10 foxes may be impacted by broadcasting bait. The island's total fox population is estimated to be 2,100 foxes (unpublished data, A. Dillon, Colorado State University) and thus we could expect that action to impact 0.5% of the total population. However, we could require minimisation measures that could include fox trapping in the affected area immediately before treatment to remove as many individuals as possible for translocation to lower density areas of the island. Island foxes are easily trapped using box traps, and we expect we could remove as many as 5–8 foxes from the affected area with just one night of trapping. Similar calculations using known home range data and population estimates for raptors could be used to determine worst-case scenarios if rodenticide is to be broadcast and also if bait stations are to be used. These estimates would also include risk of transient birds entering the treatment zone, possibly in response to availability of contaminated carcasses. The output of that analysis may assist managers in deciding on the best tool to use after an incursion at a specific site. We are likely to assume 100% mortality of the native mouse population in the project area and a buffer zone if broadcast rodenticide is used. These calculations and considerations must be built into an action flowchart which could be used by managers and federal agencies to assess the ecological cost of a broadcast baiting response action relative to the likelihood and ecological cost of a rat population establishing on the island.

CONCLUDING REMARKS

The benefits of proactive planning and rapid response to biodiversity protection and ecosystem function are clear-cut. Substantial evidence exists which suggests that if rats establish themselves on new California Channel Islands, they would have population-level effects on seabirds (Howald, et al., 2005) and potentially many of the listed plant species (Corry & McEachern, 2009). The approach we used to increase the ability to undertake rapid response – via proactive environmental review and compliance and having at the ready the necessary materials to respond – could be followed for other islands in the USA and, with appropriate modifications, for islands in other nations.

Although we have made substantial progress in our ability to respond rapidly to a rat detection, we still face a significant limitation in our ability to detect new incursions in a timely manner. This is due in large part to the size, ruggedness, and inaccessibility of the Channel Islands. Although we have a camera array at sites of suspected higher risk of incursion, the frequency at which we can retrieve data from these cameras, and process the images they capture, represents an important weakness in our current programme. We are hopeful that with emerging technologies – in particular mobile and networking technologies – we will be able to retrieve these data in real time (Pimm, et al., 2015). Advances in machine learning and image recognition (Lillesand, et al., 2014) also can be applied to speed up processing of images and flag suspicious images promptly to managers.

Rapidly developing technologies may play another role as we adaptively improve our planning documents. Specifically, we are considering the effect of emerging molecular methods of rat control on how we might assess risk and uncertainty, especially with regards to evaluation of non-target impacts. For example, even though rats, if established, would have negative impacts on many native species of the islands, the most successful tool currently used to eliminate rats on islands, rodenticide, has impacts to a broad array of non-target taxa (Kaukeinen & Rampaud, 1986; Brown, et al., 1998; Eason & Spurr, 1995; Eason, et al., 2002; Howald, et al., 2010). Alternative technologies such as gene drives that produce "daughterless" offspring may be available for use on invasive mammals in the relatively near term (Regalado, 2017). Such technology has already been developed for some species of insects (Gantz, et al., 2015). While there remains uncertainty about whether such technologies could accomplish eradication objectives, or would be suitable for use in low-density populations, substantial progress has been made towards advancing the technology in the past two years (Committee on Gene Drive Research in Non-Human Organisms, 2016; Regalado, 2017). Clearly, there are numerous and complex ecological, ethical, philosophical, and policy issues associated with field application of these technologies and identifying and resolving those issues will present new planning and permitting considerations, constraints and timelines. However, given the pace of developing technologies, and the known concerns with existing technologies, biologists should weigh current ecological costs of action against the likelihood of future technologies becoming available and the possibility that their use would ultimately provide better ecological outcomes. For example, a rapid response using existing technologies might be advisable – even in the face of non-target impacts – if catastrophic and irreversible damage to native species, such as extinction or a severe reduction in genetic diversity, were deemed likely to result before new rat-control technologies could be available for use, even under optimistic scenarios.

The proactive planning and permitting approach we outline can be applied broadly to conservation challenges that require the ability to respond rapidly to foreseen episodic or biologically threatening emergencies. We recognise that in island ecosystems, that experience high extinction rates and frequent state changes, managers must be nimble and quickly direct management actions for the preservation of biodiversity. The model we outline to proactively invest in planning, reviewing, and permitting essential biosecurity response measures, will improve our ability to protect the native biota of islands in the USA, if not worldwide.

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REFERENCES

Atkinson, I.A.E. (1985). 'The Spread of Commensal Species of *Rattus* to Oceanic Islands and their Effects on Island Avifaunas'. In: P.J. Moors (ed.) *Conservation of Island Birds: Case Studies for the Management of Threatened Island Species*, pp. 35–81. Cambridge: International Council for Bird Preservation.

Banks, P.B. and Hughes, N.K. (2012). 'A review of the evidence for potential impacts of black rats (*Rattus rattus*) on wildlife and humans in Australia'. *Wildlife Research* 39: 78–88.

Bassett, I.E., Cook, J., Buchanan, F. and Russell, J.C. (2016). 'Treasure islands: biosecurity in the Hauraki Gulf Marine Park'. *New Zealand Journal of Ecology* 40: 250–266.

Boser, C.L., Cory C., Faulkner K.R., Randall J.M., Knapp J.J. and Morrison S.A. (2014). 'Strategies for biosecurity on a near-shore island in California'. *Monographs of the Western North American Naturalist* 7: 412–420.

Boser, C.L. Hanna, C., Holway, D.A., Faulkner, K.R., Naughton, I., Merrill, K., Randall, J.M., Cory, C. Choe, D.-H. and Morrison, S.A. (2017). 'Protocols for Argentine ant eradication in conservation areas'. *Journal of Applied Entomology* 141: 540–550.

Broome, K. (2007). 'Island Biosecurity as a Pest Management Tactic in New Zealand'. In: G.W. Witmer, W.C. Pitt, and K.A. Fagerstone (eds.) *Managing Vertebrate Invasive Species: Proceedings of an International Symposium*, pp. 104–106. Fort Collins, Colorado: National Wildlife Research Center.

Brown, R.A., Hardy, A.R., Greig-Smith P.W. and Edwards, P.J. (1988). 'Assessing the impact of rodenticides on the environment'. *EPPO Bulletin* 18: 283–292.

Campbell, D.J. and Atkinson, I.A.E. (2002). 'Depression of tree recruitment by the Pacific rat (*Rattus exulans* Peale) on New Zealand's northern offshore islands'. *Biological Conservation* 107: 19–35.

[CDFW] California Department of Fish and Wildlife. (2017a). *Natural Diversity Database. Special Animals List*. Periodic publication.

[CDFW] California Department of Fish and Wildlife. (2017b). *California State Oil Spill Contingency Plan*. California Department of Fish and Wildlife Office of Spill Prevention and Response.

Committee on Gene Drive Research in Non-Human Organisms. (2016). 'Recommendations for Responsible Conduct; Board on Life Sciences; Division on Earth and Life Studies'; *National Academies of Sciences*. Washington, DC: National Academies Press.

Corry, P.M. and McEachern, A.K. (2009). 'Patterns in Post-grazing Vegetation Changes Among Species and Environments. San Miguel and Santa Barbara Islands'. In: C.C. Damiani and D.K. Garcelon (eds.) *Proceedings of the 7th California Islands Symposium* pp. 201–214. Arcata, California: Institute for Wildlife Studies.

Cory, C. and Knapp J.J. (2014). 'A program to eradicate twenty-four nonnative invasive plant species from Santa Cruz Island'. *Monographs of the Western North American Naturalist* 7: 455–464.

Eason, C.T. and Spurr E.B. (1995). 'Review of the toxicity and impacts of brodifacoum on non-target wildlife in New Zealand'. *New Zealand Journal of Zoology* 22: 371–379.

Eason, C.T., Murphy E.C., Wright G.R.G. and Spurr, E.B. (2002). 'Assessment of risks of brodifacoum to non-target birds and mammals in New Zealand'. *Ecotoxicology,* 11: 35–48.

Gantz, V.M., Jasinskiene, N., Tatarenkova, O., Fazekas, A. Macias, V.M., Bier, E. and James, A.A. (2015). 'Highly efficient Cas9-mediated gene drive for population modification of the malaria vector mosquito *Anopheles stephensi'*. *Proceedings of the National Academy of Sciences* 112: E6736–E6743.

Howald, G.R., Faulkner K.R., Tershy B., Keitt B., Gellerman H., Creel E.M., Grinnel M., Ortega S. and Croll D.A. (2005). 'Eradication of black rat from Anacapa Island: biological and social considerations'. In: D. Garcelon and K Schwemm (eds.) *Proceedings of the Sixth California Islands Symposium*, pp. 299–312. Arcata, California: Institute for Wildlife Studies.

Howald, G., Donlan, C.J., Galván, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Howald, G., Donlan, C.J., Faulkner, K.R., Ortega, S., Gellerman, H., Croll, D.A. and Tershy, B.R. (2010). 'Eradication of black rats *Rattus rattus* from Anacapa Island'. *Oryx* 44: 30–40.

Jones, H.P., Tershy, B.R., Zavaleta, E.S., Croll, D.A., Keitt, B.S., Finkelstein, M.E. and Howald, G.R. (2008). 'Severity of the effects of invasive rats on seabirds: a global review'. *Conservation Biology* 22: 16–26.

Jouventin, P., Bried, J. and Micol, T. (2003). 'Insular bird populations can be saved from rats: a long-term experimental study of white-chinned petrels *Procellaria aequinoctialis* on Île de la Possession (Crozet archipelago)'. *Polar Biology* 26: 371–378.

Kaudeinen, D.E. and Rampaud, M. (1986). 'A review of brodifacoum efficacy in the U.S. and worldwide'. In: T. Salmon (ed.) *Proceedings of the Twelfth Vertebrate Pest Conference,* pp. 16–50. Davis: University of California.

Lillesand, T., Kiefer, R.W. and Chipman, J., (2014). *Remote sensing and image interpretation*. USA: John Wiley & Sons.

Matos, J., Little, A. Broome, K., Kennedy, E., Méndez Sánchez, F, Latofski-Robles, M., Irvine, R., Gill, C., Espinzoa, A., Olthof, K., Ball, M., Howald, G. and Boser, C.L. (In press). 'Connecting island communities on a global scale: Case studies in island biosecurity'. *Western North American Naturalist.*

McChesney G.J. and Tershy, B.R. (1998). 'History and status of introduced mammals and impacts to breeding seabirds on the California Channel and northwestern Baja California Islands'. *Colonial Waterbirds* 21: 335–347.

McEachern, K., Atwater, T., Collins, P., Faulkner, K. and Richards, D.V. (2016). 'Managed Island Ecosystems'. In: H. Mooney and E. Zavaleta (eds.) *Ecosystems of California,* pp. 755– 778. University of California Press, Oakland, California.

Morrison, S.A. (2007). 'Reducing Risk and Enhancing Efficiency in Non-native Vertebrate Removal Efforts on Islands: A 25-year Multi-taxa Retrospective from Santa Cruz Island, CA'. In: G.W. Witmer, W.C. Pitt and K.A. Fagerstone (eds.) *Managing Vertebrate Invasive Species: Proceedings of an International Symposium*, pp. 398–409. Fort Collins, Colorado: National Wildlife Research Center.

Morrison, S.A., Macdonald, N., Walker, K., Lozier, L. and Shaw, M.R. (2007). 'Facing the dilemma at eradication's end: uncertainty of absence and the Lazarus effect'. *Frontiers in Ecology and the Environment* 5: 271–276.

Morrison, S.A. (2011). 'Trophic considerations in eradicating multiple pests'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 208–212. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Pimm, S.L., Alibhai, S., Bergl, R., Dehgan, A., Giri, C., Jewell, Z., Joppa, L., Kays, R. and Loarie, S. (2015). 'Emerging technologies to conserve biodiversity'. *Trends in Ecology & Evolution* 30: 685–696.

Regalado, A. (2017). 'First gene drive in mammals could aid vast New Zealand eradication plan'. *MIT Technology Review* [online periodical]. <www.technologyreview.com/s/603533/first-gene-drive-in-mammals-could-aid-vast-new-zealand-eradication-plan/>. Accessed 15 June, 2017.

Rout, T.M., Moore, J.L., Possingham, H.P. and McCarthy, M.A. (2011). 'Allocating biosecurity resources between preventing, detecting, and eradicating island invasions'. *Ecological Economics* 71: 54–62.

Ruiz, G.M. and Carlton, J.T. (2003). *Invasive Species: Vectors and Management Strategies*. Washington, DC: Island Press.

Russell, J.C., Beaven, B.M., MacKay, J.W.B., Towns, D.R. and Clout, M.N. (2008a). 'Testing island biosecurity for invasive rats'. *Wildlife Restoration* 35: 215–221.

Russell, J.C., Towns, D.R. and Clout, M.N. (2008b). 'Review of Rat Invasion Biology: Implications for Island Biosecurity'. *Science for Conservation 286*. Wellington, New Zealand: Department of Conservation.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?' *Biological Invasions* 8: 863–891.

Tershy, B. R., Croll, D.A. and Howald, G.R. (1997). 'Options for Removing Introduced Black Rats (*Rattus rattus*) from Anacapa Island, Channel Islands National Park'. Unpublished Report, Island Conservation and Ecology Group, Santa Cruz, CA.

[USFWS] US Fish and Wildlife Service. (2016). 'Endangered and threatened wildlife and plants; Removing the San Miguel Island fox, Santa Rosa Island fox, and the Santa Cruz island fox from the Federal list of endangered and threatened wildlife, and reclassifying the Santa Catalina Island fox from endangered to threatened'. *Federal Register 81(156)*: 53315–53333.

How do we prevent the obstacles to good island biosecurity from limiting our eradication ambitions?

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Abstract Island pest eradication and biosecurity spring from the same imperative. Yet in practice biosecurity is often subordinated to the demands of eradication in planning processes so that it is not well prepared at the outset and is poorly sustained in the aftermath. Why is this when, by any logic, eradication should not proceed unless the means are in place to safeguard treated islands from renewed pest invasions? We draw on New Zealand's conservation experience to explore key reasons why biosecurity tends to lack the muscle and stamina required to protect eradication investments and their priceless biological pay-offs. We take biosecurity to be an unassailable good and discuss remedies to the challenges it must overcome if shortfalls and failures are not to become brakes on eradication ambitions. We think that eradicating pests from islands is essentially a technical business and is thus the easier part of the pest-free equation. In contrast, the problems associated with keeping islands pest-free in perpetuity are inherently more human than technical, though the latter are taxing enough. Certainly, biosecurity's work is more open-ended, less glamorous and not so dramatically rewarded. A primary concern is that the complex social dimensions of biosecurity are usually neglected. We attribute this to limited investment in social psychology tools and our habit of relying on biologists to manage behaviour-change in island visitors of all kinds. As a result, biosecurity effort can be distorted by an over-determined emphasis on surveillance and incursion response—an unduly reactive strategy but more familiar technical ground than quarantine which is inherently human-oriented. Our discussion of what works and what does not for these issues does not claim to offer universal remedies. In addition to protecting existing assets competently, our aim is to prepare biosecurity to serve tomorrow's eradication ambitions, particularly those on the testing new frontier of pest removal from inhabited islands.

Keywords: behaviour-change, complacency, improvement, priority, social psychology

INTRODUCTION

In recent decades, we have witnessed an expansion in the scale and biogeographic reach of projects to eradicate invasive organisms from islands (Howald, et al., 2007; DIISE, 2015). This diversifying of effort has consolidated eradication as a requisite measure in island restoration strategies. Successes have given comforting cues to funders and sponsors contemplating eradication as a conservation investment (see <www.milliondollarmouse.org.nz>). Eradication ambitions are clearly intensifying worldwide as operational experience accumulates and scalable advances in tools and methods overcome technical constraints on feasibility (<www.iucn.org/theme/species/our-work/invasive-species/honolulu-challenge-invasive-alien-species/commitments-towards-achieving-honolulu-challenge>; <www.predatorfreenz.org/>).

The rise of eradication as a restoration tool argues for a closely correlated strengthening of biosecurity functions (Russell, et al., 2007). These two lines of work define the mutually dependent parts of the pest-free equation. Without effective quarantine, surveillance and invasion response in place to minimise the likelihood of pests recolonising, it is reckless at best, and futile at worst, to proceed with eradication. If island stewardship lacks good biosecurity, defence of very substantial resource investments and biological pay-offs becomes a lottery in which pest organisms dictate the odds.

Biosecurity is arguably more potent than eradication as a restoration tool since most subsequent conservation investments make sense only if the island remains pest-free. Thus, biosecurity is the cornerstone, not an adjunct, of eradication work (see the panel below). Yet, however its priority is framed, we see uncomfortable signals in rates of pest invasion and reinvasion on protected islands (Clout & Russell, 2006; DIISE, 2015; Vincent, 2017) that biosecurity's practices and mind-sets have not advanced adequately to meet eradication's expectations of them.

Recurrent or not, some invasions will have been inevitable. Near-shore sanctuaries will always be vulnerable. Nor can we always predict the behaviour of animal pests or the distances over which they can travel by their own means (Veale, 2012). But other lapses will have had preventable causes: inadequate resourcing, limited preparation, hesitant follow-through, blind spots in coverage, poorly developed tools and practices, cultures of complacency, or simply outright neglect. These problems signify that operational planners have not given biosecurity the priority it requires.

These shortfalls are amenable to remedy because they arise from judgments humans make. Uncorrected, they result in biosecurity which is more often than not poorly sustained once eradication work concludes. Until the limitations in current practices are addressed, poor biosecurity will frustrate eradication ambitions.

Island restoration experience in New Zealand confirms these conclusions. In this paper, we discuss key reasons in the New Zealand context why biosecurity tends to lack the muscle and stamina of its indivisible eradication twin. These issues relate more to the social and psychological dimensions of biosecurity than to the technical challenges we face in upgrading our tools and methods, though they, too, are taxing enough.

In eradication literature, appeals for effective island biosecurity regimes focus overwhelmingly on the mechanics of pest interception. If mentioned, awareness-raising through educational outreach is usually promoted as the means to invoke helpful biosecurity behaviour in island communities and other public audiences (see for example, Boser, et al., 2014). But a growing body of empirical studies shows that heightened awareness does not necessarily alter behaviour (McKenzie-Mohr, 2013). The cryptic attitudinal barriers to biosecurity uptake are rarely assessed in public audiences (Bassett, et al., 2016) or indeed within the ranks of conservation practitioners. These social matters have immediate bearing on biosecurity's effectiveness.

We conclude our appraisal of obstacles to good biosecurity with a brief review of the measures we are

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taking to address them. Though the problems are likely to be common to most biosecurity contexts, we do not claim to offer universal remedies. Our overall aim is to protect today's pest-free islands more effectively and to prepare biosecurity to serve tomorrow's eradication ambitions, particularly those on the new frontier of permanently inhabited islands. Here the social challenges are amplified and more diverse.

THE NEW ZEALAND CONTEXT

New Zealand's Government has overall responsibility for funding pre- and post-border biosecurity. The Ministry for Primary Industries (MPI) takes the lead under the Biosecurity Act 1993 to manage pest threats to human health, the economy and environment (<http://www.mpi.govt.nz/law-and-policy/legal-overviews/biosecurity/>). Under this influential Act MPI can delegate biosecurity duties to other central and local government agencies.

In the day-to-day division of biosecurity labour, the Department of Conservation (DOC) has customarily acted as the sentinel and conscience for protection of biodiversity values. Obligations to protect valued islands from invasive pests are explicit in its own mandating legislation, the Conservation Act 1987 (<http://legislation.govt.nz/>).

Today, DOC has biosecurity obligations to more than 400 pest-free islands. Up to 240 of these have been cleared historically or in more recent times of pest organisms (DIISE, 2015). Others have never been invaded. Tenure varies from public land administered by local or central government to fully freehold.

This trusteeship of recognised sanctuaries and others in-the-making extends from the Kermadec Islands in the sub-tropics to Campbell Island in the high southern latitudes. Islands at the extremities are generally well protected by isolation and strict controls on access. Conversely, others closer to New Zealand's main landmasses are within easier reach of humans and commensal pests.

Nationwide, about 85 of DOC's 2000 staff contribute in some way to island biosecurity. Typically, the majority operate part-time as gate-keepers screening traffic to and between pest-free islands. A small but growing number are also handlers of pest-detection dogs—graduates of DOC's rigorous certification programme (DOC, 2015). Rangers residing full-time on New Zealand's signature pest-free islands are obligate biosecurity gate-keepers.

Within DOC, island biosecurity operations are supported in three ways. Two national advisors lead a well-defined improvement programme discussed shortly in this paper. They negotiate for social and technical research too. Specialist community rangers organise public outreach throughout the country. And eradication veterans on DOC's Island Eradication Advisory Group (IEAG; Broome, et al., 2011) act as conscience, critics and confidants for biosecurity affairs.

Codifying of New Zealand's island biosecurity standards and practices first commenced in earnest in the late 1990s, when the rising number of pest-free islands under watch (Russell & Broome, 2016) required a more systematic approach to the work. In addition to validating biosecurity as a specialist function in its own right, the Standard Operating Procedures and Manual of Best Practice (DOC, 2008) of this time strove for national consistency and a persuasive culture of vigilance. Beforehand, biosecurity had been left in the hands of collegially isolated, largely untutored conservation practitioners.

ISSUES AT HOME

In 2012, DOC reacted to a disquieting rise in the number and costs of pest invasions on protected islands by launching a penetrating review of DOC's biosecurity fitness (DOC, 2012). The report into practices, attitudes and capacity testified to a contagious culture of complacency over invasion risks in many parts of DOC's jurisdiction (Broome, 2013). Biosecurity arrangements lacked coherence and firm, visible leadership.

Recommendations for remedy drew on examples of good practice still in place and were formalised into a determined programme of improvement (Broome & Kennedy, 2014). This acquired national priority through high-level sign-off in DOC.

The programme is still in train today. It aims to normalise a vital culture of vigilance in every part of DOC's structure. It seeks, for island biosecurity, the unqualified functional priority given to fire-fighting and Health & Safety, along with comparable prerogatives and resourcing.

In addition to confronting unfamiliar new pest threats such as the arrival of myrtle rust on public conservation lands (<www.doc.govt.nz/our-work/biosecurity/myrtle-rust/>), the upgrade programme is adjusting to new complexities in an evolving social environment. First, contemporary trends in the socialising of island management compound invasion risks by partitioning control, subordinating the biological significance of sanctuaries to other values or by condoning independent rights of access. Second, co-management agreements between DOC and owners, other regulatory agencies or communities are increasingly common. Third, many pest-free islands are passing into iwi (Maori) ownership through Treaty of Waitangi redress for colonial seizures of land. And it is business as usual for DOC itself to promote the rare biota on near-shore islands as a reason for the public to visit or camp as they wish (see for example, <www.doc.govt.nz/parks-and-recreation/places-to-go/auckland/hauraki-gulf-marine-park/visiting-islands-and-marine-reserves/>).

LINES OF IMPROVEMENT

The action plan addresses cultural, capacity and technical issues (Broome & Kennedy, 2014). Its original lines of improvement have been directed internally to ensuring that pest animals, weeds and —in special cases— pathogens did not reach islands through DOC's very frequent traffic to them.

National advisors

The two national advisors appointed to guide all aspects of the upgrade programme are extraordinary roles in the DOC structure. These sole-purpose appointments signal serious intent to make progress. The advisors are authorised to think beyond the action plan to explore emerging needs and new lines of improvement. Their operating mandate extends across all functional divisions in DOC.

Practitioner networks

The advisors have created three regional networks through which biosecurity practitioners can interact more readily with their own kind. The networks aim for peer-mediated migration of knowledge and standards across internal boundaries. Staff exchanges strengthen trust and linkages between all three networks. External associates in local government and NGOs frequently attend annual workshops.

A declared imperative is to build a strong biosecurity collegiate nationwide. Invoking the powerful unifying benefits of collegial interaction promotes horizontal accountability to peers (rather than vertically to managers) and is intended to lift productivity under conditions of capacity shortage at the workface.

Pest-detection dogs

DOC is augmenting its pest-detection dog programme through a formative partnership with New Zealand's Kiwibank (<www.doc.govt.nz/about-us/our-partners/our-national-partners/kiwibank/>). Gaps in capacity revealed through internal review (Vincent, 2015) are to be filled

Truisms for the eradication–biosecurity partnership		
Biosecurity is eradication's cornerstone	Eradication investments will come to nothing without confident biosecurity already in place	
Ask two feasibility questions before proceeding	Can the pest be eradicated? Can the island be defended from reinvasions?	
Poor preparation has a long legacy	If biosecurity is not prepared well at the outset, it will likely be sustained poorly in the aftermath	
Anticipate long lead times	Biosecurity's social and technical complexities require longer preparatory timeframes	
Biosecurity requires people of the Right Stuff	Eradication experts may not have the skills to manage biosecurity on its very different horizons and time-scales	
Quarantine is the best investment. Period.	Quarantine puts biosecurity on the front foot, where it is strategically and tactically most potent	

with new handlers and dogs trained to detect more types of organisms. Dog work will now take priority for handlers with mixed conservation duties. New full-time handlers will rove nationally to points of need. The programme has recently acquired its own national advisor and manager, appointments intended to empower the work through a more autonomous occupational structure.

Best-practice prescriptions

One national advisor acts as the go-to keeper of best-practice knowledge. S/he will codify all aspects of New Zealand's island biosecurity work through updated operational prescriptions and standards. We intend to share these prescriptions with the global community. Codes of practice will be living guides amended as new knowledge is acquired from experts, research or field experience. Learning from informal experimenting in the field is a further source of new knowledge, one traditionally overlooked.

Pragmatic biosecurity plans

Outdated island biosecurity plans in all DOC regions are to be revised. We are piloting a simplifying short-plan format (Kennedy, 2016a; 2017) designed to hasten updates and approvals. The new format restricts plans to defining briefly what has to be done (activities, rules, standards and roles), without long explanatory narratives. Thus, plans will act strictly as operational blueprints, not as textbooks or technical manuals. 'How-to' prescriptions prepared independently of plans will instruct rangers in the technical details of biosecurity tasks. The revised plan for Maud Island (Caldwell & Higgott, 2017) exemplifies the new approach.

CIMS-based incursion responses

We have recently adopted the Co-ordinated Incident Management System (CIMS) as the standard mode of response to pest invasions (Corson, 2018a, b). CIMS clarifies response roles and organises support for ground operations by co-ordinating inputs from relevant experts. We assemble technical advisory groups (TAGs) as needed to guide CIMS decision-makers on appropriate measures. Biosecurity novices are apprenticed to CIMS teams for training on the job.

Upgrade of quarantine facilities

We are ranking DOC's 38 quarantine facilities for complete renewal or internal refits in coming years. These secure stores at mainland points of departure and island landings are the primary pivot-points for movements of DOC people and freight. Facilities on the mainland also function as biosecurity's public face. Design principles (Kennedy, 2016b) are awaiting translation into new architectural and construction standards as part of the best-practice review.

Peer-review of biosecurity practices

Systematic audits of local biosecurity practices and culture have resumed (see for example Kennedy & Chappell, 2013; Kennedy & Trainor, 2016; Broome & Corson, 2017). Experienced practitioners lead the reviews, usually assisted by a novice. Audits are our most decisive means of detecting lapses in standards and propagating successful practices. As peer-reviews, they strengthen mutual trust in DOC's biosecurity ranks. Reviewers, themselves, are instructed by the process of critique and counselling. Currently, audit recommendations are not yet binding.

As expected, these and other lines of operational upgrade have had to compete for resources and priority within a complex organisation attending to demands on multiple conservation fronts. Obstacles to progress associated with biosecurity's social dimensions were under-estimated.

UNEXPECTED OBSTACLES TO GOOD BIOSECURITY

Managing human behaviour is unavoidable in the business of keeping islands pest-free. In contrast, eradication is more a technical discipline and therefore the easier part of the pest-free equation.

DOC's internal biosecurity arrangements cannot be intensified successfully, nor can they be sustained in perpetuity, if biosecurity behaviour is not normalised in organisational thought and practice. The same applies to public traffic to islands. By necessity then, we have extended our upgrade programme to invoke beneficial behaviours in public audiences whose multiple forms of contact with pest-free islands are less within DOC's control.

Of the two broad classes of audience (professional and public), our colleagues have proven to be the more resistant to the right behaviours. This is perplexing, since they are closely invested in conservation and should be acutely aware of invasive pest threats to insular biota. Yet compliance with expected behaviours still appears to be more obligatory than voluntary.

Expert mind-set

We observe an ‘expert’ mind-set at play in our colleagues. It reasons that as conservation specialists they are best placed to judge risk and thus how much they need to comply with rules. Treating compliance as discretionary translates biosecurity from an essential good into a nuisance (rationalised perhaps as an intrusion on professional judgement). Ough Dealy (2016) reports a similar 'experts-know-best' phenomenon in dog owners landing their pets illegally on pest-free islands in New Zealand’s far north. Biosecurity’s logic is not sufficient on its own to alter this mind-set.

Colleagues resist further through arguments that there is little point in DOC staff submitting to quarantine checks when the public can visit the same islands without doing so. This surprisingly prevalent thinking underwrites pressure to reduce quarantine standards (authors, pers. obs.). Its disabling logic supposes that biosecurity can extinguish all risk. We argue instead that it can only minimise the probability of pest incursions. So, if some of an island’s visitors are guaranteed pest-free, the risk is reduced accordingly.

Image problems

Biosecurity is inherently prone to neglect. It has a poor image. In contrast with the heroic character of eradication operations, biosecurity work is less glamorous, inherently open-ended and not so dramatically rewarded. Reputations are simply not made in the business. Arguably then, biosecurity does not appeal to practitioners motivated vocationally to make a demonstrable difference for conservation (Kennedy, 2003).

Perversely, biosecurity is a casualty of its own success. When everything works as it should, nothing happens. This is particularly penalising in today’s over-determined goal-directed working environments. For resource-stressed colleagues in DOC, intercepting very few invasive pests, year after year, argues for shifting effort from biosecurity to more obviously productive work.

We find as a result that our field and management staff are inclined to treat biosecurity as an insurance policy on which it is safe to avoid paying the premiums. Too often, they subordinate biosecurity to lower but more immediate priorities. Even financially punishing biosecurity failures (for example, the > $NZ200,000 mouse invasion of Maud Island; Broome, et al., this issue) have had only a limited chastening effect on this habit.

OVERCOMING ATTITUDINAL OBSTACLES IN DOC

We are in a stronger position to legislate compliance with this captive audience. Traffic to islands is governed by behavioural rules applying to all DOC staff, our associates and our freight. Quarantine inspections are obligatory (see for example, Hiscock, 2016) and must be allowed for in operational timetables. At points of departure, authorised biosecurity gate-keepers are mandated (regardless of their occupational rank) to prohibit travel until quarantine standards are met.

Ultimately, normalising biosecurity will be achieved in this audience by playing on the powerful human instinct to conform to peer-pressure. This influence on attitudes and behaviour is a defining quality of collegiality, itself articulated through peer-mediated understanding of common purpose, values and identity. DOC’s biosecurity networks are a resolute first step towards this goal. Creative celebrating of successes and champions will help. Loss of occupational autonomy under DOC’s line-management arrangements is a barrier to progress (Kennedy, 2003).

OBSTACLES TO BEHAVIOUR-CHANGE IN PUBLIC AUDIENCES

Island biosecurity in New Zealand is faltering most conspicuously in its managing of public access to pest-free islands. DOC is still seriously under-invested here, most obviously because the tools of social psychology are not yet used adequately by DOC itself or trusted by biologically-minded practitioners.

Problematic messaging

In piecemeal approaches or hesitant use of creative messaging, outreach strategies reflect their uncertainty on how to bring behaviour-change about. Neglecting outreach as a quarantine measure because of its social uncertainties denies biosecurity a powerful range of interventions. A distorting emphasis on surveillance and response is likely to result, particularly as these activities are more comfortable technical ground for practitioners unskilled in modifying public behaviour.

For instance, three years after cats and rodents were eradicated from Great Mercury Island (1,872 ha) in New Zealand’s southern Hauraki Gulf, on-island surveillance and response regimes are in place (Collins & Corson, 2016) but quarantine lacks any coherent messaging strategy to manage visitor risk. As a result, biosecurity to manage the differing forms of public contact with the island is dangerously off-balance and inherently reactive.

In the absence of insights from social research, the customary response is to fall back on orthodox messaging through signs and pamphlets. Typically, this relies on a mix of appeals to protect natural values and prohibitions on unwanted behaviour, all conveyed in alienating official language. We are not confident that this messaging or its media are effective. Much of it amounts to shouting at audiences.

Branding is similarly problematic. In 2009, Auckland Council and DOC launched the *Treasure Islands* brand (<www.treasureislands.co.nz>) to engender biosecurity behaviour in the hundreds of thousands of public and commercial travellers to 44 pest-free islands in the Hauraki Gulf Marine Park. Uncertain of its effectiveness, brand design was under constant review (Jack, 2011). Variations were visible everywhere, from bill-boards to bait stations, pamphlets to piers. Later surveys of biosecurity awareness in island residents and visitors showed limited knowledge of the brand as it had been communicated (Auckland Council, 2010a, 2010b; Lysnar, 2016). Fraser, et al. (2016) concluded that face-to-face conversations with ferry passengers were a more effective means of outreach.

We consider that the finer-scale insights of qualitative social research studies are more likely than quantitative to determine which messages and media will prompt target audiences to become willing biosecurity actors—to quarantine their personal gear before departure. Social research shows already that those messages will likely resonate with audiences’ own reasons for visiting islands, not with our notions of nobility in conservation values or biosecurity need (see McKenzie-Mohr, 2013).

Biosecurity’s unwitting social offences

Invoking the quarantine habit in public audiences faces cryptic psychological barriers. We regard credibility as a crucial issue. As with our sceptical colleagues, visitors arriving on controlled pest-free islands are disinclined to believe that they have a mouse or a rat in their bag when asked to check (Tyrrell, 2012). This is a pivotal moment at which biosecurity’s legitimacy is questioned along with the gate-keeper’s sanity.

Winning the contest for credibility is made more difficult in that emptying bags and pockets in the company of strangers can be socially awkward, particularly in the congested conditions of an enclosed quarantine room. At worst it may be regarded as an intrusion on personal privacy.

Remedies for this and many other unwitting social offences seem intractable to practitioners who are suspicious of the arcane social sciences. This is yet another barrier to overcome. In fact, remedies will be more likely in the hands of social psychologists; they can no longer be left to chance or to force of personality in biosecurity gate-keepers.

ADDRESSING THE CHALLENGES

DOC's investments to strengthen island biosecurity have had to extend beyond the internal focus of its original action plan. Thinking and initiatives are advancing along the following lines.

Biosecurity's functional identity and appeal

We see potential to lift biosecurity's functional appeal and effectiveness in-house by giving the work its own occupational structure, dedicated line-manager and credentials. This thinking recognises that biosecurity is a specialist field in its own right, entitled to greater operational autonomy, its own leadership prerogatives and requiring operators uniquely suited to its distinctive modes of work. Not all field ecologists have the necessary manner or motivations. Discrete occupational identity breaks with today's unhelpful assumption that biosecurity is merely an adjunct function of eradication and other specialist work.

Audits of readiness

Biosecurity readiness for eradication operations and ongoing quarantine would benefit from more determined auditing. An expert group resembling the IEAG appeals as a means for authoritative scrutiny. The two should operate in parallel, even if they share experts from time to time.

Testing our own attitudinal barriers to good practice

We see immediate value in social research to investigate the attitudinal barriers to biosecurity uptake in our own colleagues. Likewise, we are learning to test the assumptions we make routinely about who our public audiences are, what they understand and what motivates them. Inexpert assumptions here make unreliable stepping-off points for changing public behaviour.

Seeing the improbable

We are adopting the principle that our colleagues and public alike must see for themselves before they will accept the improbable. This means replacing talking and preaching with evidential photos, videos, eye-witnesses and stories. In the quarantine inspection room on Matiu/Somes Island in Wellington Harbour, for instance, incoming visitors can see and handle a sobering collection of pest organisms intercepted in bags and pockets.

Measuring the costs of failure

The time and dollar costs of incursion responses are now recorded more carefully using new reporting templates to prompt for essential information (Kennedy, 2015a). These data are collated to argue for better resourcing and priority. Each incursion is treated as an experiment from which lessons are now extracted more systematically (Kennedy, 2015b; Kennedy, 2016c). All reports are added to DOC's comprehensive database of island incursions (Vincent, 2017).

Making biosecurity sexy

DOC's pest-detection dogs are now employed more purposefully as our ambassadors and champions. Their unique ability to convey biosecurity messages by their very presence surpasses the best of human efforts. Contact time with public audiences of all types is now built into work schedules.

Social research

We have commenced two qualitative social research projects designed to find ways of stimulating beneficial behaviour in island visitors. Both use community-based social marketing methods (CBSM; McKenzie-Mohr, 2013) and are intended to generate solutions applicable throughout New Zealand.

The first project is searching for effective ways to convince recreational boat users to leave their pet dogs at home. Dogs landed with families picnicking on pest-free open-access islands in New Zealand's far north are killing endangered kiwi chicks crèched there for conservation purposes (Ough Dealy, 2016; Ough Dealy & Greig, 2017). Findings from this modest inaugural study have confounded most of our assumptions about the target audience and its motivations to take dogs ashore illegally (further information is available from the authors).

The second study aims to persuade recreational boat users to check their vessels and gear for pests before they leave home. This study will test our hypothesis that willingness to check at home will be greater than at the launch ramp where time and other pressures intrude. Consistent with CBSM methods, this study has attempted to isolate specific behaviours for change while exploring the character of the target audience (Harbrow, 2017).

Conversing with audiences

Though still only a concept, we aim to converse with rather than talk at public audiences. Experiments with more relaxed, colloquial language on conventional signage have been contemplated but not yet launched. Likewise, we are considering the head-turning potential in non-conventional signage (say, 3-D models of rodents) as a means of distinguishing our biosecurity messages from the walls of rectangular 2-D bill-boards island visitors expect to see at departure points.

We have also adopted the principle of conversing continuously with audiences well in advance of their arrival at departure points. Visitors of all kinds travelling to controlled-access islands receive information ahead of time on how to prepare for their quarantine inspection. YouTube videos (<https://youtu.be/yXNOpfW7PPQ>) are used too. Bounce-back messages on booking sites are under consideration. Open-access islands present greater difficulties. Here, a progression of messages along approach routes, conversing by various media, would avoid the unsatisfactory situation of today where visitors are confronted with conditions on travel at the last moment, when they are least able or prepared to comply.

Reaching back into supply chains

Following the lead of Morgan, et al. (2014), we regard it as imperative to check freight for pests early in the supply chain. We award pest-free warrants (<www.doc.govt.nz/parks-and-recreation/places-to-go/auckland/hauraki-gulf-marine-park/know-before-you-go/treasure-islands/pest-free-warrant/>) to businesses guaranteeing to supply pest-free goods or services. These businesses are used and publicised as preferred suppliers. Not only is quarantine at source more efficient but it safeguards biosecurity gate-keepers from pressures to pass suspect goods or freight for shipping under operational urgency at departure points.

CONCLUSION

Advances of these kinds are intended to make good on biosecurity's duties to its eradication partner. They spring from lessons learnt during determined upgrading of DOC's island biosecurity performance. We recognise now that biosecurity must be the cornerstone of island restoration work, not a secondary function of pest eradication or other conservation investments. Its effectiveness is not assured, however, unless it is respected as a highly specialised discipline, resourced accordingly and trusted to credentialed practitioners attuned to the particular demands of quarantine, surveillance and response. We have become acutely conscious of the need to address the discipline's complex social dimensions. These have been neglected at home historically because biosecurity work was usually left to field ecologists unfamiliar with social research tools. Biosecurity cannot avoid managing human behaviour if it is to acquire its necessary reach and balance. More systematic programmes of social investment are needed to address the obstacles to good biosecurity today. They will be outstanding preparation for the future in which the formidable social challenges of eradicating pests from populated islands, or indeed from entire countries (e.g. Predator Free New Zealand 2050) will demand more sophisticated tools and thinking than we possess now.

REFERENCES

Auckland Council. (2010a). *Report on a Survey of Passengers on Sealink's Great Barrier Island Service, Summer 2009–2010*. Unpublished internal report. Auckland, NZ.

Auckland Council. (2010b). *Report on a Survey of Boat-users in the Hauraki Gulf, 2009–2010*. Unpublished internal report. Auckland, NZ.

Bassett, I.E., Cook, J., Buchanan, F. and Russell, J.C. (2016). 'Treasure Islands: biosecurity in the Hauraki Gulf Marine Park'. *New Zealand Journal of Ecology* 40 (2): 250–266.

Boser, C.L., Corey, C., Faulkner, K.R., Randall, J.M., Knapp, J.J. and Morrison, S.A. (2014). 'Strategies for biosecurity on a nearshore island in California'. *Monographs of the Western North American Naturalist* 7 (1): 412–420.

Broome, K.G., Cromarty, P., Cox, A., Griffiths, R., McClelland, P. and Golding, C. (2011). 'The Island Eradication Advisory Group (IEAG)—A model for effective technical support for eradication project planning and management'. In: C. R. Veitch, M. N. Clout and D. R. Towns (eds.) *Island invasives: eradication and management*, p. 524. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Broome, K.G. (2013). *Island biosecurity: Improvements to DOC's Practice and Culture*. Unpublished project report, DOCDM-1118748. Hamilton, NZ: Department of Conservation.

Broome, K.G. and Kennedy, E.S. (2014). *Action Plan for Island Biosecurity*. Unpublished internal plan, DOCDM-1372876. Christchurch, NZ: Department of Conservation.

Broome, K.G. and Corson, P. (2017). *Assessment of Biosecurity for Whenua Hou Nature Reserve*. Unpublished internal audit report, DOCCM-2968470. Hamilton, NZ: Department of Conservation.

Broome, K.G., Brown, D., Brown, K., Murphy, E., Birmingham, C., Golding, C., Corson, P., Cox, A. and Griffiths, R. (This issue). 'House Mice on Islands: Management and Lessons from New Zealand'.

Caldwell, S. and Higgott, F. (2017). *Te Pākeka Maud Island Biosecurity Plan: Quarantine, Surveillance and Incursion*. Unpublished operational plan, DOCCM-2834023. Picton, NZ: Department of Conservation.

Clout, M.N. and Russell, J.C. (2006). 'The Eradication of Mammals from New Zealand Islands'. In: F. Koike, M.N. Clout, M. Kawamichi, M. de Poorter and K. Iwatsuki (eds.) *Assessment and Control of Biological Invasion Risks,* pp.127–141. Gland, Switzerland: IUCN.

Collins, L. and Corson, P. (2016). *Ahuahu Great Mercury Island Biosecurity Plan*. Unpublished operational plan, DOCDM-1374415. Whitianga, NZ: Department of Conservation.

Corson, P. (2018a). *Island Biosecurity Incursion Responses: Flow Diagrams by Incursion Phase*. Unpublished internal guideline, DOCCM-5387656. Rotorua, NZ: Department of Conservation.

Corson, P. (2018b). *Roles and Responsibilities for Island Biosecurity Pest Incursions*. Unpublished internal guideline, DOCCM-5388157. Rotorua, NZ: Department of Conservation.

DOC. (2008). *Island Biosecurity Best Practice Manual (v.2.2)*. Internal publication, DOCDM-20171. Wellington, NZ.

DOC. (2012). *Island Biosecurity: Improvements to DOC's Culture and Practice*. Unpublished project brief, DOCDM-1003645. Hamilton. NZ: Department of Conservation.

DOC. (2015). *Standard Operating Procedure for the Conservation Dog / Handler Team*. Unpublished internal SOP, DOCCM-749423. Wellington, NZ.

DIISE. (2015). The Database of Island Invasive Species Eradications, developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand. <http://diise.islandconservation.org>.

Fraser, D., Dabb, H. and Graham, C. (2016). Biosecurity Awareness of Ferry Passengers Travelling to Islands in the Hauraki Gulf, New Zealand. Auckland, NZ: Environmental & Animal Sciences, Unitec. <http://hdl.handle.net/10652/3866>.

Harbrow, M. (2017). *Auckland Boat-show Survey, 23 April–22 May 2017*. Unpublished summary of data (PDF). Wellington, NZ: Department of Conservation.

Hiscock, J. (2016). *Subantarctic Expeditions: Operational Procedures for Management and Research Expeditions in the New Zealand Subantarctic*. Unpublished rules and guidelines, DOCDM-1351346. Invercargill, NZ: Department of Conservation.

Howald, G., Donlan, C.J., Galván, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, C.R., Genovesi, P., Pascal, P., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Jack, S. (2011). *Treasure Islands – an Overview of the Biosecurity Campaign for the Hauraki Gulf*. Unpublished internal briefing paper, DOCDM-725070. Auckland, NZ: Department of Conservation.

Kennedy, E.S. (2003). 'Managerialism as a Professionalising Catalyst for the Front-line Practitioner Community of New Zealand's Department of Conservation'. M.Appl.Sc Thesis. Lincoln, NZ: Lincoln University. <http://researcharchive.lincoln.ac.nz/handle/10182/1031>.

Kennedy, E.S. and Chappell, R. (2013). *Preliminary Report: Audit of DOC's Quarantine and Surveillance Arrangements for the Islands of the Marlborough Sounds*. Unpublished audit report, DOCCM-1318761. Christchurch, NZ: Department of Conservation.

Kennedy, E.S. (2015a). *Summary of Pest Incursion Costs on Islands*. Unpublished report template, DOCDM-1560956. Christchurch, NZ: Department of Conservation.

Kennedy, E.S. (2015b). *Debrief Report for Island Incursion*. Unpublished report template, DOCDM-2619461. Christchurch, NZ: Department of Conservation.

Kennedy, E.S. (2016a). *How to Write a Genuinely Useful Biosecurity Plan in a Jiffy*. Unpublished presentation (PPT), DOCCM-2844257. Christchurch, NZ: Department of Conservation.

Kennedy, E.S. (2016b). *Quarantine Store Design*. Unpublished summary of design and structural elements, DOCCM-3186359. Christchurch, NZ: Department of Conservation.

Kennedy, E.S. (2016c). *Norway Rat Incursion, Ulva Island, September–December 2015*. Unpublished internal report using the debrief template, DOCDM-2808849. Christchurch, NZ: Department of Conservation.

Kennedy, E. S. and Trainor, S. (2016). *Audit of DOC's Biosecurity Store at Halfmoon Bay, Rakiura Stewart Island*. Unpublished audit report, DOCCM-2874997. Christchurch, NZ: Department of Conservation.

Kennedy, E.S. (2017). *Proposed Three-part Structure for Biosecurity Plans*. Unpublished internal pamphlet, DOCCM-3186052. Christchurch, NZ: Department of Conservation.

Lysnar, P. (2016). *Summary of Findings from Hauraki Gulf Treasure Islands Campaign Survey*. Unpublished contract report. Auckland, NZ: Environmental Services, Auckland Council.

McKenzie-Mohr, D. (2013). *Fostering Sustainable Behaviour—An Introduction to Community-Based Social Marketing*. Canada: New Society Publishers.

Morgan, S., Marshall, A., Cook, J., Waipara, N., Ward, D., McNatty, A., Adams, N., and Fraser, D. (2014). 'An Assessment of Risk Pathways from Commercial Businesses to Pest-free Islands of the Hauraki Gulf'. Presentation to the New Zealand Biosecurity Institute (NETS) conference, 29 July–1 August 2014, New Plymouth, New Zealand. <http://hdl.handle.net/10652/3097>.

Ough Dealy, H. (2016). *Dogs on Pest-free Island Beaches in the Bay of Islands, Northland*. Unpublished summary of interim research findings, DOCCM-2845381. Bay of Islands, NZ: Department of Conservation.

Ough Dealy, H. and Greig, E. (2017). *Visitor and Dog Monitoring on Urupukapuka Island, 2014–2017*. Unpublished internal research report, DOCCM-3010021. Bay of Islands, NZ: Department of Conservation.

Russell, J.C. and Broome, K.G. (2016). 'Fifty years of rodent eradications in New Zealand: another decade of advances.' *New Zealand Journal of Ecology* 40(2): 197–204.

Russell, J.C., Clout, M.N. and Towns, D.R. (2007). 'Early Detection and Eradication of Invading Pests'. In: G.W. Witmer, W.C. Pitt and K.A. Fagerstone (Eds.) *Managing Vertebrate Invasive Species 43: Proceedings of an International Symposium,* pp. 268–272. Fort Collins, Colorado, USA: USDA/APHIS Wildlife Services, National Wildlife Research Center. <http://digitalcommons.unl.edu/nwrcinvasive/43>.

Tyrrell, L. (2012). *2012 Summer Survey Anecdotal Report (March 2012)*. Unpublished survey report to Auckland Council. Unitec, Auckland.

Veale, A.J. (2012). 'Observations of stoats (*Mustela erminea*) swimming'. *New Zealand Journal of Zoology* 40: 166–169.

Vincent, K. (2015). *Pest Detection Dogs Programme Review*. Unpublished internal report, DOCCM-1534015. Wellington, NZ: Department of Conservation.

Vincent, K. (2017). *Database of Island Invasion Incidents*. Unpublished internal database, DOCDM-53231. Wellington, NZ: Department of Conservation.

Mexico's island biosecurity programme: collaborative formulation and implementation

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Abstract Mexico's National Strategy on Invasive Species (2010) and the National Strategy for the Conservation and Sustainable Development of Islands (2012) embrace its steady and positive 20-year trajectory on island restoration. Mexico has come halfway to having all islands free of invasive mammals. To sustain these results in the long-term, biosecurity became a priority. To implement these national strategies, the National Commission for Knowledge and Use of Biodiversity (CONABIO) and the National Commission for Natural Protected Areas (CONANP) integrated a participatory programme to develop the country's capacities for managing invasive alien species (IAS). With funding from the Global Environment Facility (GEF) through the United Nations Development Programme (UNDP), and private donors, the IAS programme is now under operation. Grupo de Ecología y Conservación de Islas, A.C. (GECI), a professional Mexican NGO, is implementing the programme on islands, along four lines of action: (1) island biosecurity; (2) environmental learning and capacity building; (3) control and eradication of IAS; and (4) monitoring to assess ecological recovery. While promoting the importance of biosecurity amongst all social actors, the focus is on long-term formal implementation, culture and everyday life. The methods therefore vary: workshops with authorities, integration of specific biosecurity protocols, and art and conservation activities with the local communities, particularly with children and young adults. The experience to date shows that enforcement by authorities and integration of the subject by local fishermen communities and island users are two key factors in sustaining the valuable and tangible results achieved to date over the long term.

Keywords: biosecurity, conservation, early detection, eradication, prevention, rapid response

INTRODUCTION

As one of the world's megadiverse countries, Mexico acknowledges the importance of safeguarding its biodiversity and over 10,000 endemic species (Llorente-Bousquets & Ocegueda-Cruz, 2008). Invasive alien species (IAS) pose the most important threat to biodiversity worldwide (Reaser, et al., 2007; Towns, 2011), and have caused 67% of the extinctions of Mexican vertebrates (Aguirre-Muñoz, et al., 2011a). Consequently, a National Advisory Committee for the Strategy on Invasive Species (CANEI, for its Spanish acronym) was created in 2008. It is comprised of governmental and academic institutions, as well as non-profit civil society organisations. Coordinated by the National Commission for the Knowledge and Use of Biodiversity (CONABIO), the CANEI developed the "National strategy on invasive species: prevention, control and eradication" in 2010. Its vision is to address the problems of IAS, by creating efficient prevention, early detection and rapid response systems, as well as a legal framework to mitigate, control and eradicate these species (CANEI, 2010).

The nearly 4,000 Mexican islands, as do most of the islands around the world, host a disproportionate amount of the country's biodiversity (Whittaker & Fernández-Palacios, 2007). They are hotspots of endemism richness, with 14 times more endemic species than the mainland (Aguirre-Muñoz, et al., 2016a). In recognition of the need to protect this biodiversity as well as the livelihoods of island communities, the Mexican government has included all islands in the National System of Natural Protected Areas (Aguirre-Muñoz, et al., 2017a) with the recent decree of the Islas del Pacífico de la Peninsula de Baja California Biosphere Reserve (DOF, 2016). Therefore, the formulation of the National Strategy for the Conservation and Sustainable Development of the Mexican Island Territory (2012) was an important step forward. This national strategy sets priorities to work on three tactical lines – sovereignty, conservation and sustainable development – through four transverse lines of action – knowledge, public policies, inter-institutional coordination and financing (CANTIM, 2012).

ISLAND CONSERVATION IN MEXICO

The history of island conservation in Mexico delivers a restoration success story. Through to 2017, 60 populations of 11 invasive mammal species have been eradicated from 39 islands, which represents over 59,000 ha restored (Aguirre-Muñoz, et al., 2018). Thanks to these efforts, at least 147 endemic taxa of mammals, reptiles, birds and plants are protected. Furthermore, 227 highly vulnerable seabird colonies are recovering from the impacts of IAS (Aguirre-Muñoz, et al., 2016b). A growing network of collaborating federal government agencies, e.g. the National Commission for Protected Areas (CONANP), CONABIO, the National Institute of Ecology and Climate Change (INECC), and the Department of the Environment and Natural Resources (SEMARNAT), academic institutions, local communities, fishing cooperatives, civil society organisations and donors (national and international) has been fundamental to achieving success. Working in close collaboration with the multiple partners, Grupo de Ecología y Conservación de Islas, A.C. (GECI) has implemented all but two of the island eradications in Mexico and is currently executing other eradication projects on several islands. GECI is a Mexican civil society organisation, which works with an interdisciplinary and comprehensive approach toward the restoration, conservation and sustainable development of islands (Aguirre-Muñoz, et al., 2011b).

GECI's goal, as outlined in the IUCN's Honolulu Challenge, is to remove invasive mammals from all islands of Mexico by 2030 (IUCN, 2017). To achieve it, we need to eradicate a further 70 populations of invasive mammals from 34 islands. To do so, we aim to eradicate invasive mammals following restoration priorities, including where

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endemic species are vulnerable, eradications are feasible and risk of reinvasion is lower (Latofski-Robles, et al., 2014). Therefore, the implementation of a National Programme for Island Biosecurity – the policies, measures and actions to protect island biodiversity from IAS by preventing their arrival and establishment (Roberts, 2003; Russell, et al., 2008) – is vital to ensure that successes achieved remain in the long term, and that the investment in conservation measures, such as eradications, has the highest return rates (Broome, 2009). Implementing biosecurity will also further Mexico's achieved international commitments, in line with Aichi Biodiversity Target #9 which states: "By 2020, IAS and pathways are identified and prioritised, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment" (CBD, 2010). Additionally, new restoration projects will benefit from building biosecurity capacities beforehand. Thus, biosecurity becomes a transverse line of action amongst all of GECIs restoration projects (Aguirre-Muñoz, et al., 2016b).

FORMULATING THE NATIONAL ISLAND BIOSECURITY PROGRAMME

Islands significantly contribute to the country's megadiversity. They harbour 8.3% of all vascular plant and terrestrial vertebrates (CANTIM, 2012). They also support the livelihood of more than 200,000 people, most of which rely on the valuable marine resources that thrive in adjacent waters. However, some islands have faced the negative impacts of IAS, particularly mammalian predators, for centuries. The introduction of such problematic species to islands in Mexico has been mainly due to anthropogenic reasons, either intentionally or accidentally. Before the 20th century, introduction of IAS was mainly related to the harvesting of marine mammals and guano mining. Nowadays, the sources of introductions have diversified and include commercial and sport fishing, as well as tourism related activities (Aguirre-Muñoz, et al., 2011b). At first, restoration projects were all about solving the problem already at hand, eliminating the IAS; however, as we free islands of their IAS, we must change our way of thinking and become proactive in preventing reintroductions or new introductions. In order to halt the introduction of IAS, intentional or accidental, we need a society that is aware of the root causes and problems associated with the loss of biodiversity and the ecosystem services it provides. We need the social construction of a new paradigm, of everyone feeling a sense of privilege every time we visit an island and acknowledging that the conservation of such a special place is in our own hands.

Therefore, GECI's restoration projects are accompanied by an environmental learning and outreach campaign that is designed for that specific island and its local community's characteristics. We seek to boost the local community identity, by publicising the island's biodiversity, as well as its endemic or more charismatic species. We produce and distribute different outreach materials (e.g. posters, photographic catalogues, wristbands, colouring books, puzzles, etc.) that showcase the island's uniqueness and what you can do to protect it. We also give varied talks to different sectors, such as schools, universities, fishing cooperatives and tourist operators, about the restoration project and the outcomes expected. Moreover, we learn about the way local communities understand, interact with and feel about their environment through their artistic expressions. We provide the opportunity for youngsters to express their connection to nature through music, painting, drawing and story-telling workshops, and have documented beautiful results.

GECI's efforts to make island biosecurity a subject matter and common topic amongst island users and managers became systematic with the nationwide project to implement the Strategy on Invasive Species in Mexico. With funding from the Global Environment Facility (GEF) in coordination with the United Nations Development Programme (UNDP), the CONABIO and the CONANP lead the inter-sectorial project to implement this Strategy. Implementing biosecurity protocols and building capacities on managing IAS are two priority actions established in the Invasive Species Strategy (CANEI, 2010). The project is implemented in priority areas of conservation and focuses on preventing the arrival and establishment of IAS through prevention measures, early detection systems and rapid response (Born-Schmidt, et al., 2017).

The project began the planning stage in 2012, and GECI, who is coordinating the island programme, started by identifying priority protected areas for implementation and setting action guidelines. The lines of action, with a 2015–2018 implementation horizon, are: 1) Biosecurity: development, implementation and evaluation of biosecurity protocols, creation of biosecurity committees; 2) Environmental learning and outreach: producing outreach materials, developing awareness campaigns about IAS, building capacities for local groups on early detection and rapid response; 3) Restoration: management of the IAS, as well as native species present; 4) Monitoring: documenting ecosystem responses to eradication of IAS (Aguirre-Muñoz, et al., 2013). Six priority protected areas are our pilot project areas where the biosecurity project is currently being implemented (Table 1, Fig. 1). The project is being replicated in the Gulf of California, in a group of islands known as the Midriff Islands.

DESIGNING AND IMPLEMENTING ISLAND BIOSECURITY PROTOCOLS

In order for biosecurity to fulfil its purpose, we need to analyse and take into account all the particular activities that different sectors carry out on the island. Consequently, we decided on a "bottom-up" strategy to create site-specific biosecurity protocols in an adaptive and participatory manner (Aguirre-Muñoz, et al., 2013). With every sector involved in the protocol design from the beginning, they provide the information needed to make an informed risk analysis and detect critical control points (González-Martínez, et al., 2017). Furthermore, by being involved, the communities are more likely to approve and adopt prevention measures that need to be carried out in everyday life and with a long-term vision.

Biosecurity protocols are documents where all the components of biosecurity are detailed; so that each stakeholder understands what will be implemented, and how he/she is involved. The main components of biosecurity are prevention, early detection and incursion response (Russell, et al., 2008). The key behind prevention

Fig. 1 Map of the islands and their coastal areas of influence for the Biosecurity Programme.

Table 1 Biosecurity pilot project areas.

Island	Location	Previous eradications	IAS present	Local community
Isla Guadalupe Biosphere Reserve	Pacific Ocean (260 km off the coast of the Baja California Peninsula)	rabbit & donkey (2002) horse (2004) goat (2006) dog (2007) cat (in progress)	Plants 47 Reptiles 0 Birds 5 Mammals 2	100 people, comprising a fishermen's camp, a Navy Station and GECI´s station.
Isla Cedros – Pacific Peninsula of Baja California Biosphere Reserve Islands: Cedros & San Benito Oeste	Pacific Ocean (25 km off the coast of Baja California Sur Peninsula)	Cedros: dog (in progress) San Benito Oeste: rabbit & goat (1998) donkey (2005) cactus mouse (2013)	Cedros: Plants unknown Reptiles 0 Birds 4 Mammals 6 San Benito Oeste: Plants 9 Reptiles 0 Birds 4 Mammals 0	10,000 people comprising a fishermen's cooperative, the Navy Station, and the salt exporter.
Archipiélago de Revillagigedo National Park Islands: Socorro & Clarión	Pacific Ocean (480 km off the coast of Baja California Sur)	Socorro: sheep (2010) cat (in progress) Clarión: sheep & pig (2002)	Socorro: Plants 47 Reptiles 1 Birds 5 Mammals 2 Clarión: Plants unknown Reptiles 1 Birds 5 Mammals 1	Socorro: 40 people at the Navy Station Clarion: 15 people at the Navy Station
Isla Espíritu Santo – Gulf of California Islands Protected Area	Gulf of California (25 km off the coast of Baja California Sur)	cat (2017/absence confirmation stage) goat (in progress)	Plants 5 Reptiles 0 Birds 0 Mammals 1	No permanent settlement, however during fishing season around 90 people camp there. Highly visited tourist spot.
Banco Chinchorro Biosphere Reserve Islands: Cayo Centro, Cayo Norte Mayor & Cayo Norte Menor.	Caribbean Sea (30 km off the coast of Quintana Roo)	Cayo Centro: black rat & cat (2015) Cayo Norte Mayor & Menor: black rat (2012)	Plants 6 Reptiles 1 Birds 2 Mammals 0	Cayo Norte Mayor: 12 people Navy Station Cayo Centro: 3 people CONANP station, 100 people fishermen's camps. Tourist visitors.
Arrecife Alacranes National Park Islands: Pérez, Pájaros, Muertos, Desterrada & Chica.	Gulf of Mexico (140 km off the coast of Yucatan)	Pérez: black rat (2011) Muertos & Pájaros: house mouse (2011)	Plants 5 Reptiles 0 Birds 1 Mammals 0	Pérez: 15 people from the Navy Station and CONANP station. During fishing tournaments around 40 camp.

is to set as many obstacles as possible throughout the pathways of introduction, to reduce the probability for IAS to get to the islands. Early detection means a surveillance method through detection devices, such as traps, to determine if there is an incursion. Surveillance is a long-term strategy that requires funding and local capacity building. Finally, an incursion response plan, in case an IAS is detected or suspected, aims not only to confirm the incursion but also to eliminate the IAS (Moore, et al., 2010). Biosecurity protocols contemplate, at least, the following aspects: 1) Identifying the main potential IAS; 2) Identifying possible pathways and vectors of introduction; 3) Establishing prevention measures on the mainland; 4) Establishing early detection systems at disembarking sites; 5) Establishing an incursion response plan; 6) Establishing stakeholders responsibilities (PII, 2013).

Since 2014, we have held workshops for the participative formulation of biosecurity protocols for our pilot areas (and others). We invite local authorities (CONANP, SEMAR, port authorities), fishermen and tourist operators, and we go through all stages of biosecurity and discuss the sites most visited, frequency, and type of transportation. Afterwards, we vote on prevention measures and where to implement them. Additionally, we do a field practice about surveillance and early detection devices commonly used.

To date, we have six unique, specific, updated, island biosecurity protocols, created in a participatory manner. The protocols contain priorities for prevention measures and the most cost-effective and site-specific tools and methods. Protocols are currently under review by the corresponding authorities (Latofski-Robles, et al., 2017). Protocols were formally validated through workshops with the Advisory Council for each island. Furthermore, we strive to create Biosecurity Committees that are a subgroup of said Advisory Councils. These Committees will be in charge of implementation, evaluation and updating of the protocols, as well as fundraising for biosecurity to continue in the long run.

ISLAND BIOSECURITY AT WORK

The most relevant component of biosecurity is prevention. However, all stakeholders need to communicate and coordinate in order for it to be effective. Prevention is closely linked to outreach and environmental education campaigns (Parkes, 2013). An analysis of costs from the Mexican island experience, overwhelmingly demonstrates the importance of investing in biosecurity prevention measures. Recent rodent eradications in Mexico, show that, on average, it costs 20 times more to perform an eradication project than to prevent the arrival of IAS (Aguirre-Muñoz, et al., 2017b).

Early detection is of critical importance to discover any elusive individual that managed to escape the prevention measures. Thus, it also helps to evaluate the prevention strategy. Local capacity building, strong partnerships and straightforward communication between local communities, island managers and other stakeholders (e.g. tourist operators) is critical for a swift and effective incursion response. Furthermore, the *ad hoc* design and wide distribution of outreach materials for each island is vital to raise awareness of the problem of IAS.

As our National Biosecurity Programme unfolds, we have had two effective incursion response events that have successfully stopped the establishment of rodents in Arrecife Alacranes. This is a positive sign that the outreach campaign and workshops are having an effect, and that people are now aware that islands should be IAS-free and their involvement is needed to achieve that (Latofski-Robles, et al., 2016; Matos, et al., 2018). Much has been learnt from incursion events, and the lessons must be adopted nationwide to strengthen prevention measures and community involvement.

INSTITUTIONALISING BIOSECURITY

Building capacity amongst protected area managers and users regarding island biosecurity methods and techniques is crucial to protect the islands from the impacts of IAS. The threat of IAS is considered as important in most of the protected areas management plans; however, preventing their accidental introduction is not commonly featured.

The first step toward building biosecurity capacities for the Mexican islands was the "Island Biosecurity Workshop for managers, park rangers and users of protected areas" in 2014. It was held by GECI with funding from the US Fish & Wildlife Service and the CONABIO. Twenty-six people from all island protected areas in Mexico, gathered in Ensenada, Baja California for three days, during which we discussed biosecurity measures and practiced with early detection devices in Todos Santos Sur Island. Representatives from all agencies regarding islands came together. There were people from CONANP, CONABIO, the Mexican Navy, and the SEMARNAT Office for Wildlife (DGVS). We also analysed the challenges and opportunities to implement biosecurity protocols, prevention measures and early detection systems (Méndez-Sánchez, et al., 2014).

Moreover, GECI has had a solid collaboration history with the Mexican Navy (Secretaría de Marina, Armada de Mexico). They are invaluable partners in the conservation of the Mexican islands, always providing their support on projects (Aguirre-Muñoz, et al., 2017b). We have also had talks with their central offices about the need to adopt biosecurity measures in every port and for all ships. During our restoration projects, we give talks to personnel at SEMARs stations at the islands, but we also strive to provide training, so that there is always at least one person who knows about surveillance methods and early detection techniques on all islands with Navy stations.

The successful two-decade trajectory of island restoration in Mexico contributes to meet the country's goals in sustainable development and conservation (Aguirre-Muñoz, et al., 2016a). The National Biosecurity Programme must become a formally recognised, institutionalised, inter-agency, inter-sectorial agreement for it to be effective. We need to establish collaboration arrangements with several agencies, such as CONANP, SEMAR, SEMARNAT, the Federal Agency for Environmental Protection (PROFEPA), and port authorities. Once we are all working hand in hand, the restoration efforts for Mexico's island biodiversity will be reinforced and protected over the long term.

RECOMMENDATIONS AND LESSONS LEARNT

Outreach and environmental learning campaigns are of the utmost importance, and hence need to be permanent and not just for short periods of time. Only then, will people become aware of the problem and actually adopt the habits required to prevent the accidental introduction of IAS.

Working with the Protected Areas Advisory Council is the best strategy to strengthen biosecurity protocols. It also helps the project to become integrated with the area manager's work.

Communities that recently participated on an eradication project are more likely to be interested and active in keeping the island free of IAS.

Incursion response cases may have economic costs that are not specifically budgeted for, so the creation of a national biosecurity fund for emergencies is an important step forward.

Early detection alerts are a way of evaluating if the outreach campaign is working, so that even if it turns out to be just a false alarm, we now know people are aware that they should report if they see something different.

We need to sign and publish institutional collaboration agreements between government agencies in order to reinforce biosecurity measures and make sure all stakeholders comply with them.

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REFERENCES

Aguirre-Muñoz, A., Samaniego-Herrera, A., Luna-Mendoza, L., Ortiz-Alcaraz, A., Rodríguez-Malagón, M., Méndez-Sánchez, F., Félix-Lizárraga, M., Hernández-Montoya, J.C., González-Gómez, R., Torres-García, F., Barredo-Barberena, J.M. and Latofski-Robles, M. (2011a). 'Island restoration in Mexico: ecological outcomes after systematic eradications of invasive mammals'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 250–258. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Aguirre-Muñoz, A., Samaniego-Herrera, A., Luna-Mendoza, L., Ortiz-Alcaraz, A., Rodríguez-Malagón, M., Félix-Lizárraga, M., Méndez-Sánchez, F., González-Gómez, R., Torres-García, F., Hernández-Montoya, J.C., Barredo-Barberena, J.M. and Latofski-Robles, M. (2011b). 'Eradications of invasive mammals on islands in Mexico: the roles of history and the collaboration between government agencies, local communities and a non-government organisation'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 386–394. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Aguirre-Muñoz, A., Méndez-Sánchez, F., De la Rosa-Conroy, L., Latofski-Robles, M. and Manríquez-Ayub, A. (2013). *Diagnóstico de especies exóticas invasoras en las reservas de la biosfera y áreas naturales protegidas insulares seleccionadas, a fin de establecer actividades de manejo para las mismas.* Tercer informe de actividades presentado a la Comisión Nacional para el Conocimiento y Uso de la Biodiversidad dentro de la fase preparatoria del proyecto GEF "Aumentar las capacidades de México para manejar especies exóticas invasoras". Grupo de Ecología y Conservación de Islas, A.C. Ensenada, Baja California, México. 128 pp + 2 Anexos. <http://www.biodiversidad.gob.mx/especies/Invasoras/gef/pdf/04-diagnostico-seleccion-grupos-islas.pdf>.

Aguirre-Muñoz, A., Méndez-Sánchez, F. and Latofski-Robles, M. (2016a). 'Conservación y restauración de las islas Mexicanas'. In: M. Llano and H. Fernández (eds.) *Análisis y Propuestas para la Conservación de la Biodiversidad en México*, Pp. 66–71. Ciudad de México.

Aguirre-Muñoz, A., Samaniego-Herrera, A., Luna-Mendoza, L., Ortiz-Alcaraz, A., Méndez-Sánchez, F. and Hernández-Montoya, J. (2016b). 'La restauración ambiental exitosa de las islas de México: una reflexión sobre los avances a la fecha y los retos por venir'. In: E. Ceccon and C. Martínez-Garza (eds.) *Experiencias Mexicanas en la Restauración de los Ecosistemas*, pp. 487–512. Ciudad de México: Comisión Nacional para el Conocimiento y Uso de la Biodiversidad. Universidad Nacional Autónoma de México. Universidad Autónoma del Estado de Morelos.

Aguirre-Muñoz, A., Méndez-Sánchez, F., Latofski-Robles, M. and Salizzoni-Chávez, K. (2017a) 'Mejoras en el marco legal para la conservación y restauración integral de las islas de México'. In: *Reporte CESOP,* no. 103, Febrero 2017. pp. 19–23. Publicación mensual del Centro de Estudios Sociales y de Opinión Pública de la Cámara de Diputados.

Aguirre-Muñoz, A., Méndez-Sánchez, F., Latofski-Robles, M., Salizzoni-Chávez, K., Luna-Mendoza, L., Ortiz-Alcaraz, A., Hernández-Montoya, J., Bedolla-Guzmán, Y. and Rojas-Mayoral, E. (2017b). 'Avances en la conservación y restauración integral de las islas de México y mejoras al marco legal para consolidar la trayectoria positiva'. In: G. Born-Schmidt, F. de Alba, J. Parpal and P. Koleff (eds). *Principales Retos que Enfrenta México ante las Especies Exóticas Invasoras,* pp. 225–255. Ciudad de México: Centro de Estudios Sociales y de Opinión Pública. Cámara de Diputados / LXIII Legislatura.

Aguirre-Muñoz, A., Bedolla-Guzmán, Y., Hernández-Montoya, J., Latofski-Robles, M., Luna-Mendoza, L., Méndez-Sánchez, F., Ortiz-Alcaraz, A., Rojas-Mayoral, E., and Samaniego-Herrera, A. (2018). 'The conservation and restoration of the Mexican Islands, a successful comprehensive and collaborative approach relevant for global biodiversity'. In: Ortega-Rubio, A. (ed). *Mexican natural resources management and biodiversity conservation,* pp. 177-192. Springer International Publisher AG, part of Springer Nature.

Born-Schmidt, G., Parpal-Servole, J. and Koleff, P. (2017). 'De la elaboración a la implementación de la estrategia nacional sobre especies invasoras'. In: *Reporte CESOP,* no. 103, Febrero 2017, pp. 5–9. Publicación mensual del Centro de Estudios Sociales y de Opinión Pública de la Cámara de Diputados.

Broome, K. (2009). 'Beyond Kapiti – a decade of invasive rodent eradications from New Zealand islands'. *Biodiversity* 10: 14–24.

CANEI (Comité Asesor Nacional sobre Especies Invasoras). (2010). *Estrategia nacional sobre especies invasoras en México, prevención, control y erradicación.* Ciudad de México: Comisión Nacional para el Conocimiento y Uso de la Biodiversidad, Comisión Nacional de Áreas Naturales Protegidas, Secretaría de Medio Ambiente y Recursos Naturales.

CANTIM (Comité Asesor Nacional sobre el Territorio Insular Mexicano). (2012). *Estrategia Nacional para la conservación y el desarrollo sustentable del territorio insular mexicano.* Ciudad de México y Ensenada, B.C: Secretaría de Medio Ambiente y Recursos Naturales, Instituto Nacional de Ecología, Comisión Nacional de Áreas Naturales Protegidas, Secretaría de Gobernación, Secretaría de Marina-Armada de México y Grupo de Ecología y Conservación de Islas, A.C.

CBD (Convention on Biological Diversity). (2010). Strategic Plan for Biodiversity 2011–2020. *Aichi Targets.* <https://www.cbd.int/sp/targets/>. Accessed: 27 June 2017.

DOF (Diario Oficial de la Federación). (2016). *Decreto por el que se declara área natural protegida, con el carácter de reserva de la biosfera, la región conocida como islas del pacífico de la península de Baja California.* Ciudad de Mexico.

González-Martínez, A.I., Barrios-Caballero, Y. and De Jesús, S. (2017). 'Análisis de riesgo para especies invasoras en México'. In: *Reporte CESOP,* no. 103, Febrero 2017, pp. 24–29. Publicación mensual del Centro de Estudios Sociales y de Opinión Pública de la Cámara de Diputados.

IUCN (International Union for Conservation of Nature). (2017). *Commitments toward achieving the Honolulu Challenge.* <https://www.iucn.org/theme/species/our-work/invasive-species/honolulu-challenge-invasive-alien-species/commitments-towards-achieving-honolulu-challenge>. Accessed: 28 June 2017.

Latofski-Robles, M., Aguirre-Muñoz, A., Méndez-Sánchez, F., Reyes-Hernández, H. and Schlüter, S. (2014). 'Prioritizing restoration actions for the islands of Mexico'. *Monographs of the Western North American Naturalist* 7: 435–441.

Latofski-Robles, M., Méndez-Sánchez, F., Aguirre-Muñoz, A., Jáuregui-García, C. and Castro-Girón, A. (2016). *Diagnóstico de especies exóticas invasoras en seis áreas naturales protegidas insulares, a fin de establecer actividades para su manejo.* Reporte de actividades del año 1 (2015) dentro del proyecto GEF 0081866 "Aumentar las capacidades de México para manejar especies exóticas invasoras a través de la implementación de la Estrategia Nacional de Especies Invasoras". Grupo de Ecología y Conservación de Islas, A.C. Ensenada, B.C., México. <http://www.biodiversidad.gob.mx/especies/Invasoras/gef/pdf/2.1-1-reporte-extenso-gef-2015.pdf>.

Latofski-Robles, M., Méndez-Sánchez, F., Aguirre-Muñoz, A., Jáuregui-García, C., Salizzoni-Chávez, K. and Fernández-Ham, G. (2017). *Diagnóstico de especies exóticas invasoras en seis áreas naturales protegidas insulares, a fin de establecer actividades para su manejo.* Reporte de actividades del año 2 (2016) dentro del proyecto GEF 0081866 "Aumentar las capacidades de México para manejar especies exóticas invasoras a través de la implementación de la Estrategia Nacional de Especies Invasoras". Grupo de Ecología y Conservación de Islas, A.C. Ensenada, B.C., México. <http://www.biodiversidad.gob.mx/especies/Invasoras/gef/pdf/2.1-1-reporte-extenso-gef-2016.pdf>.

Llorente-Bousquets, J. and Ocegueda-Cruz, S. (2008). 'Estado del conocimiento de la biota'. In: J. Soberón, G. Halffter, and J. Llorente-Bousquets, (Eds.), *Capital natural de México Vol. I Conocimiento actual de la biodiversidad,* pp. 283–322. México, D.F.: Comisión Nacional para el Conocimiento y Uso de la Biodiversidad.

Matos, J., Little, A., Broome, K., Kennedy, E., Méndez-Sánchez, F., Latofski-Robles, M., Irvine, R., Gill, C., Espinoza, A., Howald, G., Olthof, K., Ball, M. and Boser, C.L. (2018). 'Connecting island communities on a global scale: Case studies in island biosecurity'. *Monographs of the Western North American Naturalist* 78(4): 959-972

Méndez-Sánchez, F., Aguirre-Muñoz, A., Latofski-Robles, M., Jáuregui-García, C., Luna-Mendoza, L, Ramos-Rendón, K., De la Rosa-Conroy, L., Félix-Lizárraga, M., Manríquez-Ayub, A., Castañeda-Rodríguez, I. and Guzmán-Cázarez, E. (2014). *Curso-taller de bioseguridad insular para administradores guardaparques y usuarios de áreas naturales protegidas.* Final Report to U.S. Fish & Wildlife Service, Wildlife Without Borders, Division of International Conservation. Grupo de Ecología y Conservación de Islas, A.C. Ensenada, B.C. 28 pp + 1 Anexo.

Moore, J.L., Rout, T.M., Hauser, C.E., Moro, D., Jones, M., Wilcox, C. and Possingham, H.P. (2010). 'Protecting islands from pest invasion: optimal allocation of biosecurity resources between quarantine and surveillance'. *Biological Conservation,* 143(5):1068–1078.

PII (Pacific Invasives Initiative). (2013). *Biosecurity Plan Template.* Resource Kit for Rodent and Cat Eradication. <http://rce.pacificinvasivesinitiative.org/tools/Templates.html >. Accessed 3 April 2018.

Parkes, J. (2013). *Strategies to detect and manage incursions of exotic species on Mexican islands.* Report presented to UNDP-Mexico and CONABIO as part of the preparatory phase of the GEF-ENCIS Project. Christchurch, New Zealand: Kurahaupo Consulting. <http://www.biodiversidad.gob.mx/especies/Invasoras/gef/pdf/09-edrr-islands-en.pdf>.

Reaser, J.K., Meyerson, L.A., Cronk, Q., De Poorter, M., Eldrege, L.G., Green, E. and Vaiutu, L. (2007). 'Ecological and socioeconomic impacts of invasive alien species in island ecosystems'. *Environmental Conservation* 34: 98–111.

Roberts, A. (2003). *Island biosecurity SOP Best Practice Manual*. Southland, New Zealand: Department of Conservation.

Russell, J.C., Towns, D.R. and Clout, M.N. (2008). 'Review of Rat Invasion Biology: Implications for Island Biosecurity'. *Science for Conservation* 286. Wellington, NZ: Department of Conservation.

Towns, D.R. (2011). 'Eradications of vertebrate pests from islands around New Zealand: what have we delivered and what have we learned?'. In: C.R. Veitch, M.N. Clout and D. Towns (eds.) *Island invasives: eradication and management*, pp. 364–371. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Whittaker, R.J. and Fernández-Palacios, J.M. (2007). *Island Biogeography: Ecology, Evolution, and Conservation.* Oxford: Oxford University Press.

Achieving post-eradication biosecurity on South Georgia

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Abstract The world's largest island rodent eradication programme to date was carried out on South Georgia between 2009 and 2018 (baiting on island in 2011, 2013 and 2015) by the South Georgia Heritage Trust (SGHT). A comprehensive survey in 2017/18 found no signs of rodents. Although SGHT planned and executed this eradication under permits issued by (and with collaboration from) the Government of South Georgia and the South Sandwich Islands (GSGSSI), the scale and complexity of the multi-year project was not conducive to prior agreement on post-eradication biosecurity to prevent rodent re-invasion. Thus by 2013, two years after initial baiting, biosecurity measures for rodents remained inadequate, relying mainly on rodent detection boxes on vessels and at the island's main point of entry. The more substantive post-eradication biosecurity measures implemented by other administrations were absent. In late 2014, after more than three years with no rodent sign, there was unambiguous evidence of a rat within the island's settlement. This coincided with a vessel berthed alongside a nearby jetty. Between 2015 and 2017, SGHT formally submitted recommendations to the GSGSSI on enhanced biosecurity provisions. Some of these recommendations have been implemented but arguably the most important, relating to vessel berthing and secure handling of imported cargo, remain to be addressed effectively. We summarise what remains to be done, recognising the logistic and financial challenges involved, but conscious that, until all measures are in place, there is significant risk of re-infestation of South Georgia by invasive rodents, compromising a decade of work (and funding) by multiple stakeholders.

Keywords: baiting, poison, rodents, sub-Antarctic

INTRODUCTION

Tentative plans for the possible eradication of rodents from the sub-Antarctic island of South Georgia date back to 2000 (S. Poncet, pers. comm.). The subsequent success of the eradication programme on Campbell Island (McClelland & Tyree, 2002; Towns & Broome, 2003) encouraged the Government of South Georgia and the South Sandwich Islands (GSGSSI) to undertake a feasibility study of the practicalities of a large-scale (island-wide) eradication of rats and mice on South Georgia (Christie & Brown, 2007). However, due to resource limitations at the time, the Government opted not to proceed.

Nevertheless, the small UK charity (NGO) South Georgia Heritage Trust (SGHT) agreed to take up the challenge and started to develop fundraising and project management structures and initiatives to address this. GSGSSI accepted the involvement and lead role of SGHT in principle and practice, subject to the project conforming to the relevant legislation and permitting processes.

In 2009, SGHT established a Steering Committee (SC) to oversee the management of the whole operation. The SC comprised Trustees from SGHT and the Friends of South Georgia Island (FOSGI), key GSGSSI officials (Chief Executive and Environmental Officer), representatives of British Antarctic Survey (BAS), and the SGHT Project Director (Prof. Tony Martin of the University of Dundee). The SC met quarterly from 2010 to 2015 and its main roles were to ensure the effective execution of the plans for the acquisition and shipment of equipment, vessels, helicopters and staff, and that all documentation required by the regulatory authorities (mainly GSGSSI but also the UK's Civil Aviation Authority (CAA)) was submitted on time. A list of all such documentation can be found at <http://www.sght.org/newsletters-and-publications/>.

From 2010, there followed three phases of baiting using brodifacoum poison bait distributed by helicopters. Each baiting phase was spaced two years apart (2011, 2013 and 2015) to allow both for further fundraising between baiting seasons and for evaluation of methods and results before proceeding further. This work is reported on elsewhere (Martin, 2015; SGHT, 2016; Martin & Richardson, 2017).

In the 2017/18 austral summer, a comprehensive monitoring survey, organised and led by SGHT in collaboration with GSGSSI, was undertaken to determine the results of the eradication project. The five-month survey deployed over 1,500 inert devices (chew-sticks, tunnel and camera traps and analogous devices) and, augmented by trained rodent-detection dogs (which travelled 2,420 km), covered a minimum of 8,600 ha across 120 sites. No signs of rodents were detected, allowing the conclusion that the eradication phases had been successful.

This paper aims to review the rodent-related biosecurity status of South Georgia before and during the eradication project and to summarise proposals to enhance this in the light of events during the project, and after its successful conclusion. It highlights the remaining measures to be implemented to minimise the risk of inadvertent re-introduction of rodents.

RODENT BIOSECURITY AT SOUTH GEORGIA PRIOR TO 2014

The need for biosecurity measures to be integral to any eradication efforts on South Georgia was recognised back in 2007, with a governmental report stating then that: *'First and foremost, an effective and robust biosecurity regime needs to be in place on South Georgia before eradication is attempted'* (Christie & Brown, 2007).

Although SGHT submitted Biosecurity Plans to GSGSSI for each of the three phases of baiting, those plans dealt with biosecurity solely in relation to the operational requirements of the project itself – for example, the importation into South Georgia of materials needed for the baiting operation, or the movement of equipment, including helicopters, within the island. The wider issue of South Georgia's biosecurity (the responsibility of Government), was not addressed in discussions between SGHT and GSGSSI either before or during the earlier years of the eradication project. In hindsight, this lapse was the result of both organisations trying at that time to cope with the considerable challenges of the baiting operations. Faced

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with what was clearly going to be a multi-year, complex operation it would have been difficult in the initial 'proof-of concept' stages of the project to have developed a realistic and pragmatic prescription for post-eradication biosecurity. In consequence, biosecurity arrangements were held, at least in respect of rodents, under relatively rudimentary provisions. For example, the governmental Biosecurity Protocols of 2013 and 2014 (GSGSSI, pers. comm. 2013 and 2014) did little more than stipulate the need for rat guards on vessels, the deployment of rodent bait boxes (a requirement for yachts only), and the requirement that all vessels be inspected for the presence of rodents.

Despite what, in retrospect, was a deficiency in project planning, the HR Project progressed well. By mid-2014 (more than three years after the initial baiting) the Phase 1 area (c. 14,000 ha) had been tentatively declared free of rodents, and a relatively extensive survey in March of that year by SGHT detected no signs of rodents in the 60,000 ha. of the more extensive Phase 2 area.

Deficiencies in the biosecurity provisions became evident on 23 October 2014 when the unambiguous signs of a rat were seen in newly fallen snow at King Edward Point (KEP) – the administrative centre of the island in the heart of the Phase 1 area. The Government rapidly set in train its contingency plan for just such an incident. Brodifacoum bait was spread by hand out to an arc perimeter of 1.5 km from the sighting and many more rat traps were placed around the KEP base area. In the event, no more sign of this animal was seen; nor was a corpse found. This was unfortunate since, through DNA analysis, the origin of this lone animal could have been determined (see Piertney, et al., 2016). The presumption was that the rat succumbed to the poison bait.

The origins of this one known rat could only be speculated on. It could have been a survivor (or offspring of a survivor) from the 2011 baiting phase. However, this is unlikely in the most inhabited part of South Georgia where there had been no rodent signs in the preceding 3.5 years since baiting. Alternatively, it could have been imported in one of the small vessels based at KEP from another part of the island yet to be baited or swam ashore from a vessel anchored offshore. The latter scenarios are not impossible but seem implausible. Given the coincidental timing, the most probable source for this rat was from one or other of two vessels that had recently tied-up alongside the nearby KEP jetty. Records showed that one vessel had visited a number of times between 5–22 October whilst another vessel had arrived on 22 October and was still moored alongside the jetty the following day at the time of the incident (GSGSSI, in litt. to SGHT).

The general conclusion was that this latter vessel was the most likely source of this incursion. The Government concurred through a statement that "*the rat was most likely to have originated from a ship tied up at KEP in the previous days, though it was impossible to prove this*" (GSGSSI, 2015 in litt. to SGHT).

Whatever the means of introduction, this rat had managed to evade all prevention and detection measures in place at the time – bait boxes and traps deployed both on the vessels and extensively around the base area. Its presence was only detected due to recent snow cover.

RODENT BIOSECURITY AT SOUTH GEORGIA SINCE 2014

Although this incident apparently involved only a single animal, SGHT assumed that it would rapidly trigger a major Government-led review of biosecurity arrangements, in order to implement more robust measures. However, it was mid-2015, following completion of the last phase of baiting, that the Government requested SGHT input to an apparent major review of South Georgia's biosecurity arrangements. The SGHT response, submitted in late July, was a series of 10 recommendations to enhance island-wide biosecurity (Table 1).

These recommendations were based on the fact that, with aircraft unable to operate into South Georgia, the re-introduction of rodents to South Georgia could only come about via shipping. That is: by shipwrecks on the coast, or by animals swimming ashore from a vessel; gaining access along mooring warps or down gangplanks; or coming ashore in cargo or luggage. Although none of these potential introduction pathways can be ruled out, the greatest risk of a rodent re-introduction to South Georgia is most likely to be via one or other of the last two routes.

SGHT's recommendations included the requirement to maintain an adequate supply (at least three tonnes) of in-date brodifacoum bait at KEP, the need for a series of pre-baited box traps (which would be inspected frequently) around the base area, and the deployment of effective rat-guards on vessels moored alongside.

The Trust's four main recommendations are shown in bold in Table 1. These were: the use of rodent-detection dogs at ports in the Falkland Islands and on vessels destined for South Georgia; prohibiting the mooring alongside of vessels except for tightly prescribed activities; the erection of rodent-proof fences around offloading jetties in South Georgia; and the construction of rodent-proof containment areas at KEP within which shipping containers and other large-scale cargo could be held, and unpacked, in a biosecure manner.

Totally eliminating the risk of a rodent reintroduction to South Georgia cannot be guaranteed. However, SGHT was of the view that comprehensive implementation of its recommendations would very substantially reduce the risk of rodents either getting to South Georgia in the first place or, if that failed, at least preventing their escape from the immediate surroundings of the cargo unloading/ handling areas at KEP/Grytviken. The recommendations were considered by SGHT to be necessary, realistic, practical, cost-effective (especially in terms of the cost of mounting a subsequent eradication operation) and based on international best practice.

The presumption was that these proposed provisions would be included within a new, strengthened governmental Biosecurity Plan. Instead, the Biosecurity Handbook, published in December 2015 (GSGSSI, 2015) simply re-stated the existing provisions. It took no account of the SGHT recommendations. This caused SGHT to re-state its case to GSGSSI in January 2016 and again in February 2017 (SGHT, 2016; 2017 in litt. to GSGSSI). Unfortunately, we are unaware of any substantive change in biosecurity practices at South Georgia, with one important exception, relating to the trial use of rodent-detection dogs (see below).

IMPLEMENTATION OF BEST PRACTICE BIOSECURITY AT SOUTH GEORGIA

GSGSSI's policy, in principle, over biosecurity is rightly predicated (as is best practice) on the concept that: *"The most effective way of dealing with biosecurity is to have pre-border measures in place...the aim is to prevent an alien reaching the island, not try and deal with it on arrival"* (Christie & Brown, 2007). Furthermore, in its five-year Strategy Paper (GSGSSI, 2016) the Government advocated that *"Biosecurity protocols should be reviewed on a regular basis and best practice adopted"*

Table 1 Biosecurity recommendations* submitted to South Georgia Government (GSGSSI) – 2015/16 and 2017.

Recommendation	Implementation
Vessels/cargo checked by rodent-detection dogs (in Falklands) then during transit to, and on arrival at, South Georgia.	Trial underway in 2018
Vessels (other than yachts) must be prohibited from mooring alongside except when unloading/loading cargo or other strictly prescribed activities; then for minimum time only. In all other circumstances vessels must either anchor off or moor to buoy.	Only partial
When moored alongside, or to the shore, all mooring warps must have effective rat guards fitted.	Yes, though design of guards needs further attention
When moored alongside, gangway ashore must only be in place when necessary, and for minimum time.	?
Rodent-proof fence must be constructed around every dock area (KEP/ Grytviken)	None
No loose cargo (other than personal effects) must be offloaded. All cargo must be carried in sealed shipping containers which must be (a) fumigated, and (b) contain rodent bait stations.	Only partial
A rodent-proof containment area suitable for shipping containers must be constructed at KEP. Containers must only be opened and unpacked within the containment area.	Under consideration. Construction potentially starting 2019; completion 2021?
In the event of known or suspected rodent incursion, pre-planned response action must be activated immediately. Must include setting of traps and spreading poison out to stipulated radius from incursion.	Yes
A network of pre-baited trap boxes must be installed permanently around any dock area and checked frequently (daily when vessel moored alongside).	Yes
Suitable quantity (minimum 3 tonnes) in-date brodifacoum bait must be held at KEP as contingency. Such bait must be replenished as appropriate.	Yes, but whether in date is unknown

* Recommendations in bold are the most substantive ones.

In relation to the recommendations of SGHT (Table 1) and practices prevailing at the closest analogue operation, that following the comprehensive and successful eradication programme on Macquarie Island (Springer, 2016), we review the current situation at South Georgia below:

a) Pre-border measures: cargo checking on embarkation and in transit

In its most recent policy statement (GSGSSI, 2017), pre-border biosecurity measures in relation to rodents rely principally on the use of rat-guards on vessels, requiring rodent detection boxes to be carried onboard vessels and the use of bait stations within cargo shipping containers. However, even taken together, we contend that these measures are unlikely to be effective. For example, despite GSGSSI having trialled a number of rat-guard designs, none to date has proved capable of coping with the harsh weather conditions prevailing in South Georgia.

At Macquarie Island, the deployment of rodent-detection dogs is now routine. Dogs check all cargo twice before it departs Australia and then again during passage to, and on arrival, at Macquarie where the environs of the research station are then also subject to survey by dogs. With financial assistance from SGHT, GSGSSI embarked in early 2018 on a trial deploying rodent-detection dogs at embarkation points in the Falkland Islands and on ships destined for South Georgia. This is a very welcome initiative which, it is hoped, will be converted into a permanent procedure.

Nevertheless, the most likely pathway for a rodent to gain access to South Georgia is from vessels moored alongside a jetty at KEP/Grytviken either via offloaded cargo or by simply "jumping ship".

b) Vessel mooring

At Macquarie Island, the risk of further rodent invasion is reduced still further by there being no harbour or jetty facilities on the island. This means that, unlike South Georgia, all cargo transfers from ship to shore are performed either by helicopters or amphibious lighters, enabling more stringent biosecurity checks to be made. Vessels anchoring well offshore, beyond the swimming distance of rats, reduce the risk that any shipborne rodents may gain direct access to the island through their ability (documented in both the UK and Falklands) to swim up to 1–2 km between, or out to, islands. The Macquarie situation has the added advantage that ship movements are confined largely to transits between Tasmania and the island. This again enables far greater biosecurity control.

At South Georgia, in contrast, whilst ships depart to South Georgia from a variety of locations (e.g. South American ports), those that are currently allowed to tie-up alongside at KEP are invariably governmental vessels arriving from the Falklands, where the embarkation ports are known to be infested with rats and lack fully appropriate facilities for biosecure handling of cargo. SGHT has recommended that the practice of mooring alongside, the most likely route for a rodent incursion, should be prohibited, except for cargo handling and other closely prescribed activities (such as undertaking necessary mechanical repairs to a vessel or for safety). The current criteria allowing alongside mooring include activities such as the "transfer of personnel, or "allowing for crew rest periods". Given the biosecurity risks that alongside mooring poses, convenience per se should not be a valid justification for continuation of this practice. This is particularly so given that the many thousands of tourists who visit South Georgia (and KEP/Grytviken) annually do so from vessels anchored offshore.

c) Cargo-handling ashore

Biosecurity handling facilities at KEP are currently restricted to a single shed where small-scale cargo can be unpacked and checked within a confined space. There are no facilities, in the form of a rodent-proof fenced area, within which large-scale cargo (i.e. shipping containers) can be stored and then opened securely. Such a facility, coupled with rodent-proof fencing around the jetty areas at KEP/Grytviken, would provide some measure of constraint for any rodent that either managed to escape from a vessel or survived inside a shipping container.

It is evident that the rodent incursion of October 2014 was of an animal that had apparently circumvented both shipboard measures and the numerous traps and bait boxes around KEP. Those measures, at least then, had proved wanting.

BIOSECURITY COSTS AND RISKS

It is important to contrast the respective costs of eradication and biosecurity. Investment in the eradication project by SGHT has been considerable, with direct costs of around £7.5 million, rising to c. £10 million when indirect costs are included. Over 80% of the project costs have been raised through charitable donations and sponsorship; although GSGSSI provided extensive staff and logistic assistance throughout the project it made no other contribution to direct costs. In contrast, we estimate that the capital costs of the additional recommended biosecurity measures are unlikely to exceed £0.5 million. In December 2016, SGHT offered to fundraise to help pay for those capital costs.

SGHT recognises that implementation of the more substantive measures would come with additional costs (including ongoing maintenance costs), alternative risks, and the need for specific design considerations to meet South Georgia's harsh conditions. For example, any rodent-proof structure on the island must be able to withstand extremely high wind loadings, ice and snow accumulation as well as the attention of other wildlife such as southern elephant seals (*Mirounga leonina*).

Even significantly reducing the practice of alongside mooring would not entirely eliminate risk. The alternatives are either anchoring offshore or mooring to a suitable buoy. Large vessel buoyage is no longer available at South Georgia and its provision and maintenance would be both expensive and not without liability for the regulatory authority (GSGSSI). Vessels at anchor can also be at risk. Weather conditions at South Georgia can change at short notice and be severe. In March 2000, three long-lining fishing vessels were driven ashore one night in Cumberland Bay in extreme weather conditions. One managed to re-float; the other two were completely wrecked on an inshore reef. Whether there were rodents on either of those vessels, and whether they escaped ashore is not known but the incident emphasises that some risks of rodent re-introduction will always remain. This makes it even more imperative to address those risks which can be mitigated or eliminated.

CONCLUSIONS

The South Georgia Habitat Restoration Project has been the largest island rodent eradication yet undertaken. The overall effectiveness of the three seasons of baiting over five years with brodifacoum has recently been confirmed following a comprehensive monitoring survey in the 2017/18 season. This found no signs of rodents in any of the baited areas, leading to the conclusion that the eradication programme has been successful. Concurrently, the increase in the numbers and distribution of some breeding birds (including endemic species) since baiting, has already been dramatic.

Notwithstanding that result, two major lessons can be taken from this important project; one highly positive, the other less so. On the former, large scale eradications (e.g. Campbell Island (McClelland & Tyree, 2002) and Macquarie Island (Springer, 2016)) have usually relied on the extensive resources of governments. In contrast, the South Georgia project has demonstrated that, through extensive fundraising and competent project planning and implementation, even relatively modest or small-sized NGOs can take on the challenge of large-scale eradications.

The downside of the South Georgia operation has been the lack of a close synergy between eradication and biosecurity. Again, previous large eradications have had the benefit of intra-governmental co-ordination with often the same governmental agency (e.g. New Zealand's Department of Conservation or the Tasmanian Parks and Wildlife Service) having responsibility for both elements. The problems in the case of South Georgia were complicated by the fact that two organisations, of highly contrasting status, undertook or were responsible for the eradication and biosecurity elements.

To ensure that these two equally important aspects are taken forward in close harmony, we make the following recommendations for future rodent eradication projects. That:

- adequate biosecurity measures must be in place before, during and after eradication;
- in those instances where responsibilities for eradication and biosecurity may reside with different organisations, agreement must be reached in advance between those entities; and that:
- such agreements should set out the respective responsibilities, objectives and timetables for both parties before eradication is allowed to commence.

Experience from South Georgia has shown that in the absence of any such prior agreements, eradication and biosecurity may get out of step either in their timing or effectiveness – or both. Such a situation creates a potential risk that the considerable investment in eradication and its corresponding environmental benefits may be jeopardised subsequently by inadequate biosecurity provisions.

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REFERENCES

Christie, D. and Brown, D. (2007). *Recommendations and Concerns Regarding Proposed South Georgia Rat Eradication*. Stanley, Falkland Islands: Report prepared for the Government of South Georgia and the South Sandwich Islands. <http://www.sght.org/newletters-and publications/>.

GSGSSI. (2015). *Biosecurity Handbook*. Stanley, Falkland Islands Government of South Georgia and the South Sandwich Islands. <http://www.gov.gs/docsarchive/Environment/Biosecurity/Biosecurity%20Protocols%202015.pdf>.

GSGSSI. (2016). *South Georgia and the South Sandwich Islands Strategy 2016–2020*. Stanley, Falkland Islands: Government of South Georgia and the South Sandwich Islands. <http://www.gov.gs/docsarchive/GSGSSI/Strategy/Final%20Published%20Strategy%20-%20PDF%20Version.pdf>.

GSGSSI. (2017). *Biosecurity Handbook 2017–2018*. Stanley, Falkland Islands: Government of South Georgia and the South Sandwich Islands. <http:www.gov.gs/docsarchive/Environment/Biosecurity/Biosecurity%20Handbook2017-2018FINAL.pdf>.

Martin, A.R. (2015). *Reclaiming South Georgia*. South Georgia Heritage Trust, Dundee.

Martin, A.R. and Richardson, M.G. (2017). 'Rodent eradication scaled up: clearing rats and mice from South Georgia'. *Oryx*: doi.org/10.1017/S003060531700028X.

McClelland, P. and Tyree, P. (2002). 'Eradication: the clearance of Campbell Island'. *New Zealand Geographic* 58: 86–94.

Piertney, S.B., Black, A., Watt, L., Christie, D., Poncet, S. and Collins, M.A. (2016). 'Resolving patterns of population genetic and phylogeographic structure to inform control and eradication initiatives for brown rats *Rattus norvegicus* on South Georgia'. *Journal of Applied Ecology* 53: 332–339.

SGHT. (2016). *The South Georgia Habitat Restoration Project; Final Report*. Dundee, Scotland, UK: South Georgia Heritage Trust. <http://www.sght.org/wp-content/uploads/2018/07/SGHT-Habitat-Restoration-Project-final-report.pdf>.

Springer, K. (2016). 'Methodology and challenges of a complex multi-species eradication in the sub-Antarctic and immediate effects of invasive species removal'. *New Zealand Journal of Ecology* 40: 273–278.

Towns, D.R. and Broome, K.G. (2003). 'From small Maria to massive Campbell: forty years of rat eradication from New Zealand islands'. *New Zealand Journal of Zoology* 30: 377–398.

Beyond biodiversity: the cultural context of invasive species initiatives in Gwaii Haanas

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Abstract Haida Gwaii is a remote island archipelago located off the north Pacific coast, approximately 100 km from mainland British Columbia, Canada. The southern part of the archipelago, Gwaii Haanas, is designated a National Park Reserve, National Marine Conservation Area Reserve, and Haida Heritage Site, and is cooperatively managed by the Government of Canada and the Council of the Haida Nation. Haida Gwaii has been the site of many invasive mammal introductions including rats, racoons and deer. It was also where the first successful North American rat eradication took place, on Langara Island in the 1990s. Since then, a number of invasive mammal eradication and control projects have taken place in Gwaii Haanas. Most recently, a project to eradicate invasive deer on six islands began in 2017 and is ongoing. The conservation gains from eradications are well documented in various systems around the globe, but the cultural context in Gwaii Haanas is unique. The Gwaii Haanas experience presents an example of how active restoration can move beyond biodiversity to develop IAS projects that are also culturally meaningful.

Keywords: cooperative management, cultural resource, Gwaii Haanas, Haida Gwaii, traditional knowledge, traditional use

INTRODUCTION

Invasive alien species (IAS) are a significant driver of declines in species and ecosystem services worldwide (Pejchar & Mooney, 2009). In Canada, IAS negatively affect 22% of endangered species (Wilcove, et al., 1998; Venter, et al., 2006). Although invasion science spans the fields of social science, ecology, economics and conservation biology (Simberloff, et al., 2013), it rarely considers the cultural context in which invasive species eradications take place. For example, the impacts of IAS on cultural practices including the harvest of traditional foods and medicines are not usually considered.

In this paper, we describe IAS management in Gwaii Haanas, a land-and-sea protected area located in northern North America that is cooperatively managed by the Council of the Haida Nation and the Government of Canada. The Gwaii Haanas experience shows how cooperation between indigenous and federal governments in IAS initiatives allows projects to expand beyond biodiversity to become culturally meaningful as well.

STUDY AREA

Haida Gwaii is an archipelago of over 350 islands located approximately 100 km off the coast of British Columbia, Canada. Gwaii Haanas is a 5,000 km^2 protected area in the southern third of Haida Gwaii. It is known for its diverse ecosystems, distinctive flora and fauna, rich marine life, and living Haida culture.

Gwaii Haanas first garnered international attention in 1985, when the Council of the Haida Nation (CHN) led a non-violent blockade on Athlii Gwaay (Lyell Island) to protest against logging in the area. The CHN declared Gwaii Haanas (terrestrial and marine) a Haida Heritage Site in the same year. Soon after, the Government of Canada designated the Gwaii Haanas terrestrial area a National Park Reserve, and cooperative management of the land began in 1993 when the two governments signed the *Gwaii Haanas Agreement*. In 2010, the CHN and the Government of Canada signed the *Gwaii Haanas Marine Agreement* (2010), which committed the two governments to cooperative management of the Gwaii Haanas marine area. In the same year, Gwaii Haanas was designated a National Marine Conservation Area Reserve by the Government of Canada.

Gwaii Haanas is cooperatively managed by the Council of the Haida Nation and the Government of Canada through the Archipelago Management Board (AMB). The AMB is made up of equal representation from the Haida and Canadian governments and is responsible for all aspects of planning, operation, management and use of Gwaii Haanas. Decisions are made by consensus and based on the constitutions of both nations (Canadian Constitution Act, 1982, Haida Nation Constitution, 2014).

ECOLOGICAL CONTEXT

Haida Gwaii is ecologically significant in a global context. Supporting approximately 1.5 million breeding seabirds of 12 different species, it is the nesting location for one half of the entire global breeding population of ancient murrelets (*Synthliboramphus antiquus*) and for one fifth of the world's breeding Cassin's auklets (*Ptychoramphus aleuticus*) (Harfenist & Cober, 2003). Many of these important seabird breeding areas are situated within Gwaii Haanas.

There are 10 extant native mammals in Haida Gwaii, including the genetic sub-variant black bear (*Ursus americanus carlottae*). Currently, there are 12 introduced terrestrial vertebrates on Haida Gwaii, including two rats (*Rattus rattus* and *R. norvegicus*) and the Sitka black-tailed deer (*Odocoileus hemionus*), which have been the focus of recent eradication efforts in Gwaii Haanas.

Although Gwaii Haanas is protected in legislation, seabird habitat quality has continued to decline as a result of the impacts of IAS such as rats, raccoons (*Procyon lotor vancouverensis*) and Sitka black-tailed deer. Indeed, introduced species have been identified as the main threat to ecological integrity in Gwaii Haanas (AMB, 2018). Impacts from deer browsing include simplification of the

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forest structure and lack of regeneration of culturally and ecologically important species such as western redcedar (*Thuja plicata*) and yellow cedar (*Chamaecyparis nootkatensis*) (Pojar, 2008). Rats have had similarly devastating impacts on seabirds. For example, some islands that previously had murrelet colonies numbering up to 8,000 breeding pairs, are now effectively at zero (Rodway, et al., 1994).

HAIDA CULTURAL CONTEXT

Archaeological evidence of human habitation in Gwaii Haanas goes back more than 12,000 years (Fedje, et al., 2011). It is estimated that perhaps more than 30,000 Haida lived in villages and seasonal camps across the archipelago. Following European contact, diseases such as smallpox and influenza reduced the Haida population to as low as 550 and the remaining people gathered in two villages. These villages have since developed into the present-day towns of Old Massett and Skidegate. The current Haida population estimate is 5,000, and approximately half live on Haida Gwaii.

The Council of the Haida Nation is mandated to steward the lands and waters of Haida territory on behalf of the Haida Nation, including the perpetuation of Haida language, culture, art and traditional ways, for future generations (Haida Nation, 2017). No treaty was ever signed for Haida Gwaii and, in 2002, the Haida Nation filed a legal case with the Supreme Court of Canada for the title to Haida Gwaii. This case challenges the idea that Haida Gwaii is owned by the Canadian government.

While this ownership dispute is resolved in the courts, Gwaii Haanas continues to be managed by the AMB. This is possible because the dispute over title to Gwaii Haanas is explicitly laid out in the *Gwaii Haanas Agreement* (1993), which also describes how the two parties will set aside that disagreement in order to focus on shared objectives concerning the care, protection and enjoyment of the archipelago.

IAS PROJECTS IN GWAII HAANAS

Haida Gwaii was the first place in North America to carry out a successful rat eradication, on Langara Island in 1995, and remains the world's largest bait station-based eradication. The Council of the Haida Nation provided direction, and several Haida community members worked on this eradication, which used bait station transects throughout the 32.7 km^2 Island.

In Gwaii Haanas, rats were removed from four islands through the SGin Xaana Sdiihltl'lxa (Night Birds Returning) project (2009–2016). All were successful, and today three islands remain rat free and one has been reinvaded. The origin of the reinvasion is unclear but genetic testing results show that it is not related to rats from the adjacent island or those that were present prior to the eradication in 2011. In 2017, a deer eradication project, Llgaay gwii sdiihlda (Restoring Balance), began on six islands in Gwaii Haanas. This project is currently in progress and aims to restore the forest understorey community that has been decimated by deer browsing.

With the deer population reduction, there is evidence of recovery including culturally important plants such as:

- Ts'uu (western redcedar – *Thuja plicata*) – construction, carving;
- Kayd (Sitka spruce – *Chamaecyparis nootkatensis*) – roots, pitch, construction;
- K'aang (hemlock – *Tsuga heterophylla*) – fish hooks, food;
- Ts'iihlinjaaw (devil's club – *Oplopanax horridus*) – medicine;
- SGiidllGuu (huckleberry – *Vaccinium parvifolium*) – medicine, berries;
- Hldaan (blueberry – *Vaccinium alaskaense* and *V. ovalifolium*) – wooden pegs, berries;
- Sk'idGan (salal – *Gaultheria shallon*) – berries.

A HAIDA PERSPECTIVE ON IAS

Invasive species on Haida Gwaii have a direct impact on Haida culture, including:

- Loss of culturally significant plants and animals including western redcedar, a species important for carving, weaving and house building;
- Lack of opportunity to access medicinal and edible plants;
- Loss of opportunities to pass on knowledge of traditional harvesting teachings between generations; and
- Loss of traditional food sources such as fruiting plants, seabirds and seabird eggs.

These losses affect Haida citizens' ability to exercise their rights and practice their culture.

The primary objective of the Council of the Haida Nation (CHN) is to achieve legal title to Haida Gwaii and the surrounding waters. However, the CHN also works to achieve conservation gains in Haida Gwaii. For example, Land Use Orders and forest management based on ecosystem management principles have been implemented to protect culturally significant areas and to ensure Haida values are maintained in areas where logging occurs. In addition, a 1000-year cedar strategy is in development. This strategy will ensure that there are large, monumental cedars available in perpetuity for Haida Nation citizens to utilise for pole or canoe carving, weaving or house building projects. The CHN has also established new protected areas and designed effectiveness monitoring programmes, collecting data to support decision-making and research.

While the CHN has participated in many IAS-related projects, collaborating with provincial and federal governments, it is now working to develop a broader vision and targets concerning IAS on Haida Gwaii. This involves setting priorities and assessing current-day challenges. For example, the invasive Sitka black-tailed deer has become a significant food source for the Haida community and its hides, hooves and antlers are now incorporated into ceremonies. Therefore, initiatives need to balance the impacts of IAS on culturally important species while also considering the value these species have in the present-day culture. Generally, CHN-led IAS initiatives will focus on species that impact cultural activities, with a goal of managing IAS on Haida Gwaii in order to create healthy ecosystems for future generations.

HAIDA PLANNING PRINCIPLES

The CHN applies several basic principles to all planning initiatives, including invasive species management. These principles are:

Yahguudang – *Respect*

Respect for each other and all living things is rooted in our culture. We take only what we need, we give thanks, and we acknowledge those who behave accordingly.

'Laa guu ga kanhllns – *Responsibility*

We accept the responsibility passed on by our ancestors to manage and care for our sea and land. We will ensure that our heritage is passed onto future generations.

Gina 'waadluxan gud ad kwaagid – *Interconnectedness*

Everything depends on everything else. The principle of interconnectedness is fundamental to integrated planning and management. This comprehensive approach considers the relationships between species and habitats and accounts for short-term, long-term and cumulative effects of human activities on the environment. Interrelationships are accounted for across spatial and temporal scales and across agencies and jurisdictions.

Giid tlljuus – *Balance*

The world is as sharp as the edge of a knife. Balance is needed in our interactions with the natural world. If we aren't careful in everything we do, we can easily reach a point of no return. Our practices and those of others must be sustainable.

Gina k'aadang.nga gii uu til k'anguudang – *Seeking wise counsel*

Our elders teach us about traditional ways and how to work in harmony. Like the forests, the roots of our people are intertwined. Together we consider new ideas and information in keeping with our culture, values and laws.

Isda ad dii gii isda – *Giving and receiving*

Reciprocity is a respected practice in our culture, essential in our interactions with each other and the natural world. We continually give thanks to the natural world for the gifts that we receive.

CONCLUSION

The ecological impacts of IAS eradications are well studied and documented by the ecological research community. Less-often considered is the importance of eradications to cultural integrity and the continuity of indigenous cultures such as the Haida Nation. The Gwaii Haanas experience demonstrates how partnerships with indigenous governments can broaden the scope of, and support for, IAS projects and make them culturally as well as ecologically meaningful.

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REFERENCES

AMB. (2018). 'Draft Gwaii Haanas Land-Sea-People Management Plan for the National Park Reserve, National Marine Conservation Area Reserve, and Haida Heritage Site'. <https://www.pc.gc.ca/en/pn-np/bc/gwaiihaanas/info/public>.

Canadian Constitution Act (1982). 'The Constitution Act, 1982, being Schedule B to the Canada Act 1982 (UK)'. <http://canlii.ca/t/ldsx>.

Fedje, D., Mackie, Q., Lacourse, T. and McLaren, D. (2011). 'Younger Dryas environments and archaeology on the Northwest Coast of North America'. *Quaternary International* 242: 452–462.

Gwaii Haanas Agreement. (1993). 'Gwaii Haanas Agreement'. <www.haidanation.ca/wp-content/uploads/2017/03/GwaiiHaanasAgreement.pdf>. Accessed July 2017.

Gwaii Haanas Marine Agreement. (2010). 'Gwaii Haanas Marine Agreement'. <www.pc.gc.ca/en/pn-np/bc/gwaiihaanas/info/index/~/media/662F1B36258E462DA46C2E8372B62E3A.ashx>. Accessed July 2017.

Haida Nation Constitution. (2014). 'Constitution of the Haida Nation'. <www.haidanation.ca/wp-content/uploads/2017/03/HN-Constitution-Revised-Oct-2014_official-unsigned-copy.pdf>. Accessed July 2017.

Harfenist, A. and Cober, A. (2003). *Seabird Colonies Background Report for Haida Gwaii/Queen Charlotte Islands Land Use Plan*. Report prepared for Ministry of Forests, Lands and Natural Resource Operations, Queen Charlotte, BC. <www.for.gov.bc.ca/tasb/slrp/lrmp/nanaimo/haidagwaii/docs/Seabird-Rpt-Low-Resolution.pdf>. Accessed July 2017.

Pojar, J. (2008). 'Changes in Vegetation of Haida Gwaii in Historical Time'. In: A.J. Gaston, T. E. Golumbia, J-L. Martina, and S.T. Sharpe (eds.) *Lessons from the Islands: Introduced Species and What They Tell Us about how Ecosystems Work. Proceedings from the Research Group on Introduced Species 2002 Symposium,* pp. 32–38. Queen Charlotte, BC: Government of Canada.

Pejchar, L. and Mooney, H.A., (2009). 'Invasive species, ecosystem services and human well-being'. *Trends in Ecology and Evolution* 24(9): 497–504.

Rodway, M.S., Lemon, M.J.F. and Kaiser, G.W. (1994). *British Columbia Seabird Colony Inventory: Report 6 – Major Colonies on the West Coast of Graham Island.* Canadian Wildlife Service, Pacific and Yukon Region Technical Report Series 95. Ottawa, ON, Canada: The Queen's Printer.

Simberloff, D, Martin, J.-L., Genovesi, P., Maris, V., Wardle, D.A., Aronson, J., Courchamp, F., Galil, B., Garcia-Berthou, E., Pascal, M. and Pysek, P. (2013). 'Impact of biological invasions: What's what and the way forward'. *Trends in Ecology and Evolution* 28(1): 58–66.

Venter, O., Brodeur, N.N., Nemiroff, L., Belland, B., Dolinsdk, I.J. and Grant, J. (2006). 'Threats to endangered species in Canada'. *BioScience* 56(11): 903–910.

Wilcove, D.S., Rothstein, D., DuBow, J., Phillips, A. and Losos, E. (1998). 'Quantifying threats to imperilled species in the United States'. *BioScience* 48(8): 607–615.

Cooperative natural resource and invasive species management in Hawai'i

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Abstract From the arrival of Polynesians before 1200 AD until Western contact in 1778, Hawaiian land use and resource distribution were centred on the "ahupua'a" system of watershed-based management areas that made their inhabitants nearly self-sufficient. Those units were grouped into larger regions, each called a "moku," led by lower chiefs who were in turn governed by the high chief or king of each island. While societal and environmental taboo or "kapu" were enforced from above, day to day neighbourhood cooperation served to protect resources, produce food, and sustain up to one million people before Western contact. Following the arrival of Europeans, land and resource management, and governance based on native Hawaiian core beliefs, were replaced by a centralised Western market economy. Modern land ownership, agency mandates and legal jurisdictions provide artificial walls that keep people from moving, but do not constrain invasive species, nor are they effective for managing public trust resources such as water or native species. Over time government and conservation organisations have come to view decentralised cooperation as a key to protecting the 50% of Hawaiian terrestrial, native habitat that persists. Current cooperative efforts serve to protect watersheds via watershed partnerships, and to detect and address invasive species through invasive species committees on each island. These and other projects are facilitated by the University of Hawaii's Pacific Cooperative Studies Unit, which plays a unique role in coordinating the use of private and government grant dollars to hire and manage 460 conservation staff in cooperative conservation projects state-wide.

Keywords: ahupua'a, invasive species, cooperative, watershed partnerships, invasive species committees

INTRODUCTION

The state of Hawai'i comprises eight large islands, seven of them inhabited. The island of O'ahu, home to nearly one million of the state's 1.4 million residents, is the seat of the central state government (<http://files.hawaii.gov/dbedt/economic/databook/db2016/db2016.pdf>). Each of the islands has different land use and economic histories. While most islands have a few major land owners, almost all the islands are currently mosaics of federal, state, county, and private properties, making it difficult to mount effective responses to invasive alien species (IAS), as these recognise neither property boundaries nor jurisdictions, or to manage public trust resources such as native species, habitats, and water resources (Ikuma, et al., 2002; Rago & Sugano, 2015). This paper presents case studies that explore how Hawai'i has responded to conservation challenges through cooperative efforts by both institutions and individuals. The list is illustrative, not exhaustive.

Natural history and the Hawaiian period

Hawai'i is one of the most isolated archipelagos on the planet. From still-forming Hawai'i Island, the archipelago progresses in age through the main islands, to the islets and atolls of Papahānaumokuākea National Monument, and the submerged Emperor Seamounts, representing 70 million years of passage over a tectonic hot spot (Heliker, 1989). Before the arrival of humans, new species became established every 175,000–15,000,000 years (Ziegler, 2002). Isolation and subsequent adaptation to a wide variety of ecological zones and habitats over millions of years produced a stunning biodiversity vulnerable to outside perturbations (Carlquist, 1974; Duffy & Vargas, 2017).

Polynesians settled in Hawai'i by 1200 AD (Kirch, 2011; Wilmshurst, et al., 2011). Initial populations were small and the first Hawaiians subsisted as hunter-gathers with limited agriculture. Even these initial actions had a massive effect on biodiversity in a terrestrial ecosystem that had not known mammalian predators. With increasing human populations and the extinction of terrestrial protein sources such as flightless birds and land crabs, agriculture became more important, requiring communal investment in infrastructure such as fish ponds and irrigation systems (Kirch, 1985; Paulay & Starmer, 2011). Land was often divided into mountain-to-sea pie-shaped wedges (*ahupua'a)* with larger units called *moku* on each island. Although trading occurred, ahupua'a tended to be internally balanced systems (Andrade, 2008). *Ahupua'a* were administered by *konohiki*, resource managers appointed by the *ali'i* (rulers) of large districts or entire islands (Gonschor & Beamer, 2014). There was also a division between the realm of man (*wao kanaka*), the agricultural and community areas, and the realm of gods (*wao akua*), the upper forests where entry was granted only to specially trained individuals following strict protocols. While tenure of the *ali'i* was subject to the political winds of fortune, the residents (*maka'ainana*) of the *ahupua'a* were permanent. Together with the *konohiki*, they made decisions about use of local resources ranging from montane forests and irrigated uplands down to coastal ponds and inshore waters and these decisions were regulated by social/religious strictures (*kapu)* (Mueller-Dombois, 2007).

The arrival of humans greatly increased the rate of species' arrivals, either deliberately for food or other economic or cultural advantages, or as accidentals, incidental to travel and commerce. The first settlers traveling east from Polynesia brought about 30 plant species and several animals, including Polynesian pigs (*Sus scrofa*) and Pacific rats (*Rattus exulans*), some perhaps as stowaways.

Despite the capacity of Hawaiian society to mobilise large numbers of people at the island or moku level to engage in major community efforts such as building *heiau* (temples) and fishponds (Kirch, 1985), we have no direct information on how pre-contact Hawaiians reacted to the impacts of invasive species. For example, archaeological evidence suggests that Pacific rats caused major changes in lowland ecosystems by eating tremendous amounts of seeds, damaging or killing plants, and preying on ground-

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nesting birds and other species (Athens, 2009). Kepelino (1932: 86) reported oral traditions that rats were a major problem for sweet potato (*Ipomoea batatas*) crops in the lowlands. The *ali'i* organised rat hunting contests, suggesting opportunity, if not necessity (Athens, 2009; Handy & Handy, 1972).

Post-European arrival

The arrival of Captain Cook in 1778 led to rapid changes, including the introduction of diseases such as smallpox to which Hawaiians had little resistance, and the introduction of Western ideals and religion that muted Hawaiian language, culture, and beliefs (Busnell, 1993). Just seventy years later, the Great Mahele (land division) of 1848 placed two-thirds of the crown lands in private hands, the majority non-Hawaiian, as most Hawaiians could not conceive of a world where they needed to claim rights to the land they had always lived on. This alienation of land further weakened the traditional societal structure and the *kapu* restrictions that controlled use of natural resources (Chinen, 1958; LaCroix & Roumasset, 1990).

Land ownership and political power became concentrated in the hands of the "Big Five" corporations which were primarily involved in an export economy centred on sugar production (Dorrance & Morgan, 2000). These shifts eventually led to the overthrow of the Monarchy (Kame'eleihiwa, 1992) and the resulting "plantation system" came to dominate Hawaii's social, political and economic systems with a top-down political structure centred on the island of O'ahu.

Extensive deforestation occurred as lands were converted to cane fields and streams were diverted for irrigation. The drive to export goods also led to further impacts on forests, with harvesting of sandalwood (*Santalum paniculatum*) and *pulu*, the fibre from native tree ferns (*Cibotium menziesii*), in the *wao akua* areas previously regarded as sacred and off-limits (Cuddihy & Stone, 1990). Land devoted to sugar production peaked in the 1940s and economically viable production ceased by 2015 (Dorrance & Morgan, 2000). The end of sugar as a crop left large portions of lower-elevation landscapes fallow or being converted into housing tracts and tourist developments on the coasts. Sugar has been replaced by tourism and the military as drivers of the economy, but the state retains its O'ahu-centric political orientation left over from plantation days (Kalapa, 1992).

Polynesian pigs were initially barnyard animals, but after 1778 they mixed with introduced European strains and soon found their way into upper-elevation forests. Captains Cook and Vancouver left cows (*Bos taurus*), sheep (*Ovis aries*) and goats (*Capra hircus*) as gifts for Hawaiians (Tomich, 1986). Cows were placed under royal protection by King Kamehameha after their introduction in 1794. Protection lasted until 1830 by which point the population had greatly expanded and caused significant deforestation (Tomich, 1986). Cats (*Felis catus*) were brought to the islands both as novelties and to curb rodent populations (Duffy & Capece, 2012). With the rise of the sugar plantations, species such as cane toads (*Rhinella marina*), mongoose (*Urva auropunctata*) and parasitic wasps were introduced to reduce rats and insects that feed on the cane. While their effectiveness is debatable, their negative consequences are not (Doty, 1945; Peck, et al., 2008). Accidental introductions included black and Norway rats (*R. rattus* and *R. norvegicus*) (Tomich, 1986), and mosquitoes (Culicidae) that were stowaways on ships. Earthworms (Lumbricidae) and ants (Formicidae) were absent from pre-contact Hawai'i, but probably arrived in soil and plants, as did numerous other invertebrates with largely undocumented but likely enormous impacts (Gillespie & Reimer, 1993).

More than 100 plant species arrived in the 60 years following Cook's arrival (Nagata, 1985). More recently Loope & Kraus (2009) reported that, during 1995–2003, 89 species per year became established. To date, over 10,000 plant species have been introduced for cultivation in the islands (Imada, et al., 2005). Birds and mammals were introduced for hunting and for human entertainment (Walker, 1967; Long, 1971). Accidental introduction and deliberate smuggling of herptiles have been a problem, with some becoming invasive (McKeown, 1996; Kraus & Cravalho, 2001; Kraus, 2009). Aquatic species have arrived as deliberate introductions for fisheries, through the aquaculture/pet/aquarium trade, in ballast water and as biofouling (Eldredge & Smith, 2001; Brasher, et al., 2006; Carlton & Eldredge, 2009). Finally, pathogens have been a continuing problem for both humans and the rest of the biota since European contact (e.g. Wilbar, 1947; Warner, 1968; Bushnell, 1993).

Responding to alien invasive species

The Kingdom of Hawai'i enacted the first biosecurity measure for the islands, banning the import of coffee beans to prevent alien disease from affecting the islands' own crops (Holt, 1996). Later, import of sugar cane and other grasses was restricted because they might bring in new diseases and pests of the dominant agricultural crop (Territory of Hawai'i, 1941). By 1975, deliberate introductions of organisms had to be approved by the Department of Agriculture. This rule remains in place; however, the vast majority of plants and plant parts are still not effectively restricted from entry (Loope & Kraus, 2009).

King Kalākaua (Kalākaua 1876) began a programme of fencing to exclude feral ungulates from watersheds to protect the water supply. By the turn of the century and the fall of the monarchy, the territorial legislature recognised the continued impact on watershed forests by feral animals and the unregulated harvest of forest products. This led to a massive re-planting of fast-growing non-native trees with the hope that this would sustain watershed function. However, several of the trees became invasive (<https://www.nature.org/media/hawaii/the-last-stand-hawaiian-forest.pdf>; Cox, 1992; Woodcock, 2003; Kaiser, 2014).

During the territorial period and following statehood, legislation created several state governmental agencies to address alien invasive species (AIS) and to protect or manage natural resources. With various name changes over time, the Department of Health dealt with disease vectors such as rats and mosquitoes, while the Department of Land and Natural Resources dealt with establishing forest reserves, managing aquatic and hunting resources, and reducing the impact of invasive species on state-owned watersheds and native ecosystems. The Department of Agriculture dealt with invasive species of importance to agriculture, and the importation of agricultural goods and species. No agency was or is responsible for a holistic assessment or response to the continued arrival of additional invasive species (Rago & Sugano, 2015).

In the decades since these laws were created, there have been major changes in Hawaii's economic drivers, agricultural crops, frequency and quantities of imports, and the rise of air cargo (2.72%/year from 1990 to 2016: DBEDT, 2017), with a resulting increase in magnitude of risk from invasive species. Unfortunately changes to the laws and policies that reduce or address invasive species risks or impacts have been piecemeal and insufficient, with gaps within or between agency mandates (Miller & Holt, 1992; Ikuma, et al., 2002; Loope & Kraus, 2009; Rago & Sugano, 2015).

At the federal level, the National Park Service and the National Wildlife Refuge System, both under the Department of Interior, have broad mandates to manage invasive species and protect natural resources and habitats within their holdings, but their authority and actions historically have been confined within their property boundaries. The Department of Defense (DOD), another large landowner, did not focus effort or attention on mitigating impacts on the natural environment or protecting natural resources unless they interfered with military activities, as did a dengue outbreak during World War II when martial law allowed the agency to ignore property rights to deal with the outbreak (Wilbar, 1947). More recently, DOD has become more active and pre-emptive, in part because of federal laws such as the U.S. Endangered Species Act of 1973 (ESA) and the National Environmental Policy Act of 1969 (NEPA), where actions that may have an impact on Federally listed endangered species and non-compliance might restrict the military mission.

For prevention of new alien invasive species, among other mandates, the Department of Homeland Security Customs and Border Protection is responsible for regulating the importation of goods and conveyances from foreign sources into the U.S., while the U.S. Department of Agriculture focuses on foreign and domestic agricultural imports that may carry pests and diseases. The U.S. Department of Interior (U.S. Fish and Wildlife Service) uses the Lacey Act and ESA to reduce the risk of invasive species being imported into Hawaii in specific circumstances (e.g. the Injurious Wildlife Provisions of the Lacey Act).

With the rise of rapid world trade, Hawaii's borders have become increasingly permeable to invasive species (Loope & Kraus, 2009). Responses to such species are difficult, as potentially invasive species are rarely discovered on a single property where the landowner has the knowledge, skills, interest, and funding to address the species before it spreads. Landowners can also be hesitant to allow government officials onto their properties to search for or control invasive alien species, so government often fails to detect invasive alien species before they spread (Kraus & Duffy, 2010). Action against a new invasive alien species largely depends on its location and whether the species is perceived as falling within the mandate of a particular agency. Further, the bureaucratic process for the addition of new species to official lists mandating control does not keep up with the pace of arrivals (Penniman, et al., 2011). Finally, cooperation between state, federal and county authorities has at times been limited and intermittent (Warren, 2006). In consequence in the last two decades, cooperative, often informal approaches have increasingly supplemented top-down formal efforts. We present four such cooperative models that range from the intergovernmental, to agreements between landowners, to groups open to anyone sharing a common objective.

Pacific Cooperative Studies Unit (PCSU)

One of the earliest natural resource management organisations in Hawaii to extend beyond top-down management was the Cooperative National Parks Studies Unit (CPSU), which formed in 1973 through an agreement between the National Park Service (NPS) and the University of Hawaii. CPSU initially provided collaborative research and technical support for Hawai'i Volcanoes and Haleakala National Parks. Following the passage of three key federal laws: the National Environmental Policy Act, the Endangered Species Act, and the General Authorities Act of 1970 (<https://www.nps.gov/parkhistory/hisnps/NPSHistory/timeline_annotated.htm>), NPS lacked the internal capacity to conduct the research necessary to respond to these mandates. It also lacked a mandate to protect national parks before threats actually reached them, making parks legal but not ecologically sustainable islands. Following initial surveys documenting the native flora and fauna and threats to these species, including from non-native species, UH scientists built a small test ungulate exclosure, which produced rapid recovery of native plants. NPS engaged the CPSU to build more fences, removing ungulates, and monitoring the subsequent recovery. Based in part on this work, fencing as a management tool was adopted by NPS, other federal and state agencies, and non-profit organisations. In response to increasing recognition of threats to Hawaii's biodiversity and perceived gaps caused by narrow agency mandates and jurisdictions, CPSU morphed into what is now the Pacific Cooperative Studies Unit (PCSU) working with a range of state and federal agencies, as well as non-profits and private companies and individuals. PCSU provides research, resource management and outreach expertise via collaborative projects, while also increasing employment opportunities in conservation. Over the last 20 years it has grown from 150 employees to more than 450 (Fig. 1), mentoring and staffing a range of organisations dealing with invasive and endangered species (see below).

Coordinating Group on Alien Pest Species (CGAPS)

A second key development was the formation of the Honolulu-based interagency Coordinating Group on Alien Pest Species (CGAPS) in 1995. In 1992, The Nature Conservancy of Hawaii (TNCH) and the Natural Resources Defence Council published a report on Hawaii's biosecurity measures and the gaps that would likely lead to the arrival and establishment of major new pests such as brown tree snakes (*Boiga irregularis*) and red fire ants (*Solenopsis invicta*) (Miller & Holt, 1992). The report concluded that, although there were funding and policy gaps, the most serious problem was a lack of interagency, and sometimes intra-agency, communication and cooperation, and that many such gaps could be addressed through a coordinated effort (Miller & Holt, 1992; Holt, 1996). These reports, and other events, led to the crafting of the Hawaii Alien Species Action Plan in 1993–94 with the help of more than 80 agency and NGO leaders under a steering committee that morphed into CGAPS (Nakatani & Wilson, 1995).

Today, CGAPS continues to facilitate interagency and NGO communication and cooperation through quarterly meetings, and its steering committee plans and conducts collaborative projects to catalyse action on invasive species. CGAPS was originally administered by the Hawaii Department of Agriculture with staff time contributed by TNCH and agencies, but it now has a rotating chair structure, and staff and logistics supported by PCSU.

One of its recent projects was the crafting of a Plant Health Emergency Response Plan, which laid out how the US Department of Agriculture and Hawai'i Department of Agriculture could engage other federal, state and county

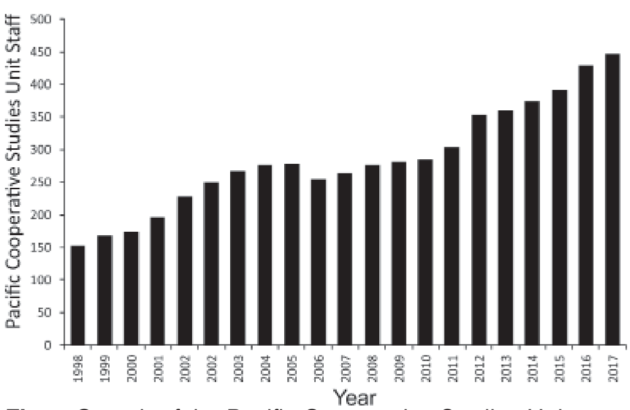

Fig. 1 Growth of the Pacific Cooperative Studies Unit.

agencies, and non-governmental organisations if a serious plant pest were to arrive, requiring a response beyond what the two federal agencies could provide (Loope & Shluker Ryon, 2013). To test the plan, CGAPS conducted a discussion-based "tabletop" exercise in November 2013, using the then-fictitious discovery of the coconut rhinoceros beetle (*Oryctes rhinoceros*, CRB). The tabletop discussion included more than 40 participants and resulted in a report that outlined legal and procedural questions that arose (Coordinating Group on Alien Species, 2013). Coincidentally, a month later, in December 2013, CRB were detected in a trap at Joint Base Pearl Harbor-Hickam, triggering the first use of the Plant Health Emergency Response Plan. Although eradication has yet to be achieved, the emergency response has been successful in many ways, from the initial staffing of the response co-led by the two responsible agencies, and supplemented by multiple federal, state, and non-government partners, to the containment of the beetles to West O‘ahu.

Invasive Species Committees

Invasive Species Committees (ISCs) are voluntary partnerships on each island that address incipient (new) invasive plants and animals on an island-wide basis. The five invasive species committees (Kaua‘i Invasive Species Committee, O‘ahu Invasive Species Committee, Maui Invasive Species Committee, Moloka‘i Invasive Species Committee, and Big Island Invasive Species Committee), represent perhaps the best collaborative efforts in the islands. Their steering committees are essentially self-recruited, made up of interested private individuals and groups as well as representatives of county, state and federal agencies. Together, each island's steering committee provides strategic direction to a paid staff and field crew for island-wide work on early detection and control or eradication of high-risk invasive species. A critical function of ISCs is to obtain right of entry to private lands through education and negotiation. The logistic, fiscal and staffing aspects of each ISC are handled by the Pacific Cooperative Studies Unit.

The first committee sprang from a pioneer effort on Maui Island. A melastome tree (*Miconia calvescens)* had been identified on Tahiti as a major threat to intact native forests (Meyer, 1996). Biologists returning from a visit to Tahiti recognised that the species occurred on Maui and might represent a similar local threat to Hawai‘i (Gagné, et al., 1992). This led to the formation of an *ad hoc* Melastome Action Committee in 1991 to address the problem on Maui (Conant, et al., 1997; Medeiros, et al., 1997). The effort subsequently expanded to Big Island (Tavares, 1998). In recognition that there might be additional IAS threats (Miller & Holt, 1992), the MAC expanded to other species, and in 1997 the Maui Invasive Species Committee was formed, soon followed by O‘ahu, Kaua‘i, Big Island and Moloka‘i committees (Martin, 2003).

The ISCs focus on early detection and rapid response leading to eradication of incipient invasive species, and they also conduct outreach and education to help the public reduce the impacts of established species. All species are chosen for their local, not state, importance, based on evaluation criteria that include risk to an island's economy or ecology, and feasibility of control or eradication (Penniman, et al., 2011). The number of staff in each ISC varies, but generally each has a field team, an outreach specialist, a GIS/data specialist, and an overall manager who is responsible for government relations, obtaining funding, and working with its steering committee and PCSU (Krauss & Duffy, 2010). Since formation, the ISC managers have worked together to develop standard methods, coordinate funding and reports, and even share field crews when advantageous. During the 2008 recession, they redistributed funding to keep all the ISCs staffed and active. Being local to each island, these committees enjoy strong county and legislative support, but funding remains a persistent problem as new invasive species continue to arrive while many of the old ones persist. As of 2010, 27 populations of emerging invasives had been removed by the ISCs, but efforts for others are likely to be drawn out because of reinvasions, persistent seed-banks, or the continued discovery of isolated individuals (Kraus & Duffy, 2010; Penniman, et al., 2011). In addition, the ISCs have worked with the Hawaii Ant Lab (another project of PCSU) and HDOA to survey for and control incipient populations of species such as little fire ants (*Wasmannia auropunctata*) and coqui frogs (*Eleutherodactylus coqui*) on islands where they are not yet established. These early detection and rapid response functions have resulted in dozens of local eradications of these pests before they could establish populations.

Watershed Partnerships

Isolated oceanic islands like the Hawaiian archipelago have limited freshwater supplies. Native forests in Hawai‘i retain water better than do island forests dominated by introduced species (Giambelluca, et al., 2009; Kagawa, et al., 2009; Cavaleri, et al., 2014), so protection of watersheds is a prudent investment toward the persistence of human populations in the islands, as well as for the maintenance of the archipelago's unique biota.

Maui has been the incubator of a number of innovations in Hawaii and so it is not surprising that the first joint effort to manage and improve watersheds across ownerships, the East Maui Watershed Partnership (EMWP), was established in 1991 through the efforts of The Nature Conservancy of Hawai‘i and the state Department of Land and Natural Resources (Loope & Reeser, 2001). There are now ten watershed partnerships or associations on five islands, covering 2.2 million acres and involving 75 land-owning partners, ranging from state and federal agencies to NGOs to private companies and individuals (<http://hawp.org/partnerships/>) (Fig. 2). These partners may have differing objectives in land management so watershed partnerships focus on those they hold in common, rather than imposing the agendas of a minority of partners (cf. Ostrom, 1990).

Like CGAPS and the ISCs, most of the watershed partnerships are informal, with the landowners and agencies functioning as steering committees to determine objectives, and with a manager and staff to address the objectives. PCSU also provides the structure and administrative capacity for most of these partnerships.

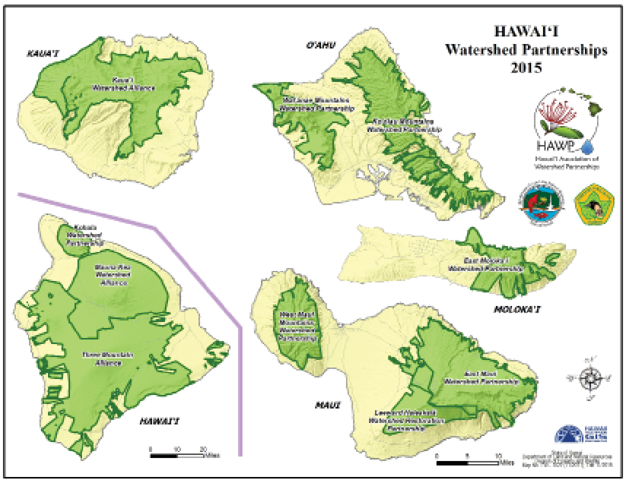

Fig. 2 Extent of watershed partnerships in the Hawaiian archipelago in 2015 (<http://hawp.org/wp-content/uploads/2011/12/WP_2015.png>).

The main task of the partnerships is the long-term protection of forests in watershed recharge areas, by removing alien plants and animals directly or through the installation and upkeep of hundreds of miles of protective fencing to exclude ungulates from sensitive areas. Watershed partnerships can do this at the landscape level without the constraints of political or property boundaries. Indirectly, by protecting native habitat across wide areas, the watershed partnerships have also become critical for the conservation of endangered species. Working with the ISCs, the partnerships are also key to locating and dispatching newly arrived potentially invasive species.

DISCUSSION

Before contact with Western culture and during the monarchy, environmental rules in Hawai'i were mandated at the island or *moku* level. The introduction of Western concepts such as private property and land ownership, coupled with the abolition of the *kapu* system and transition of the monarchs to Christianity, resulted in major changes in the management of natural resources. Major increases in resource extraction and land clearance were permitted and abetted by the republic and territorial governments controlled by the sugar companies. After statehood, strong federal laws dealing with pollution and wetlands functioned through command and control enforcement. Most recently there has been a recognition that such top-down approaches are less effective for non-point problems such as pollution, habitat destruction and managing endangered and invasive species outside government lands (Lubell, et al., 2002). Wider involvement is needed to ensure buy-in by "stake holders" who must be part of solutions (John, 1994). Some of these more participatory approaches are mandated by U.S. law, such as interagency consultation over endangered species, habitat conservation plans for endangered species, and public comments on federal government actions under the National Environmental Policy Act. Within government, *ad hoc* cooperation between agency partners through entities such as CGAPS and PCSU has often proven more nimble than statutory constructs. Outside formal government, limited-access partnerships of landowners such as watersheds partnerships and invasive species committees open to all have proven highly responsive to local conditions.

Overall, partnerships can yield multiple advantages. They can help bring together resources such as outside expertise (e.g. CPSU/PCSU) or undertake landscape-scale management by pooling resources and reducing artificial boundaries (WPs and ISCs), or they can see "the big picture" with collaborators working together to identify and address issues (CGAPS).

Partnerships are not automatic panaceas. Partnerships require flexibility and trust, and a recognition that they may not be appropriate for every problem. Partnerships also require a roughly equal distribution of power and resources. If one partner is dominant, then the partnership becomes merely an advisory group or rubber stamp. Partnerships require a working consensus on approaches. Islands have a limited spectrum of economic activity compared to the mainland so people are more likely to share a common perspective and recognition of the value of the local indigenous environment than may occur at the continental scale. However, even in Hawai'i, issues such as air-dropped rodent control agents, genetically-modified organisms, and biocontrol may remain too controversial for partnership approaches. Feral cat management on the islands of Oahu and Kauai is a particularly contentious issue, but there is hope for an emerging consensus. Twenty years ago, fencing was similarly controversial but it has now become an accepted approach to land protection.

In terms of logistics, partnerships can falter without a lead person and staff whose jobs are to move the partnership's goals forward. Partnerships appear to be less effective when they are burdened with managing such staff, as administrative concerns divert time and energy away from the "big picture" and away from consensus building around common objectives (Lubell et al., 2002). In Hawai'i, the Pacific Cooperative Studies Unit has frequently supplied the stable logistics and organisational underpinning for such partnerships, providing professional staffing, and the ability to handle financial, legal and regulatory requirements. This appears to provide a flexibility not always present in government agencies where funding can vary from year to year and priorities change from one political administration to the next.

Although partnerships and cooperative efforts can be powerful tools for conservation and we can generate general rules about what works and what doesn't, all such efforts are local, dependent on the local economy, local politics and the local environment. It is important that we better document what has worked and what hasn't for Hawaii, both for other areas that might wish to explore the use of cooperation in conservation, and as anthropogenic climate change brings new challenges to islands.

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REFERENCES

Andrade, C. (2008). *Haena: Through the Eyes of the Ancestors.* Honolulu: University of Hawai'i Press.

Athens, J.S. (2009). '*Rattus exulans* and the catastrophic disappearance of Hawai'i's native lowland forest'. *Biological Invasions* 11: 1489–1581.

Brasher, A.M., Luton, C.D., Goodbred, S.L. and Wolff, R.H. (2006). 'Invasion patterns along elevation and urbanization gradients in Hawaiian streams'. *Transactions of the American Fisheries Society* 135: 1109–1129.

Bushnell, O.A. (1993). *The Gifts of Civilization: Germs and Genocide in Hawaii.* Honolulu: University of Hawaii Press.

Carlton, J.T. and Eldredge, L.G. (2009). *Marine Bioinvasions of Hawaii.* Bishop Museum Bulletins in Cultural and Environmental Studies. Honolulu: Bishop Museum Press.

Cavaleri, M.A., Ostertag, R., Cordell, S. and Sack, L. (2014). 'Native trees show conservative water use relative to invasive trees: results from a removal experiment in a Hawaiian wet forest'. *Conservation Physiology* 2: doi:10.1093/conphys/cou016.

Chinen, J.J. (1958). *The Great Mahele: Hawaii's Land Division of 1848.* Honolulu: University of Hawaii Press.

Conant, P., Medeiros, A.C. and Loope, L.L. (1997). 'A Multi-agency Containment Program for Miconia (*Miconia calvescens*), an Invasive Tree in Hawaiian Rain Forests'. In J.O. Luken and J.W. Thieret (eds.) *Assessment and Management of Invasive Plants,* pp. 249–254. Berlin: Springer-Verlag.

Cox, T. (1992). 'The Birth of Hawaiian Forestry: The Web of Influences'. In: J. Dargavel and R. Tucker (eds.) *Changing Pacific Forests: Historical Perspectives on the Forest Economy of the Pacific Basin.* pp. 116–25. Durham, NC: Forest History Society.

Coordinating Group on Alien Species (2013). *After Action Report/ Improvement Plan.* <http://www.cgaps.org/wp-content/uploads/AAR-IP-CGAPS-FINAL-110813.pdf>.

Cuddihy, L.W. and Stone, C.P. (1990). *Alteration of Native Hawaiian Vegetation – Effects of Humans, their Activities and Introductions*. Honolulu: Cooperative National Park Resources Studies Unit, University of Hawaii at Manoa.

DBEDT (Department of Business, Economic Development & Tourism). (2017). *Air Cargo in Hawaii's Economy: Developments in Hawaii's Air Cargo Industry 1990–2016.* Honolulu: Department of Business, Economic Development & Tourism.

Dorrance, W.H. and Morgan, F. (2000). *Sugar Islands: The 165-Year Story of Sugar in Hawai'i,* Honolulu: Mutual Publishing.

Doty, R.E. (1945). 'Rat control on Hawaiian sugar cane plantations'. *Hawaiian Planter's Record* 49: 71–239.

Duffy, D.C and Capece, P. (2012). 'Biology and impacts of Pacific island invasive species. 7. The domestic cat (*Felis catus*)'. *Pacific Science* 66: 173–212.

Duffy, D.C. and Vargas, H. (2017). 'From the vagile to the sedentary: disease implications and new host relationships on islands'. Chapter 5. In: P.G. Parker (ed.) *Disease Ecology. Galapagos Birds and their Parasites*, pp. 113–138. New York: Springer.

Eldredge, L.G and Smith, C.M. (2001). *Guidebook of Introduced Marine Species in Hawaii.* Bishop Museum Technical Report 21. Honolulu: Bishop Museum.

Gagné, B.H., Loope, L.L. Medeiros, A.C. and Anderson, S.J. (1992). '*Miconia calvescens*: A threat to native forests in the Hawaiian Islands' (Abstract). *Pacific Science* 46: 390–391.

Giambelluca, T.W., Martin, R.E., Asner, G.P., Huang, M., Mudd, R.G., Nullet, M.A., DeLay, J.K and Foote, D. (2009). 'Evapotranspiration and energy balance of native wet montane cloud forest in Hawai'i'. *Agricultural and Forest Meteorology* 149: 230–243.

Gillespie, R.G. and Reimer, N.J. (1993). 'The effect of alien predatory ants (Hymenoptera: Formicidae) on Hawaiian endemic spiders (Araneae: Tetragnathidae)'. *Pacific Science* 47: 21–33.

Gonschor, L. and Beamer, K. (2014). 'Toward an inventory of ahupua'a in the Hawaiian kingdom: A survey of nineteenth- and early twentieth-century cartographic and archival records of the island of Hawai'i'. *Hawaiian Journal of History* 48: 53–87.

Handy, E.S.C. and Handy, E.G. (1972). 'Native planters in old Hawaii: Their life, lore and environment'. *Bernice P. Bishop Museum Bulletin* 233: 1–641.

Heliker, C. (1989). 'Volcanic origins of the Hawaiian Islands.' In: C.P. *Stone* and D.B. *Stone* (eds.) *Conservation Biology in Hawaii.* pp. 11–15. Honolulu: University of Hawaii Press.

Holt, A. (1996). 'An Alliance of Biodiversity, Agriculture, Health, and Business Interests for Improved Alien Species Management in Hawaii.' In: O.T. Sandlund, P.J. Schei, and A. Viken (eds.) *Proceedings of the Norway/UN Conference on Alien Species,* pp. 155–160. Trondheim: Directorate for Nature Management and Norwegian Institute for Nature Research.

Ikuma, E.K., Sugano, D. and Mardfin, J.K. (2002) *Filling the Gaps in the Fight Against Invasive Species.* Honolulu: Legislative Reference Bureau. <http://lrbhawaii.org/reports/legrpts/lrb/rpts02/gaps.pdf>.

Imada, C.T., Staples, G.W. and Herbst, D.R. (2005). *Annotated Checklist of Cultivated Plants of Hawai'i.* <http://www2.bishopmuseum.org/HBS/botany/cultivatedplants/>.

John, D. (1994). *Civic Environmentalism*. Washington, D.C.: Congressional Quarterly Press.

Kagawa, A, Sack, L., Duarte, K. and James, S. (2009). 'Hawaiian native forest conserves water relative to timber plantation: Species and stand traits influence water use'. Ecological Applications 19: 1429–1443.

Kaiser, B.A. (2014). 'Watershed conservation in the long run'. *Ecosystems* 17: 698–719.

Kalākaua, D. (1876). *An Act for the Protection and Preservation of Woods and Forests*. Honolulu: The Hawaiian Kingdom.

Kalapa, L.L. (1992). 'Why is it that the State Seems to "Bully" the Counties?' In: R.W. Ross (ed.) *The Price of Paradise.* Honolulu: Mutual Publishing.

Kame'eleihiwa, L. (1992). *Native Land and Foreign Desires.* Honolulu: Bishop Museum Press.

Kepelino, K. (1932). M.W. Beckwith (ed.) *Kepelino's Traditions of Hawaii*. *Bishop Museum Bulletin 95*. Honolulu: Bishop Museum Press.

Kirch, P.V. (1985). *Feathered Gods and Fishhooks: An Introduction to Hawaiian Archaeology and Prehistory*. Honolulu: University of Hawaii Press.

Kirch, P.V. (2011). 'When did the Polynesians settle Hawai'i? A review of 150 years of scholarly inquiry and a tentative answer'. *Hawaiian Archaeology* 12: 3–26.

Kraus, F. and Cravalho, D. (2001). 'The risk to Hawai'i from snakes'. *Pacific Science* 55: 409–417.

Kraus, F. (2009). *Alien Reptiles and Amphibians: A Scientific Compendium and Analysis. Invading Nature – Springer Series in Invasion Ecology, Vol 4.* Dordrecht, Netherlands: Springer.

Kraus, F. and Duffy, D.C. (2010). 'A successful model from Hawaii for rapid response to invasive species'. *Journal for Nature Conservation* 18: 135–141

LaCroix, S.J. and Roumasset, J. (1990). 'The evolution of private property in nineteenth-century Hawaii'. *Journal of Economic History* 50: 829–852.

Long, J.L. (1971). *Introduced Birds of the World: The Worldwide History, Distribution, and Influence of Birds Introduced to New Environments*. New York: Universe Books.

Loope, L. and Reeser, D. (2001). 'Crossing Boundaries at Haleakala: Addressing Invasive Species through Partnerships'. In: D. Harmon (ed.) *Crossing Boundaries in Park Management: Proceedings of the 11th Conference on Research and Resource Management in Parks and Public Lands,* pp. 29-34. Hancock, Michigan: George Wright Society.

Loope, L. and Kraus, F. (2009). 'Preventing Establishment and Spread of Invasive Species in Hawaii: Current Status and Needs for the Future'. In: T.K. Pratt, C.T. Atkinson, P.C. Banko, J.D. Jacobi and B.L. Woodworth (eds.) *Conservation Biology of Hawaiian Forest Birds: Implications for Island Birds*, pp. 359–380. New Haven: Yale University Press.

Loope, L. and Shluker Ryon, A.D. (2013). *State of Hawai'i Collaborative Plant Health Emergency Management Plan (PHERP) and Reference Manual.* <http://www.cgaps.org/wp-content/uploads/Hawaii-Plant-Health-Emergency-Response-Plan-Final-12-30-2013.pdf>.

Lubell, M., Schneider, M, Scholz, J.T. and Mete, M. (2002). 'Watershed partnerships and the emergence of collective action institutions'. *American Journal of Political Science* 46: 148–163.

Martin, C. (2003). *Island-based Partnerships and Statewide Coordination to Protect Hawaii.* <http://www.cgaps.org/wp-content/uploads/2002-Island-Based-and-Statewide-Report.pdf>.

McKeown, S. (1996). *A Field Guide to Reptiles and Amphibians in the Hawaiian Islands*, Los Osos, California: Diamond Head Publishing.

Medeiros, A.C., Loope, L.L., Conant, P. and McElvaney, S. (1997). 'Status, Ecology, and Management of the Invasive Tree *Miconia calvescens* DC (Melastomataceae) in the Hawaiian Islands'. In: N.L. Evenhuis and S.E. Miller (eds.) *Records of the Hawaii Biological Survey for 1996. Bishop Museum Occasional Papers* No. 48, pp. 23–36. Honolulu: Bernice Pahuahi Bishop Museum.

Meyer, J.-Y. (1996). 'Status of *Miconia calvescens* (Melastomataceae), a dominant invasive tree in the Society Islands (French Polynesia)'. *Pacific Science* 50: 66–76.

Miller, S. and Holt, A. (1992), *The Alien Pest Species Invasion in Hawaii: Background Study and Recommendations for Interagency Planning*. Honolulu: The Nature Conservancy of Hawaii and the Natural Resources Defense Council.

Mueller-Dombois, D. (2007). 'The Hawaiian *Ahupua'a* Land Use System: Its Biological Resource Zones and the Challenge for Silvicultural Restoration'. In: N.L. Evenhuis and J.M. Fitzsimons (eds.) *Biology of Hawaiian Streams and Estuaries. Bishop Museum Bulletin in Cultural and Environmental Studies* 3, pp. 23–33. Honolulu: Bernice Pahuahi Bishop Museum.

Nagata, K.M. (1985). '*Early plant introductions* in Hawaii'. *Hawaiian Journal of History* 19: 35–61.

Nakatani, J. and Wilson, M. (1995). *Hawaii Alien Species Action Plan*. <http://www.cgaps.org/wp-content/uploads/CGAPS-formation-to-implement-Hawaii-ASAP-1995.pdf>.

Ostrom, E. (1990). *The Logic of Collective Action.* New York: Cambridge University Press.

Paulay, G. and Starmer, J. (2011) 'Evolution, insular restriction, and extinction of oceanic land crabs, exemplified by the loss of an endemic *Geograpsus* in the Hawaiian islands'. *PLOS ONE* 6(5): e19916.

Peck, R.W., Banko, P.C., Schwarzfeld, M., Euaparadorn, M. and Brinck, K.W. (2008). 'Alien dominance of the parasitoid wasp community along an elevation gradient on Hawai'i Island'. *Biological Invasions* 10: 1441–1455.

Penniman, T.M., Buchanan, L. and Loope, L.L. (2011). 'Recent plant eradications on the islands of Maui County, Hawai'i'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 325–331. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Rago, F. and Sugano, D. (2015). *Can't see the forest for the (albizia) trees: an invasive species update*. Honolulu: Legislative Reference Bureau.

Stone, C.P. and *Stone, D.B.* (eds.) (1989). *Conservation Biology in Hawaii.* Honolulu: University of *Hawaii* Press.

Tavares, K. (1998). 'Big Island Melastome Action Committee: *Miconia calvescens* control program overview'. *Proceedings of the First Regional Conference on* Miconia *Control (August 26–29, 1997)*: 52–64.

Territory of Hawai'i (1941). *Concerning the Introduction of Sugar Cane, Bamboo and Other Grasses or Parts Thereof into Hawaii from Any Other Part of the United States. Effective 12/12/41*. Honolulu: Division of Entomology, Board of Commissioners of Agriculture and Forestry.

Tomich, P.Q. (1986). *Mammals in Hawai'i.* Honolulu: Bishop Museum Press.

Walker, R.L. (*1967*). 'A brief history of exotic game bird and *mammal* introductions into *Hawaii* with a look to the future'. '*Elepaio* 28: 39–43.

Warner, R.E. (1968). 'The role of introduced diseases in the extinction of the endemic Hawaiian avifauna'. *Condor* 70: 101–120.

Warren, P. (2006). *Biosecurity Systems of Hawai'i.: An Evaluation and Recommendations for Reform.* Honolulu: Hawai'i Conservation Alliance.

Wilbar, C.L. (1947). 'Control of dengue in Hawaii'. *American Journal of Public Health* 37: 663–674.

Woodcock, D. (2003). 'To restore the watersheds: Early twentieth century tree planting in Hawaii'. *Annals of the Association of American Geographers* 93: 624–635.

Ziegler, A.C. (2002). *Hawaiian Natural History, Ecology, and Evolution.* Honolulu: University of Hawaii Press.

Invasive species management in Mauritius: from the reactive to the proactive – the National Invasive Species Management Strategy and its implementation

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Abstract This account provides the context behind the need to implement an integrated, cross-sectoral approach to Invasive Alien Species (IAS) management in the Republic of Mauritius (RoM). The challenge of increased travel, tourism and transport is enumerated and the history of IAS entry, establishment and spread in the RoM before and since the formulation of the National Invasive Alien Species Strategy and Action Plan for the Republic of Mauritius (NIASSAP) (2010-2019) is reviewed to judge the effectiveness of biosecurity measures at the national and sub-national level. New incursions appear to have increased since 2010. Examples include, the papaya mealybug (*Paracoccus marginatus*; 2013), the Oriental fruit fly (*Bactrocera dorsalis*), a species that had previously been eradicated from the island (2013 and 2015) and foot and mouth disease (FMD) (*Aphthae epizooticae*; 2016). There have been some effective responses. A biological control agent was released against the papaya mealybug and fruit production has recovered, FMD has been eradicated and a campaign for eradication of *B. dorsalis* is underway. However, management approaches remain reactive and sectorally-driven with little cross-fertilisation of ideas and approaches. Biological control, for example, has been very actively pursued in the agricultural sector but has not been officially undertaken for environmental weeds since 1982. The documented incursions represent biosecurity failures that the NIASSAP was designed to address but it has yet to be systematically implemented. The growing impact of new biological invaders on all sectors of the Mauritian economy has stimulated a revival of interest in biosecurity at the governmental level and in 2016 the Government submitted a US$17M UNDP/GEF project: *Mainstreaming IAS Prevention, Control and Management*, which will provide the incremental cost to review, update and effectively initiate the implementation of the NIASSAP.

Keywords: early detection and rapid response, mainstreaming, management, NISSAP, pathways, prevention, Republic of Mauritius, risk analysis

INTRODUCTION

The expanding IAS threat in the Republic of Mauritius and the adoption of the NIASSAP

The Republic of Mauritius (RoM) comprises the main island of Mauritius and Rodrigues, about 560 km to the east of Mauritius, their associated islets, and the outer islands of Agalega, Tromelin, Cargados Carajos (St Brandon) and the Chagos Archipelago. Mauritius and Rodrigues form part of the Mascarene Islands chain located in the Western Indian Ocean. The Mascarenes belong to one of the 25 internationally recognised biodiversity 'hotspots' (Myers, et al., 2000). Tropical climate, diverse topography and over a million of years of isolation have resulted in the evolution of a diverse biota with a high degree of endemism. Invasive alien species (IAS) constitute a major threat to the remaining biodiversity in the RoM (Florens, 2013; Virah-Sawmy, et al., 2009; Cheke & Hume, 2008). IAS also have serious economic and health impacts, especially if the definition of IAS is broadened to include agricultural pests and zoonotic diseases. This broad conception of IAS was used when developing the country's National Invasive Species Strategy and Action Plan 2010–2019 (NIASSAP) (RoM, 2010), officially adopted by Cabinet in 2010. The NIASSAP is based on the premise that the problems of biological invasions are cross-sectoral in nature, so there is a need for a harmonised approach to biosecurity that cuts across traditional sectoral boundaries. Making use of the 'biosecurity umbrella' will help to ensure that all activities relating to species introductions and spread are based upon a coordinated and science-based approach that is underpinned by the assessment and management of risk. This paper describes some of the RoM's IAS invasion trends, its expanding and diversifying IAS pathways, and examples of IAS management successes and challenges ('the reactive') as a backdrop to the development of the NIASSAP ('the proactive'), its implementation to date and future prospects.

A brief history of alien species establishment in Mauritius

Vertebrate establishment

Human actions resulted in the introduction of vertebrates to Mauritius even before the first documented landing on the island, by the Dutch in 1598 (Cheke, 1987). Black rats (*Rattus rattus*) probably established themselves on Mauritius via shipwrecks and may have been responsible for the extinction of many endemic animal species even before colonisation. Between the first Dutch landing and settlement in 1638 two major animal invaders, the Javanese macaque (*Macaca fascicularis*) and the feral pig (*Sus scrofa*) became established in Mauritius. During the Dutch period (1638–1710), major introductions included Javan deer (*Cervus javanicus*) and cats (*Felis catus*) which became feral.

During French rule (1721–1810), introductions with significant negative economic and environmental effects included the brown rat (*Rattus norvegicus*), the Asian house shrew (*Suncus murinus*) and the tenrec (*Tenrec ecaudatus*).

The steady rate of vertebrate introductions continued under the British (1810–1968) with introductions including the Indian wolf snake (*Lycodon capucinus*), the red-whiskered bulbul (*Pycnonotus jocosus*), African landsnails (*Achatina* spp.) and the small Indian mongoose (*Urva auropunctata*).

Significant vertebrate deliberate and accidental introductions since independence include the Madagascar giant day gecko (*Phelsuma grandis*), the gold-dust day

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gecko (*Phelsuma laticauda*) and the red-eared slider (*Trachemys scripta elegans*). All of these introductions are believed to be due to the pet trade. It would appear that the numbers of vertebrates establishing in the wild in Mauritius is showing no signs of a levelling off (Fig. 1).

Also of concern is the spread of vertebrates and all other taxa between the islands and islets that make up the Republic of Mauritius. Rodrigues Island, the outer islands and Mauritian and Rodriguan islets harbour only a sub-set of the invasive vertebrates found on Mauritius Island. This has conservation implications. For example: carnivorous mammals have never established on Round Island thus saving several endemic reptile species from extinction (Bullock, 1986); Flat Island was home to 80% of the world's population of Bojer's skink (*Gongylomorphus bojerii*) until 2010 when shrews were accidentally introduced (possibly in building materials) from the Mauritian mainland causing their local extinction (Cole & Payne, 2015); and Rodrigues does not have Javanese macaques which, if introduced, would further threaten their already fragile native biodiversity. These examples illustrate the importance of effective inter-island pathway biosecurity.

Plant establishment

Since colonisation, more than 1,600 plant species have been introduced to Mauritius. Many of these introductions have been desirable and others have been essential as Mauritius only has one native plant species, the hurricane palm (*Dictyosperma album*) that has so far been exploited on a commercial scale. Heeroo (2000) assembled all introduced plants records from the Mauritius Herbarium between 1888 and 2000 and found that 804 of the 1,619 species were classified as 'weedy species', 141 being 'agricultural weeds' and 674 being 'naturalised weeds' (Fig. 2). It should be noted that there can be a turnover of weedy species so, assuming that the records are comprehensive, the cumulative number listed is likely to be higher than the actual numbers of weedy species in the field. Herbarium records can only approximate the rate at which species establish themselves as they are heavily dependent on collection effort but it would appear that new naturalisations levelled off between the 1980s and 2000. Data from 2000 onwards need to be consolidated to clarify recent trends.

Of the naturalised species, about 30 currently dominate the country's natural vegetation in terms of numbers of individuals and biomass. Some of the principal invasive woody and shrubby plants in Mauritius and Rodrigues include *Psidium cattleianum* (Chinese guava) which constitutes the vast majority of the biomass in much of Mauritius' humid forest (Florens, et al., 2016), *Ravenala madagascariensis* (ravenale) which forms monotypic stands in similar climatic zones (Baret, et al., 2013), *Hiptage benghalensis* (liane cerf) a woody climber which is increasing in abundance in less humid forests (C. Griffiths pers. comm. 2015) and *Syzigium jambos* (jamrosa) which dominates many riverine landscapes in Mauritius and is one of the most widespread plant invaders in Rodrigues.

Entry establishment and spread of additional species can exacerbate an already bad situation. For example, species belonging to the genus *Prosopis*, a known invasive group (Richardson, 1998) have been planted for erosion control on dry mountain slopes and a proposal for the plantation of up to 3,200 ha of *Arundo donax* (giant reed) is being considered despite its known invasiveness (Csurhes, 2009).

Insect plant pest establishment

Williams & Ganeshan (2001) documented the acceleration in insect pest establishment in Mauritius from the 1970s. Data from the Entomology Division of the Ministry of Agro Industry and Food Security (MAIFS) indicates that this rate has continued, averaging about one new pest record per year (Fig. 3). Recent insect pest introductions include the papaya mealybug (*Paracoccus marginatus*), detected in 2013, and the yellow sugar cane aphid (*Sipha flava*), detected in 2015. These newly established pests represent a well-documented burden on the country's agricultural sector which has become extremely reliant on the use of synthetic pesticides with all their concomitant drawbacks (Abeeluck, et al., 2009). The impacts of newly-established pests on native biodiversity have not been studied.

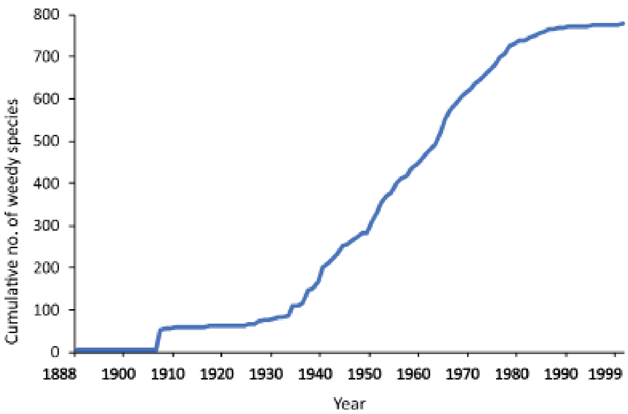

Fig. 2 Cumulative records of weedy species from Mauritius herbarium records 1888–1999. Source: Heeroo 2000.

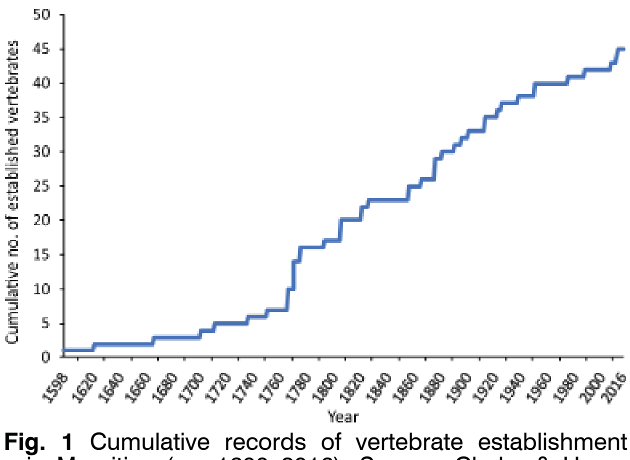

Fig. 1 Cumulative records of vertebrate establishment in Mauritius (pre-1600–2016). Source: Cheke & Hume 2008; Nik Cole (pers. obs.).

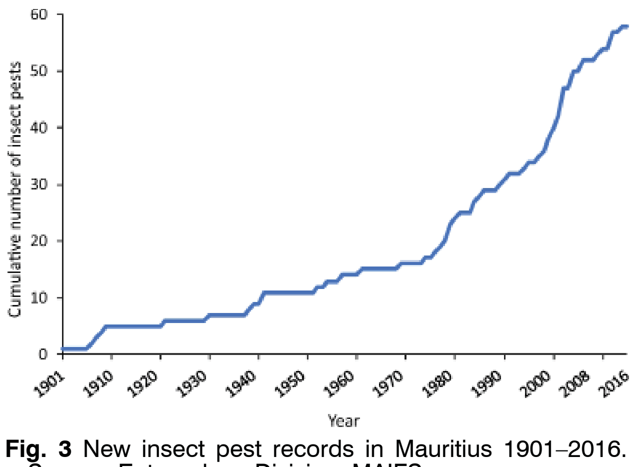

Fig. 3 New insect pest records in Mauritius 1901–2016. Source: Entomology Division, MAIFS.

An example of a repeated insect pest incursion is that of the oriental fruit fly (*Bactrocera dorsalis*): Native to Asia, *B. dorsalis* is one of the world's most destructive pests of fruit with over 300 host species. *Bactrocera dorsalis* is now found in at least 65 countries and continues to spread via infested fruit, either as cargo or carried illegally by airline passengers (CABI, 2017). It was first detected in Mauritius in 1996 and, following an eradication campaign involving bait spraying, male annihilation, fruit collection and destruction, Mauritius was declared free of *B. dorsalis* in 1998 (Seewooruthun, et al., 2000). A further incursion was detected in 2013 and eradicated using similar methods in 2014. The pest was discovered once again in 2015 but at many more locations that previously, so eradication using established methods was not possible. The population is currently being contained and suppressed while an irradiation facility for the breeding of sterile males is being constructed. The first release is scheduled for February 2018 to treat an area of 400 km^2 with the release of 15 million males per week for at least eight months. In the medium term, this programme will be expanded into eradication campaigns for the eight other fruit fly species present in Mauritius (P. Sookar pers. comm. 2017). These planned eradications are being accompanied by increased pest screening of all imported fruit and vegetables.

Disease establishment

A number of zoonotic diseases have been introduced into Mauritius in recent years. The country experienced a major outbreak of chikungunya, a debilitating mosquito-borne virus, in 2006 (Ramchurn, et al., 2008), it's first ever outbreaks of African swine fever in 2007 (Lubisi, et al., 2009) and in 2016 the first foot and mouth disease outbreak in 100 years (Hamuth-Lauloo, et al., 2016). All three diseases are no longer present in the RoM but the outbreaks had major social, political, economic and environmental impacts.

IAS pathways are expanding and diversifying

The major IAS pathways for Mauritius are international shipping and air travel but there is also a risk posed by the unknown numbers of pleasure crafts that land informally on the Mauritian mainland, Rodrigues and their associated islets and are therefore unregulated.

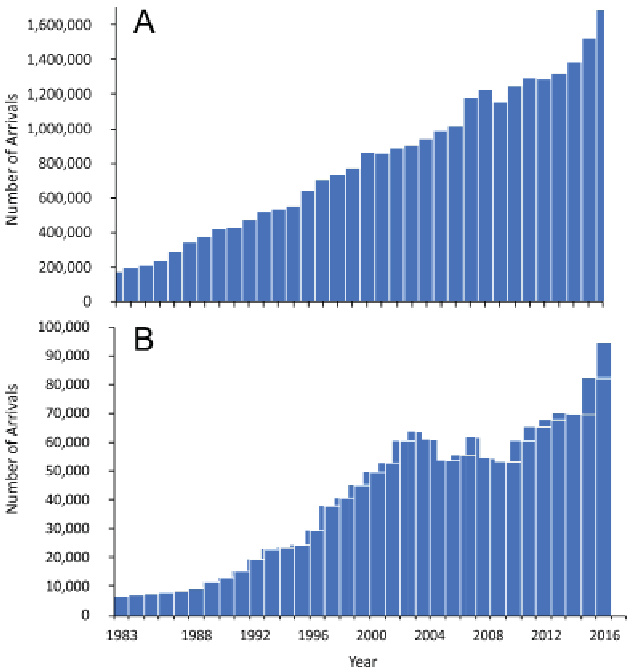

Fig. 4 Passenger arrivals in (A) Mauritius Island and (B) Rodrigues Island 1983–2016 (prior to 1994 figures exclude cruise travellers). Source: Statistics Mauritius (2017).

The volume and diversity of traffic along air and sea pathways into and within the RoM has increased substantially over the past decades. Passenger arrivals into Mauritius Island have increased nearly tenfold from 177,665 in 1983 to 1,684,835 in 2016 (Fig. 4a). Of these arrivals, 409,608 were returning Mauritian residents. Arrivals into Rodrigues over the same period have increased more than fourteenfold from 6,556 to 94,270 (Fig. 4b). Most flights to Rodrigues come from Mauritius Island with an additional scheduled service from Réunion. In June 2017, Air Mauritius, the country's national carrier, was running scheduled services to 24 destinations in 15 countries.

Despite the large increase in absolute numbers, the proportion of travellers from different regions of the world has been relatively consistent (Fig. 5). However, it is highly likely that the diversity of passenger origins has increased substantially although this is not possible to conclude definitively as comprehensive data on passenger's original port of embarkation is only available from 2013. According to Statistics Mauritius (2017), passengers began their journey in at least 110 countries or territories in 2016. Air travel to and through Mauritius is likely to further increase through the continued growth of the tourism industry and the efforts Mauritius is making to position itself as an air travel hub for the fast-growing Africa-Asia market. More and more people coming from more and more biogeographic zones has biosecurity implications for a variety of frontline agencies – currently the airport entry point is staffed by representatives of the the Mauritius Revenue Authority (Customs Department), the Ministry of Agro Industry and Food Security (Plant Protection Office) and Ministry of Health and Quality of Life.

Imports into Mauritius by weight have increased nearly seven-fold from 905,398 tonnes in 1974 to 6,007,056 tonnes in 2016 (Fig. 6). At the same time the number of countries exporting to Mauritius has increased from 33 to 61. However, in contrast to the continued growth in tonnage, the increase in numbers of exporting countries levelled off from 2000 (Fig. 7). The relative importance of exporting countries by monetary value of their exports to Mauritius has changed with two trends being particularly evident: the growth in exports from Asia (from 37–53%) and decline in exports from Europe (from 40–25%) (Fig. 8). These changes have implications for biosecurity including the increased risk that comes with greater volumes of movement, increases in the numbers of source locations and an increase in sources from the warmer parts of the world. However, the precise nature of the risks involved

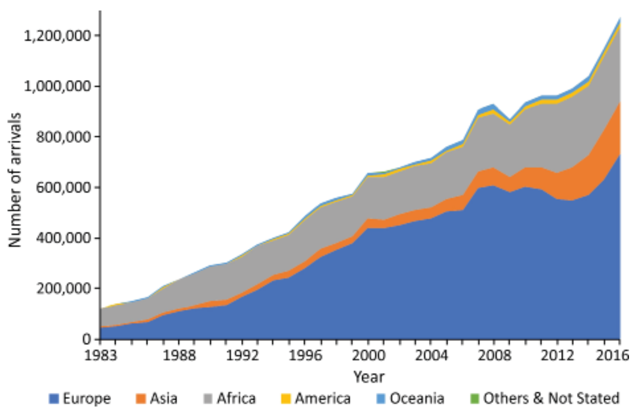

Fig. 5 Foreign passenger arrivals per region of residence. Source: Statistics Mauritius (2017).

cannot be ascertained from aggregate figures. For example, goods such as high value electronics and processed grains, which can be relatively 'clean', pose lower risks than 'dirtier' imports such as semi-processed and unprocessed food and timber, and used machinery which can harbour a wide range of invasive species.

Nearly all official shipping to Rodrigues comes from Mauritius Island, which simplifies pathway analysis. However, pleasure boats and artisanal fishing boats also operate between Mauritius and Rodrigues islands and their offshore islets. Precise numbers of local and international visitors to islets have not been recorded but are known to be in the hundreds of thousands per year. Biosecurity practices are adopted for organised tours of islet nature reserves such as Ile aux Aigrettes and for conservation missions to Round Island and Gunner's Quoin but similar protocols have yet to be formally adopted by private tour operators, pleasure craft owners or fishers.

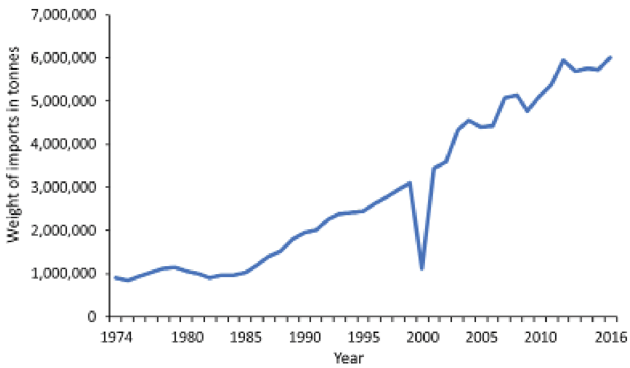

Fig. 6 Volume of imports in tonnes 1974–2016. Source: Statistics Mauritius (2017).

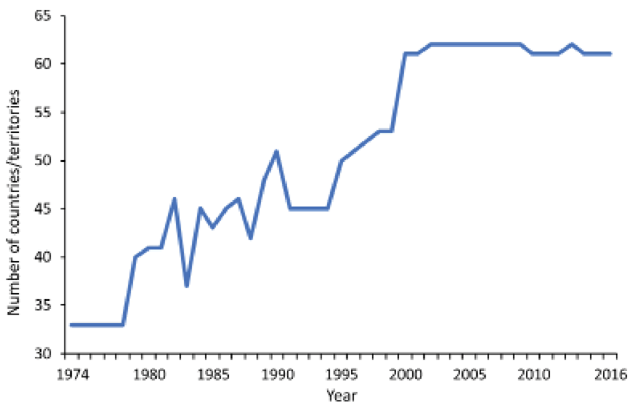

Fig. 7 Number of countries that export to Mauritius 1974–2016. Source: Statistics Mauritius (2017).

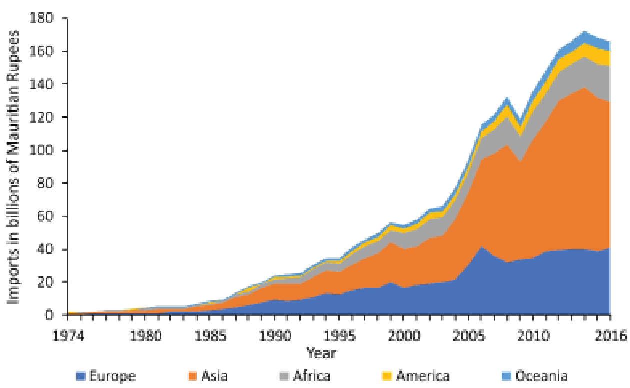

Fig. 8 Value of imports by region 1974–2016. Source: Statistics Mauritius (2017).

IAS MANAGEMENT SUCCESSES AND CHALLENGES

Prevention: keeping white grubs out of Mauritius

The white grub (*Hoplochelus marginalis*), a beetle indigenous to Madagascar, was accidently introduced to Réunion, 150 km to the west of Mauritius, in 1973 in potted ornamental plants and became a threat to Réunion's sugar industry in the 1980s (Jeuffrault, et al., 2004). Sugar cane is Mauritius' principal crop and the generalist white grub also affects other crops and wild grasses. Collaboration between Mauritius and Réunion has prevented the white grub from reaching Mauritius and includes the following measures: reduction in the population densities of white grubs in Réunion by the use of the fungal pathogen *Beauveria brongnartii*; sustained public awareness campaigns; changes in flight and boat departure times in summer when the beetle actively flies around dusk and is attracted to light; regular inspections and spraying around the Mauritius port and airport areas. This systematic approach reflects sugar cane's economic importance and the priority given to agriculture.

Early detection and rapid response: stopping redbacks in their tracks

The redback spider (*Latrodectus hasseltii*) is a venomous Australian spider, responsible for far more bites requiring antivenom than any other creature in Australia. It was found on Gunner's Quoin, an islet of key conservation importance, 8 km from the north coast of Mauritius in 2008 by scientists carrying out conservation activities (N. Cole, pers. obs.). The individual spider and three egg cocoons were found, the spider was collected to confirm identification and the cocoons destroyed, although two had previously hatched. Systematic searches were conducted and subsequently three more spiders and additional egg cocoons were detected and destroyed. Since 2010, there have not been any new detections despite intensive surveys every four to six months. It is not known how the redback was introduced but it is suspected that it could have been a stowaway on private yachts from Australia that are known to travel in the region. Since 2010, 15 invasion events by 10 invertebrate and vertebrate species have been detected on six islets surrounding Mauritius. The periodic presence of biologists on these islets has in most cases permitted rapid response resulting in seven of these invasion events being prevented from establishing or subsequently eradicated with another two eradication efforts ongoing. Increased use of the islets for tourism and leisure activities have been identified as the most significant IAS pathway. Effective biosecurity systems do not exist for most islets with the exception being Round Island which is managed for strict conservation purpose, is difficult to access, and is permanently staffed by conservationists.

Eradication: elimination of foot and mouth disease from Mauritius and Rodrigues

The following is a summary of the detailed account given by Hamuth-Lauloo, et al. (2016). From 7–27 July 2016, 62 cases of cattle illness had been reported in Rodrigues. On 31 July, a team from Mauritius observed Foot and Mouth (FMD) symptoms in cattle and pigs. This was confirmed by blood tests on 1 August. In the meantime, two consignments of livestock had been exported to Mauritius Island. The presence of FMD was confirmed in Mauritius on 5 August. The most probable source of FMD was frozen buffalo meat imported from India via Mauritius. The response comprised of stamping out, movement control, disinfection, quarantine, surveillance, destruction of animal products, official disposal of carcasses, by-products and waste, zoning and vaccination and no FMD cases have been detected from both Rodrigues and Mauritius since December 2016. Inspection of export facilities in India could have prevented the outbreak, more rapid diagnosis and better inter-island quarantine could have reduced its severity and spread and a contingency plan would have resulted in a more coordinated response than was the case.

Management: the use of biological control within an IPM approach in the agricultural sector

As outlined, there has been an increased rate of insect pest introduction to Mauritius since the 1970s. This has been one of the reasons for the growing use of pesticides in Mauritian agriculture. However, at the same time, the country, notably through the sugar sector, which barely uses insecticides, has made grounds in integrated pest management (IPM), advocating a package of measures designed to reduce the prophylactic use of pesticides. One of the main planks of this approach has been the use of biological control. This has been reflected in the consistent use of biological control agents (parasitoids, pathogens, and biopesticides) in recent decades. A major recent success was the introduction of the parasitoid *Acerophagus papayae* in 2013 to control the papaya mealybug.

The sectoral nature of IAS management – the case of biological control

Approaches developed in one sector are not necessarily adopted and adapted to other sectors. An example of this is biological control which is actively pursued in agriculture but not in the conservation sector. The priority given to biological control in agriculture, using protocols based on International Sanitary and Phytosanitary Measures (ISPMs), reflects the sector's economic importance and the clear direction offered to the plant protection sector through the International Plant Protection Convention (IPPC), to which the country has been a signatory since 1971. There has been no deliberate introduction of a biocontrol agent against an invasive plant that threatens native biodiversity since 1982, despite the fact that biocontrol of environmental weeds in Mauritius has a very successful history with a full/partial success rate of 80% (Fowler, et al., 2000). Ironically, two recent examples of possible biosecurity failures are likely to have had positive impacts on biodiversity in the RoM. Firstly, there is the movement of *Teleonemia scrupulosa* (lantana lace-bug), a biological control agent for *Lantana camara* (vieille fille) already present in Mauritius, to Rodrigues which has hugely reduced the vigour of *L. camara* in areas of conservation importance and on rangeland in Rodrigues. Secondly, the spread of the biocontrol agent *Cibdela janthina* (mouche bleu) from Réunion to Mauritius, which may have arrived in 2015 (Florens, et al., 2017), has the potential to substantially reduce the vigour of *Rubus alceifolius* (giant bramble) a major invasive plant in Mauritian forests. *C. janthina* could have conceivably dispersed naturally from Réunion but the chances of the *L. camara* agent dispersing naturally from Mauritius to Rodrigues are very low. Whatever the case, biosecurity systems need to be tightened but responsible biological control for invasive plants must to be part of an integrated approach to invasive plant management.

THE DEVELOPMENT OF A NATIONAL INVASIVE ALIEN SPECIES STRATEGY AND ACTION PLAN

Following its accession to the Convention on Biological Diversity in 1992, the Mauritian conservation community was very actively engaged in the Global Invasive Species Programme (GISP) which operated between 1997 and 2011 to encourage the adoption of measures in line with CBD Article 8h: "Each Contracting Party shall, as far as possible and as appropriate prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species" (UN, 1992). Towards this end, MAIFS established the National Invasive Alien Species Committee (NIASC) in August 2003. One of the priorities for the NIASC, which comprises representatives from the agriculture, biodiversity conservation, health, environment and education sectors as well as the private sector, was the production of a National Invasive Alien Species Strategy and Action Plan (NIASSAP) for the Republic of Mauritius (RoM, 2010). Funding was secured for its development from 2008 and the NIASSAP was approved by cabinet in 2010. The NIASSAP presents a vision in which the negative impacts of IAS on the economy, environment and society of the RoM are avoided, eliminated or minimised. The strategy was based on the assumptions that an effective biosecurity system is built upon a risk analysis framework and that its success depends upon effective collaboration between all those concerned with invasion pathways.

The Strategy comprises ten interlinked elements: five hierarchical "Management Elements" and five "Cross-Cutting Elements". The management elements are those "on the ground actions" that directly address the Strategy's vision. The cross-cutting elements are enabling actions.

The NIASSAP Management Elements, with their accompanying goal or goals are listed in order of priority based on the maxim that "prevention is better than cure":

1. Prevention – to minimise the number of unintended and intended IAS introductions to the RoM;
2. Early detection and rapid response – to minimise the number of IAS that go on to have harmful consequences once they are introduced to the RoM;
3. Eradication – an agreed framework for eradication priorities in place, eradications undertaken as necessary and results disseminated;
4. Control and management – to contain the distribution and abundance of IAS in the RoM to a long-term acceptable level; and
5. Restoration – to undertake ecosystem restoration where necessary in the RoM to achieve long-term ecosystem goals.

The Cross-Cutting Elements, again listed with their goal or goals, are:

6. Legal, policy and institutional frameworks – to have a coordinated policy and management framework that minimise the risk of IAS;
7. Capacity building and education – to make available appropriately skilled personnel to implement all aspects of IAS management in the country;
8. Information management and research – (i) To have a clear understanding of the impacts of IAS that have become established in the RoM; (ii) to have ready access to critical information that will support IAS management programmes and (iii) to provide a strong scientific basis for decision-making and resource allocation;
9. Public awareness and engagement – all stakeholders in the RoM should have a high level of awareness of IAS risks and the benefits of IAS prevention and management;
10. International cooperation – (i) the RoM should have access to the necessary information, technical and financial support and other resources it needs to effectively meet its international obligations; (ii) Mauritian IAS experiences and lessons learnt are effectively disseminated to help IAS initiatives regionally and internationally and (iii) the RoM is not a source of IAS for other countries

Partial implementation of the NIASSAP (2010–2017)

The NIASSAP has yet to be systematically implemented. Major reasons for this were the fact that lead agencies were not designated to carry out each action and timelines, milestones and estimates of resources required were not agreed upon. The National IAS Committee only met sporadically between 2010 and 2015 and was only made statutory in 2015 under the Native Terrestrial Biodiversity and National Parks Act (2015).

However, actions in line with the NIASSAP have been undertaken in Mauritius since 2010, some of which have been outlined above, but they were not implemented because of the NIASSAP.

The prospects for effective implementation of the NIASSAP received a boost with a broadly costed and timetabled provision for its implementation under the National Biodiversity Strategy and Action Plan (NBSAP) 2017–2025 as the National contribution to Aichi Target 9: “By 2025, the NIASSAP is revised and fully implemented through adequate financial and human resources commensurate to the existing challenges, and the impacts caused by IAS are minimised” (RoM, 2017). Linked to the above, from 2015–2018, has been the development of a UNDP/GEF VI Project to mainstream IAS prevention, control and management (US$20.89M project: US$3.89M from Global Environment Facility and US$17M from National co-financing).

Next steps: mainstreaming IAS prevention, control and management

The objective of the ‘IAS Mainstreaming Project’ is to safeguard globally significant biodiversity in vulnerable ecosystems through the prevention, control and management of IAS in the RoM. This will be achieved through four outcomes which are summarised below together with key outputs and activities that contribute to intended outcomes:

1. By 2024, the RoM has a gender sensitive policy, regulatory and institutional framework and capacity to manage IAS effectively:
   - The NIASSAP is reviewed and revised, with progress assessed, gaps identified and activities fully costed with precise timelines for implementation for both terrestrial and marine IAS;
   - Existing legislation is strengthened for more effective control and management of IAS;
   - A cross-sectoral policy coordination framework is established for the incorporation of IAS issues into the legal and policy framework of all relevant agencies;
   - A technical secretariat for IAS is established;
   - Capacity is strengthened in key agencies and organisations;
   - Financial sustainability of the apex agency and IAS operations are secured through the development and application of new market-based and fiscal mechanisms and incentives to support IAS management.
2. By 2024, the government effectively prevents and manages IAS threats based on risks:
   - National and inter-island biosecurity priorities and resource needs, including baselines are established;
   - A comprehensive risk assessment system is in place and being used in the Republic of Mauritius, to (1) assess the risks that new species proposed for importation to the RoM or moved between its islands may become invasive (border control), and (2) assess the risks associated with species already present in the RoM but which may not yet have become invasive there;
   - Species identified by formal risk assessment as having high invasiveness potential in the Republic of Mauritius are refused permission for importation or for translocation between its islands;
   - Procedures for controlling the unregulated (illegal) importation of species to the Republic of Mauritius or between its islands are improved (effective quarantine system with sanctions for deliberate infractions);
   - Species present in the Republic of Mauritius, with high invasive potential but still present only in limited areas, are prioritised for management and, where feasible, eradication by means of a formal risk assessment process, including, as far as possible, their declaration as “harmful”, “prohibited”, or similar;
   - Pilot biodiversity conservation and ecological restoration operations developed on key islets and in Rodrigues;
   - Equipment and infrastructure updated to help ensure that priority biosecurity measures are effectively implemented.
3. By 2024, planning, management and decision-making by all relevant stakeholders are informed by knowledge management and learning:
   - Review and survey of the status of IAS pathways, IAS distributions, the cross-sectoral economic, environmental and cultural impact of IAS and the successes and lessons learnt from past and ongoing IAS prevention, early detection and rapid response, eradication, control and mitigation and restoration;
   - Up-to-date lists of terrestrial and marine invasive species of all taxa present in the Republic of Mauritius are completed and publicly available, and a system for their regular updating is in place and is being used;
   - Pathways of introduction of new species into Mauritius and between the islands of the Republic of Mauritius are identified, their relative importance is quantified, and they are prioritised for management action to reduce the rate of arrival of new species;
   - A national IAS information system is developed and operationalised to monitor and inform risk-based management of species, pathways and ecosystems based on agreed protocols;
   - A national IAS gateway is developed to provide rapid access and dissemination of information to enhance deployment of coordinated actions between institutional partners on IAS management;
   - A national IAS communications and awareness strategy and action plan is developed and implemented;
   - IAS tools and manuals are developed to complement training courses and for use in day to day IAS management operations, and guidelines are developed to embed IAS issues into key sectors whose activities have IAS implications.

Project risks include increased liberalisation of movement and trade, the continuation of a fragmented sector by sector and case by case approach, lack of support for strengthened biosecurity measures at different levels, and economic and political pressure being used to circumvent decision-making based on a transparent risk analysis process. It is clear, therefore, that the NIASSAP represents an ambitious and costly undertaking, but the costs of not systematising IAS prevention and management (business as usual) are likely to be considerably higher.

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REFERENCES

Abeeluck, D., Dunhawoor, C. and Unmole, L. (2009). 'Development and implementation of integrated pest management in Mauritius: An overview'. *University of Mauritius Research Journal* 15: 85–97.

Baret, S., Baider, C., Kueffer, C., Foxcroft, L.C. and Lagabrielle, E. (2013). 'Threats to Paradise? Plant Invasions in Protected Areas of the Western Indian Ocean Islands'. In: L.C. Foxcroft, P. Pyšek, D.M. Richardson and P. Genovesi (eds.) *Plant Invasions in Protected Areas: Patterns, Problems and Challenges*, pp. 423–447. Dordrecht, Netherlands: Springer.

Bullock, D.J. (1986). 'The ecology and conservation of reptiles on Round Island and Gunner's Quoin, Mauritius'. *Biological Conservation* 37: 135–156.

CABI. (2017). '*Bactrocera dorsalis* (oriental fruit fly), Invasive Species Compendium'. <http://www.cabi.org/isc/datasheet/17685>. Accessed 30 April 2017.

Cheke, A.S. (1987). 'An Ecological History of the Mascarene Islands, with Particular Reference to Extinctions and Introductions of Land Vertebrates'. In A.W. Diamond (ed.) *Studies of Mascarene Island Birds*, pp. 5–89. Cambridge: Cambridge University Press.

Cheke, A.S. and Hume, J.P. (2008). *Lost Land of the Dodo: An Ecological History of Mauritius, Réunion and Rodrigues*. London: T & AD Poyser.

Cole, N. and Payne, C. (2015). '*Gongylomorphus bojerii*, The IUCN Red List of Threatened Species 2015'. <http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T62251A13482733.en>. Accessed 25 June 2017.

Csurhes, S. (2009). 'Weed Risk Assessment: Giant Reed (*Arundo donax*)'. Australia: The State of Queensland: Department of Employment, Economic Development and Innovation. <http://www.feedipedia.org/node/18477>. Accessed 10 April 2017.

Florens, F.B.V. (2013). 'Conservation in Mauritius and Rodrigues: Challenges and Achievements from two Ecologically Devastated Oceanic Islands'. In: N.S. Sodhl and P.H. Raven (eds.) *Conservation Biology: Voices from the Tropics*, pp. 40–50. Winchester: Wiley-Blackwell.

Florens, F.B.V., Baider, C., Martin, G.M.N., Seegoolam, N.B., Zmanay, Z. and Strasberg, D. (2016). 'Invasive alien plants progress to dominate protected and best-preserved wet forests of an oceanic island'. *Journal for Nature Conservation* 34: 93–100.

Florens, F.B.V., Bissessur, P., Bunsy, Y. and Ramdonee, D. (2017). '*Cibdela janthina* (Klug 1834) (Hymenoptera: Tenthredinidae), Réunion's biocontrol agent of *Rubus alceifolius* Poir., recorded on Mauritius'. *African Entomology*, 25(1): 271–274.

Fowler, S.V., Ganeshan, S., Mauremootoo, J.R. and Mungroo, Y. (2000). 'Biological Control of Weeds in Mauritius: Past Successes Revisited and Present Challenges'. In: N.R. Spencer (ed.) *Proceedings of the X International Symposium on Biological Control of Weeds*, pp. 43–50. Bozeman, Montana: Montana State University.

Hamuth-Lauloo, S.B.A., Sibartie, D. and André, A.A. (2016). 'Report of the Fact-Finding Committee on the Outbreak of Foot and Mouth Disease'. Government of Mauritius: Port Louis. <http://agriculture.govmu.org/English/Documents/Report/full%20report.pdf>. Accessed 30 April 2017.

Heeroo, R. (2000). 'Documentation of the Weed Status of Exotic Plants in Mauritius based on the Records of the Mauritius Herbarium'. BSc dissertation. Réduit: University of Mauritius.

Jeuffrault, E., Rolet, A., Reynaud, B., Manikom, R., Georger, S., Taye, T., Chiroleu, F., Fouillaud, M. and Vercambre, B. (2004). 'Vingt ans de lutte contre le ver blanc de la canne à sucre à la Réunion: Un succès, mais il reste des questionnements scientifiques pour confirmer la durabilité de la lutte biologique'. *Phytoma–La Défense des Végétaux* 573: 16–19.

Lubisi, B.A., Dwarka, R.M., Meenowa, D. and Jaumally, R. (2009). 'An Investigation into the first outbreak of African swine fever in the Republic of Mauritius'. *Transboundary and Emerging Diseases* 56: 178–188.

Myers N., Mittermeier, R.A., Mittermeier, C.G. et al. (2000). 'Biodiversity hotspots for conservation priorities'. *Nature* 403: 853–858.

Ramchurn, S.K., Goorah, S.S.D., Mungla, D., Ramsurrun, B., Pydiah, V. and Summun, A. (2008). 'A study of the 2006 Chikungunya epidemic outbreak in Mauritius' *Internet Journal of Medical Update* 3(1): 11–21.

Richardson, D.M. (1998). 'Forestry trees as invasive aliens'. *Conservation Biology* 12(1): 18–26.

RoM. (2010). 'The National Invasive Alien Species Strategy and Action Plan for the Republic of Mauritius'. Port Louis: Ministry of Agro Industry and Food Security.

RoM. (2017). 'National Biodiversity Strategy and Action Plan (NBSAP) 2017 – 2025'. Port Louis: Ministry of Agro Industry and Food Security.

Seewooruthun, S.I., Permalloo, S., Gungah, S., Soonnoo, A.R. and Alleck, M. (2000). 'Eradication of an Exotic Fruit Fly from Mauritius'. In: K-H. Tan (ed.) *Area-Wide Control of Fruit Flies and Other Insect Pests: Joint Proceedings of the International Conference in Area-Wide Control of Insects Pests, May 28–June 5, 1998, and the Fifth International Symposium on Fruit Flies of Economic Importance, June 1–5, 1998*, pp. 389–394. Pulau Pinang, Malaysia: Penebrit Universiti Sains.

Statistics Mauritius (2017). '2015–2016 Statistics Mauritius Annual Report'. Port Louis, Mauritius: Ministry of Finance & Economic Development.

UN. (1992). 'Convention on Biological Diversity'. Nairobi. <https://www.cbd.int/convention/text/default.shtml>. Accessed 30 April 2017.

Virah-Sawmy, M., Mauremootoo, J.R., Marie, D., Motala, S.M. and Sevathian, J-C. (2009). 'Rapid degradation of a Mauritian rainforest following 60 years of plant invasion'. *Oryx* 43(4): 599–607.

Williams, J.R. and Ganeshan, S. (2001). 'Insects introduced into Mauritius during the 20th century: Natural, accidental and deliberate introductions'. *Revue Agricol et Sucrière de l'Ile Maurice* 78: 33–50.

Islander perceptions of invasive alien species: the role of socio-economy and culture in small isolated islands of French Polynesia (South Pacific)

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Abstract Islands, often celebrated as natural laboratories for evolution and ecology, also provide unique experimental grounds for societal studies. Although biological invasions are widely recognised as one of the main causes of biodiversity erosion and a driver of global change, the human perception of invasive species may vary at regional and local levels, especially in societies with different levels of socio-economic development and cultures. This study was conducted in French Polynesia (South Pacific), a territory formed by 120 tropical and subtropical oceanic islands (76 being inhabited) divided into five archipelagos (Austral, Marquesas, Society, Tuamotu, and Gambier Is), comprising both highly populated and urbanised islands (such as Tahiti in the Society Is) and less populated and very small islands, sometimes very isolated (without airstrips) and where traditional life style and strong dependence on natural resources still persist. During an eight-month education and prevention campaign targeting alien plant and animal species legally declared invasive in French Polynesia, public meetings were organised on 19 small islands for a total of 2,045 consulted people in 41 different villages. Negative, positive and neutral comments made by participants on some invasive species present in their islands were recorded. Our results show that their perceived status differs from one archipelago to another, or even among islands in the same archipelago, with more positive comments (i.e. species benefits) on more isolated islands. Perception of invasiveness varied according to societal and cultural values (e.g. utilitarian or aesthetic), and often depends on the species' date of introduction ("indigenisation" of old introduced plants and animals). These surveys can provide useful baseline information on the degree to which local island communities are likely to support invasive species management, to get involved in prevention, surveillance and control efforts, and to avoid potential conflicts of interest between different stakeholders in small but sometimes complex insular societies.

Keywords: conflicts, indigenisation, invasiveness, isolation, prevention, social dimension, values

INTRODUCTION

The human or social dimension is increasingly recognised as a crucial issue for the effective management of invasive alien plants and animals (McNeely, 2000; Marshall, et al., 2011; Estévez, et al., 2014). Indeed, many control, eradication or prevention programs have been delayed or even failed because of differing public attitudes and feelings towards the targeted invasive species. The various stakeholders (such as foresters, pastoralists, horticulturists, pet shop managers, conservationists and environmentalists) may have different or opposite views of species status (e.g. "noxious/harmful" versus "useful/beneficial" species) and strong opposition by some influential groups of people or even single individuals may occur. Control or eradication programs of animals such as feral cats (*Felis catus*), feral deer (*Cervus* spp.), pigs (*Sus scrofa*), or grey squirrels (*Sciurus carolinensis*) (see references in McNeely, 2000; Estévez, et al., 2014), and of plants such as gorse (*Ulex europaeus*) in New Zealand (Hill, 1989) or strawberry guava (*Psidium cattleianum*) in the Hawaiian islands (Veitch & Clout, 2000; Warner & Kinslow, 2013) and La Réunion (Mascarene Is, Indian Ocean) are well-documented examples of social conflicts of interests, often associated with "controversies" reported in public and media opinions.

Thus, studying human perceptions and attitudes towards invasive species is often useful and sometimes an important prerequisite before starting often costly and long-term management programmes. Many recent studies have been conducted in "western" and/or well-developed regions/countries, such as Europe, Canada and USA (Bremner & Park, 2007; Garcia-Llorente, et al., 2008; Selge, et al., 2011; Fischer, et al., 2014), and New Zealand (Fraser, 2001; Russell, 2014), using questionnaires or interviews addressed to different stakeholders among different socio-professional categories. A few other studies have been conducted in developing countries where invasive species may sometimes constitute a natural resource rather than a nuisance (e.g. the potential use of water hyacinth (*Eichhornia crassipes*) as biofuel in south-east Asia, Bhattacharya & Kumar, 2010). The case of "true" island countries and territories (excluding large continental islands such as Australia, Madagascar, or Great Britain) is even less studied, although they are highly vulnerable to the impacts of invasive alien species, with many cases of native species' extinction and extirpation and stronger conservation challenges. Moreover, islands, often celebrated as natural laboratories for evolution and ecology, may also provide unique experimental grounds for societal and cultural studies, as they also harbour a high cultural diversity and different levels of socio-economic development. In this study conducted in the small tropical oceanic islands of French Polynesia (South Pacific), we tested the two following hypotheses:

Does human perception of invasive species vary with island isolation, human population and socio-economic development?

What is the influence of cultural (traditional) values on public attitudes toward introduced species in small remote islands?

MATERIAL AND METHODS

This study was conducted in French Polynesia, a European Overseas Country and Territory (OCT) located in the South Pacific, formed by about 120 small tropical oceanic islands (76 being inhabited by a total of ca. 276,000 inhabitants in 2017) divided into five archipelagos (Austral, Marquesas, Society, Tuamotu, and Gambier Is), and dispersed over a marine area as wide as Europe (Fig. 1).

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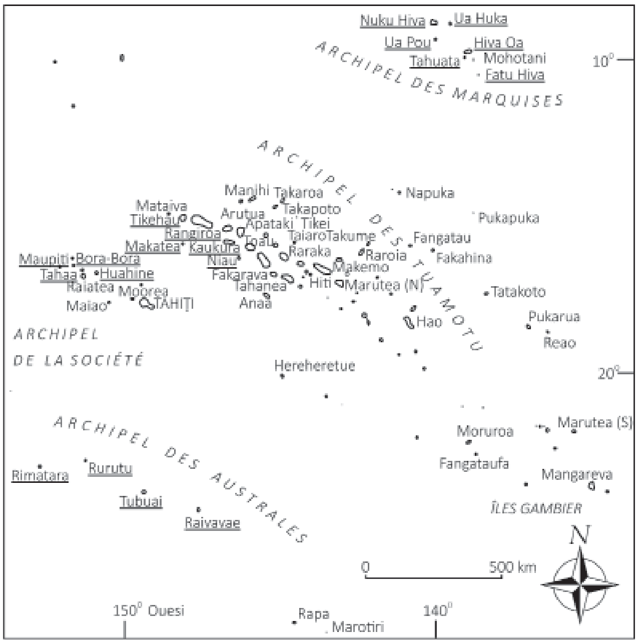

Fig. 1 French Polynesia and its 120 tropical oceanic islands located in the South Pacific. The names of the 19 surveyed small islands are underlined.

This OCT comprises both highly populated and urbanised islands (such as Tahiti, the largest with a land area of 1,045 km² and over 183,000 inhabitants) and very small isolated islands (sometimes without airstrips such as Tahaa in the Society, Fatu Iva and Tahuata in the Marquesas, Makatea in the Tuamotu, Rapa in the Austral Is with an area of only 40 km² and 515 inhabitants), which are less populated and developed, where traditional lifestyles and strong dependence on natural marine and terrestrial resources still persist. As an example, coconut plantations for copra and coconut oil production remain the main source of income in the Leeward Islands (Society), the Tuamotu atolls and the Marquesas high volcanic islands (IEOM, 2017). The island isolation or "remoteness" (distance from the most urbanised and populated island of Tahiti in km) and the number of regular flights per week departing from Tahiti or "connectivity" were used as proxies for the socio-economic development of each surveyed island.

Environmental matters and issues fall to the authorities of the French Polynesian Government, (i.e. they are different from French laws and regulation texts), with a "Code de l'Environnement de la Polynésie Française" voted by the Assembly of French Polynesia in 2003, including a chapter specifically dedicated to invasive alien species. A total of 46 species including 35 plants and 11 animals have been legally declared "a threat to biodiversity" in French Polynesia (Table 1) because of their significant negative impacts on the endemic fauna and flora. New introduction, culture or propagation, as well as inter- and intra-island transportation, of these species is banned in all islands of French Polynesia and control or eradication programmes have been set up. Their presence on each inhabited island was compiled based on literature, plant and animal databases and local expertise (Fourdrigniez, et al., 2014).

During a communication, education, prevention and capacity building campaign conducted (by the second author M.F.) between May and December 2014 (about eight months), public meetings were organised on 19 small islands (< 400 km² and 10,000 inhabitants) within 41 different villages. A total of about 2,045 people were consulted (Table 3). These meetings were held at the city halls ("mairie" in French) or community houses during the morning or the evening, and were attended mainly by adults (for a total of 1,781) and some schoolchildren.

An oral PowerPoint presentation listing and describing the 46 legally declared invasive species (38 of which were present in the surveyed small islands) was delivered, without providing details on their ecological and socio-economical impacts. Two main following questions were asked to the participants:(1) do you know or have you seen these species in your island? (2) do you consider them invasive (i.e. abundant and/or spreading) in your island, and where (i.e. which locations)?

Although no direct question was asked about species perceptions and associated values, comments were given by participants related to the negative impacts of species on biodiversity and other sectors (e.g. agriculture, health), and also their positive impacts (past and current benefits), which were systematically recorded.

RESULTS

Effects of island isolation, human population and socio-economic development

The total number of legally declared invasive alien species known to be present in each surveyed island (according to Fourdrigniez, et al., 2014) in the four archipelagos of the Leeward (Society), Austral, Marquesas and Tuamotu Is does not decrease with island remoteness (Fig. 2), comprising 44 of the 46 invasive alien species (Table 3). Invasive species diversity also does not increase with island size (Table 3) although the two largest remote islands of Hiva Oa and Nuku Hiva in the Marquesas (> 300 km² of land area) have a high proportion of species (between 50–56% of the total), probably related to their higher habitat diversity (ranging from coastal vegetation and littoral forest to dry-mesic forests, valleys and slopes rainforests, and montane cloudforests and summit ridges up to 1,200 m elevation, Lorence, et al., 2016) compared to the other surveyed islands. There is a relatively weak correlation between invasive species and the number of inhabitants (R^2=0.48, P-value < 0.01, Fig. 3a), which becomes stronger with the number of regular flights departing from Tahiti per week (R^2=0.53, P-value < 0.05, Fig. 3b), i.e. with human and goods transportation connection and frequency. This "connectivity" between Tahiti and the other French Polynesian islands constitutes a very good proxy for the socio-economic development of isolated islands. If the Tuamotu atolls are removed from the analysis, the correlation coefficient is significantly higher (R^2=0.72). Indeed, the atolls and raised atolls have fewer invasive species mainly because of their small terrestrial areas, their calcareous substrate and strong insolation

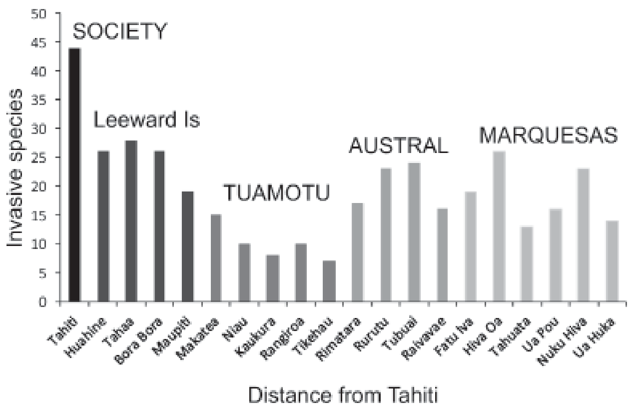

Fig. 2 Relationship between the number of invasive alien species on Tahiti and the 19 surveyed small islands according to distance from Tahiti: Leeward Is (Society Is) >170–310 km from Tahiti; Tuamotu Is >220–350 km; Austral Is >500–700 km; Marquesas Is >1,000–1,500 km (Spearman test, P-value = 0.000995).

Table 1 List of the 46 invasive alien species legally declared a "threat to biodiversity in French Polynesia" (according to the French Polynesia "Code de l'Environnement") and their presence in Tahiti and the other 19 surveyed small islands (Fourdrigniez, et al., 2014).

ANIMALS: INVERTEBRATES AND VERTEBRATES (N=11)				
Kingdom	Scientific name	Common name	Tahiti	Surveyed islands (%)
Insects	*Wasmannia auropunctata**	Little fire ant	X	0 (0%)
Molluscs	*Euglandina rosea**	Rosy wolfsnail	X	7 (36.8%)
Birds	*Acridotheres tristis**	Common myna	X	5 (26.3%)
	Bubo virginianus	Great horned owl	-	1 (5.3%)
	Circus approximans	Swamp harrier	X	4 (21.1%)
	*Pycnonotus cafer**	Red-vented bulbul	X	5 (26.3%)
Reptiles	*Trachemys scripta**	Red-eared slider	X	4 (21.1%)
Mammals	*Mus musculus**	House mouse	X	12 (63.2%)
	Rattus exulans	Pacific rat	X	19 (100%)
	Rattus norvegicus	Norway rat	X	13 (68.4%)
	*Rattus rattus**	Black rat	X	12 (63.2%)

VASCULAR PLANTS (N=35)				
Family	Scientific name (synonyms)	Habit	Tahiti	Surveyed islands (%)
Euphorbiaceae	*Antidesma bunius*	Tree	X	0 (0%)
Myrsinaceae	*Ardisia elliptica**	Small tree	X	2 (10.5%)
Moraceae	*Castilla elastica*	Tree	X	4 (21.1%)
Cecropiaceae	*Cecropia peltata**	Tree	X	6 (31.6%)
Chrysobalanaceae	*Chrysobalanus icaco*	Small tree	X	5 (26.3%)
Rubiaceae	*Cinchona pubescens**	Tree	X	0 (0%)
Hydrocharitaceae	*Egeria densa*	Aquatic herb	X	0 (0%)
Myrtaceae	*Eugenia uniflora*	Small tree	X	14 (73.7%)
Fabaceae	*Falcataria* (syn. *Albizia*) *moluccana*	Large tree	X	13 (68.4%)
Fabaceae	*Flemingia strobilifera*	Shrub	X	14 (73.7%)
Agavaceae	*Furcraea foetida*	Erect herb	X	7 (36.8%)
Crassulaceae	*Kalanchoe pinnata*	Erect herb	X	18 (94.7%)
Verbenaceae	*Lantana camara**	Shrub	X	15 (78.9%)
Fabaceae	*Leucaena leucocephala**	Small tree	X	19 (100%)
Convolvulaceae	*Merremia peltata*	Liana (woody vine)	X	8 (42.1%)
Poaceae	*Melinis minutiflora*	Grass	X	16 (84.2%)
Melsatomataceae	*Miconia calvescens**	Small tree	X	3 (15.8%)
Asteraceae	*Mikania scandens* (syn. *M. micrantha*)*	Vine	X	0 (0%)
Mimosaceae	*Mimosa diplotricha* (syn. *M. invisa*)	Shrub	X	7 (36.8%)
Passifloraceae	*Passiflora maliformis*	Liana (woody vine)	X	11 (57.9%)
Passifloraceae	*Passiflora rubra*	Vine	-	1 (5.3%)
Passifloraceae	*Passiflora suberosa*	Vine	X	2 (10.5%)
Asteraceae	*Pluchea symphytifolia*	Shrub	X	4 (21.1%)
Myrtaceae	*Psidium cattleianum**	Small tree	X	10 (52.6%)
Myrtaceae	*Rhodomyrtus tomentosa*	Small tree	X	0 (0%)
Rosaceae	*Rubus rosifolius*	Shrub	X	4 (21.1%)
Anacardiaceae	*Schinus terebinthifolius**	Tree	X	0 (0%)
Araliaceae	*Schefflera actinophylla*	Tree	X	5 (26.3%)
Bignoniaceae	*Spathodea campanulata**	Large tree	X	7 (36.8%)
Myrtaceae	*Syzygium cumini*	Tree	X	19 (100%)
Myrtaceae	*Syzygium jambos*	Tree	X	14 (73.7%)
Bignoniaceae	*Tecoma stans*	Small tree	X	9 (47.4%)
Polygonaceae	*Triplaris weigeltiana*	Large tree	X	0 (0%)
Fabaceae	*Vachelia* (syn. *Acacia*) *farnesiana*	Small tree	X	4 (21.1%)
Myrtaceae	*Waterhousea floribunda*	Tree	X	1 (5.3%)

*Listed among the "100 of the World's Worst Invasive Alien Species" (Lowe, et al., 2000).

Table 2 Number of surveyed islands, villages and people (adults) consulted during public meetings in the different archipelagos of French Polynesia.

Archipelagos	Number of surveyed islands (names)	Number of villages	No of participants (adults)
Leeward Is (Society Is)	4 (Maupiti, Tahaa, Huahine, Bora Bora)	9	494
Tuamotu Is	5 (Niau, Kaukura, Makatea, Tikehau, Rangiroa)	9	479
Austral Is	4 (Raivavae, Rimatara, Rurutu, Tubuai)	10	414
Marquesas Is	6 (Nuku Hiva, Ua Pou, Ua Huka, Hiva Oa, Fatu Iva, Tahuata)	13	394
Total	19	41	1,781

Table 3 Number and density of invasive alien species (IAS) legally declared "a threat to biodiversity in French Polynesia" in relation to geographic and demographic characteristics of islands, and plane transportation frequency or "connectivity" with Tahiti: ✈✈ island with an international airport; ✈ islands with a domestic airport or airstrip; 2012 population census (<www.ispf.pf>).

ARCHIPELAGO (distance from Tahiti in km)	Island (number of flights per week departing from Tahiti)	Area (ha)	Population (2012)	Population density (/ha)	IAS number (%)	IAS density (/ha)
SOCIETY (170-310 km)	Tahiti✈✈	104,510	183,480	1.76	44 (96%)	0.04
	Tahaa (61 via Raiatea)	9,020	5,220	0.58	28 (60.9%)	0.31
	Huahine✈ (37)	7,480	6,303	0.84	26 (56.5%)	0.35
	Bora Bora✈ (74)	2,930	9,598	3.27	26 (56.5%)	0.89
	Maupiti✈ (9)	1,140	1,223	1.07	19 (41.3%)	1.67
TUAMOTU (220-350 km)	Rangiroa✈ (20)	7,900	2,567	0.32	10 (21.8%)	0.13
	Makatea	2,950	68	0.02	15 (32.6%)	0.51
	Niau✈ (2)	2,100	226	0.11	10 (21.8%)	0.48
	Tikehau✈ (10)	2,000	529	0.26	7 (15.2%)	0.35
	Kaukura✈ (2)	1,100	475	0.43	8 (17.4%)	0.73
AUSTRAL (500-700 km)	Tubuai✈ (14)	4,500	2,170	0.48	24 (52.2%)	0.54
	Rurutu✈ (12)	3,235	2,322	0.72	23 (50%)	0.71
	Raivavae✈ (7)	2,035	940	0.46	16 (34.8%)	0.79
	Rimatara✈ (5)	953	873	0.91	17 (36.9%)	1.78
MARQUESAS (1,000-1,500 km)	Nuku Hiva✈ (15)	33,950	2,967	0.03	23 (50%)	0.07
	Hiva Oa✈ (15)	31,550	2,184	0.07	26 (56.5%)	0.08
	Ua Pou✈ (9)	10,560	2,175	0.21	16 (34.8%)	0.74
	Ua Huka✈ (6)	8,340	621	0.07	14 (30.4%)	0.17
	Fatu Iva	8,500	611	0.07	19 (41.3%)	0.22
	Tahuata	6,100	703	0.11	13 (28.3%)	0.21
TOTAL	20	250,863	222,688	0.89	46 (100%)	0.02

which constitute demanding ecological conditions for both introduced animals and plants. The Austral high volcanic islands have a cooler climate due to their southern geographical location (mean annual temperature between 18°C for Rapa Iti and 20°C for the other islands) which may also prevent the establishment and invasion of some "truly" tropical species. If the Austral islands are removed from the analysis, the correlation coefficient is slightly higher (R^2=0.57).

Perceptions of invasive species in different archipelagoos and islands

The total number of negative, positive and neutral comments (50) recorded by participants for each species was analysed for all the 19 surveyed islands. Comments were reported only for 15 of the 38 species occuring in the islands, most of them were positive (Fig. 5). More comments were made in the isolated islands of the Austral Is (> 500–700 km from Tahiti) and the Marquesas Is (> 1,000–1,500 km) with lower socio-economic development but where people seem to show a stronger interest in the use of available natural resources (Fig. 4), compared to the Leeward Is in the Society Is. Comments in the Tuamotu Is were the lowest and the number of reported invasive species is also the smallest (between 7 and 15 species, i.e. 15–33% of the total). It is noteworthy that all comments made on invasive species were positive in the Tuamotu atolls (Fig. 4), meaning they are more considered as "useful" for people than "noxious/harmful". In all surveyed islands and archipelagos, positive comments exceeded negative ones, but this rather surprising result might be biased as

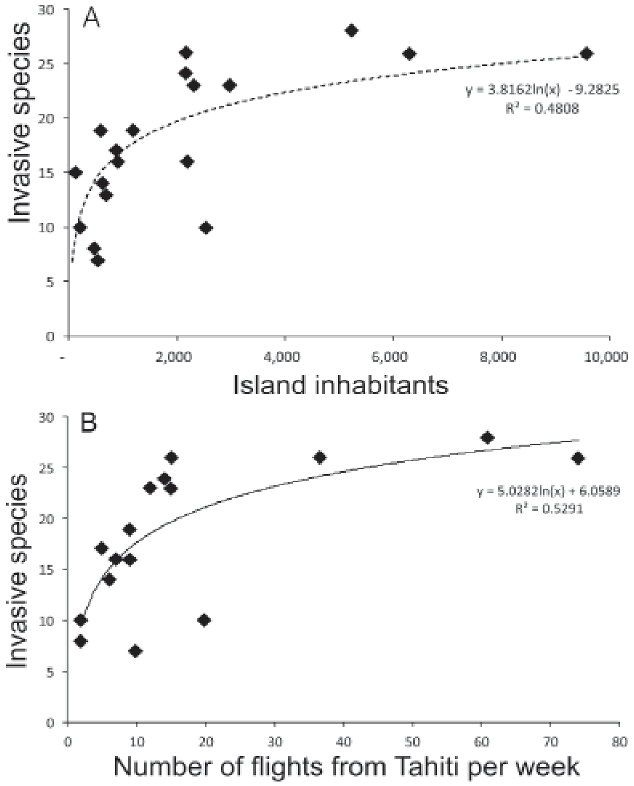

Fig. 3 A. Relationship between the number of invasive alien species and the number of inhabitants (2012 population census) in the 19 surveyed islands (Spearman test, P-value = 0.001407). B. Relationship between the number of invasive alien species and the plane transportation frequency (number of flights per week from Tahiti) in the 16 surveyed islands with a domestic airport.

most people agreeing with the invasiveness status did not make specific negative comments (e.g. for the three species of rats – *Rattus* spp.). To avoid this bias towards positive comments, future studies should explicitly ask participants for their inputs on the ecological and socio-economical impacts of the targeted invasive species.

One animal species, the common myna (*Acridotheres tristis*), has received only positive comments. This bird, first introduced to Tahiti in the early 1900s (Meyer, 2003) is indeed considered as a useful animal because it eats introduced wasps and ticks especially in the Leeward Islands of the Society archipelago (e.g. in Huahine), whereas it is subjected to an active control programme in Tahiti to protect the threatened endemic flycatcher *Pomarea nigra* (Monarchidae) (Blanvillain, et al., 2003). For vascular plants, the 29 invasive species were not considered as "noxious/harmful" in all the surveyed islands where they are present. There were many positive comments for ornamental plants or fruiting trees, especially in the most remote islands of the Austral and the Marquesas (Table 4).

It is interesting to note that the perceived status of invasive alien species differs from one archipelago to another, but also among islands in the same archipelago, such as the climbing liana *Passiflora maliformis* in the Austral Is because of its edible fruits or the large tree *Falcataria moluccana* in the Marquesas as a timber tree (Table 4). Both species are currently being controlled in areas of high conservation values in Tahiti.

DISCUSSION

Island invasibility, species invasiveness and socio-economic development

Perception of invasiveness is complex because of diverse mental representations by different key interest groups and socio-economic contexts (Garcia-Llorente, et al., 2008). An understanding of human dimensions is necessary to avoid potential social conflicts in invasive species management (Estévez, et al., 2014; Russell, 2014).

Our results conducted on small islands of French Polynesia show that the number of invasive alien species is not decreasing with island remoteness (i.e. distance from Tahiti) and island size, but is more correlated with human development (e.g. the number of inhabitants and the frequency of transportation connection with Tahiti) and habitat diversity, as documented in other islands elsewhere (Kueffer, et al., 2010). The island of Tahiti can be considered as a "transportation hub" in the South Pacific, with an international airport opened in 1960 and direct flight connections to Rarotonga (Cook Is), Australia, New Zealand, New Caledonia, California and Hawaii (USA), Chile and Japan; and a major trade port in 1962 with goods imported from Europe, North and South America and South-east Asia. The increasing development of commercial trade during the past decades (from 330,000 tons in 1989 to 980,000 tons in 2015, ISPF, 2016) was associated with a dramatic increase of accidental plant and animal introductions. Invasive insects such as fruit flies (*Bactrocera* spp., Tephrididae), the glassy-winged sharpshooter (*Homalodisca vitripennis,* Cicadellidae) and the little fire ant (*Wasmannia auropunctata,* Formicidae), first introduced to Tahiti between the 1970s and the 1990s (Meyer, 2003), have subsequently spread to many other

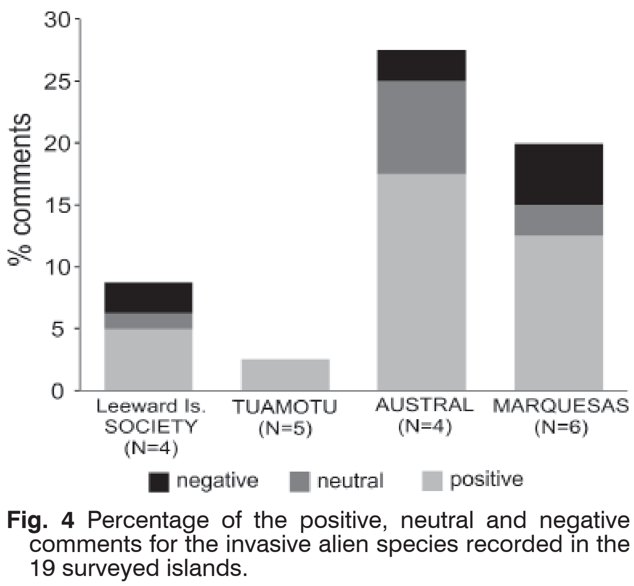

Fig. 4 Percentage of the positive, neutral and negative comments for the invasive alien species recorded in the 19 surveyed islands.

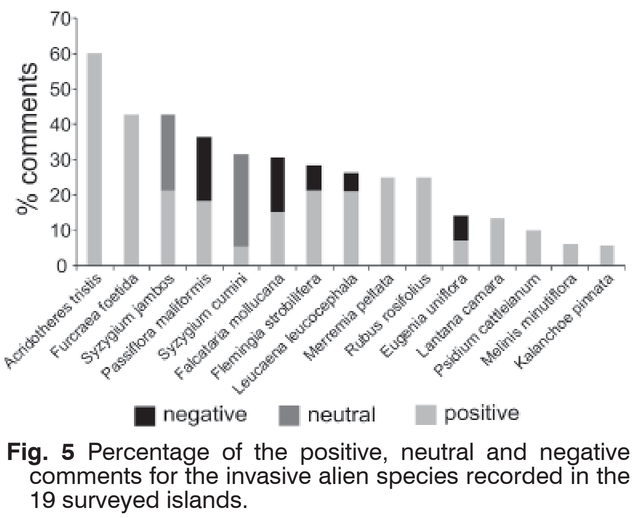

Fig. 5 Percentage of the positive, neutral and negative comments for the invasive alien species recorded in the 19 surveyed islands.

Table 4 Examples of positive and negative comments for some invasive alien plants introduced by Europeans in the surveyed islands with their date of first introduction or record in Tahiti, French Polynesia (Baas Becking, 1950; Jacquier, 1960).

Scientific name	Positive comment(s)	Island(s)	Negative comment(s)	Island(s)	Date of first introduction or record
Eugenia uniflora	Edible fruits, wood used for fish tools	Rimatara, Tubuai	Alters feral goat meat	Fatu Iva	1848
Falcataria (syn. *Albizia*) *moluccana*	Honeybee-forage plant, wood used for boats	Fatu Iva, Raivavae	Dries out rivers	Rurutu	1936
Flemingia strobilifera	Flower used in necklaces	Nuku Hiva, Ua Huka, Rimatara	Spreads in gardens	Tahuata	1937
Furcraea foetida	Formerly used for ropes & traditionnal dance skirts	Rimatara, Rurutu, Tubuai	-	-	?
Lantana camara	Ornamental garden plant	Nuku Hiva, Ua Huka	-	-	1853
Leucaena leucocephala	Forage for cattle, improves soil erosion control	Nuku Hiva Ua Huka	-	-	1845
Passiflora maliformis	Edible fruits used for jams	Rimatara	Suppresses orange and coffee trees	Fatu Iva, Tubuai	?
Syzygium cumini	Edible fruits	Tikehau, Makatea	-	-	1880
Syzygium jambos	Edible fruits	Tubuai			1890

French Polynesian islands through inter-island boat and/or plane transportation.

The perceived status of the 46 legally declared invasive species, a small subset of the total number of invasive species in French Polynesia (e.g. with more than 80 plants considered as invasive, Fourdrigniez & Meyer, 2008), differs from one archipelago to another, or even among islands in the same archipelago. They are more positively considered in the most isolated islands with lower socio-economic development and/or where natural resources are extremely limited, e.g. in atolls where invasive woody plants are used as tools or for wood construction, such as *Leucaena leucocephala*. This is very similar to the different attitudes of urban *versus* rural residents to pest species management in western developed countries or in Australia and New Zealand in the Pacific region (Fraser, 2001; Johnston & Marks, 1997). When abundant, invasive alien species are often seen as potential natural resources by islanders whereas when they are less common or rare, people agreed to eradicate introduced species. Species prioritisation that includes socio-economic values may thus contribute to a better efficiency in control or eradication by gaining support of local communities in remote islands.

Importance of cultural values

Human perceptions and attitudes vary with time, places, societies, economic conditions and culture (Dalla Bernardina, 2010; Fitzgerald, et al., 2007). The importance of cultural (traditional or ancestral) values of introduced species in the Pacific islands is well illustrated by animal species that were introduced by the first humans during their migration and colonisation, and became invasive with time, with sometimes dramatic impacts on the native biodiversity. Feral pigs (*Sus scrofa*) are still a source of dispute between conservationists and native Hawaiians who hunt them as in the past (Van Driesche & Simberloff, 2016), and Pacific rats (*Rattus exulans*) are considered a treasure brought to New Zealand by their Maori ancestors, thus may be worshipped and of high significance (Haami, 1994; Veitch & Clout, 2000). Some plants introduced by the first Polynesians for ritual, aesthetic or utilitarian values (Whistler, 2009) have also spread into native lowland forests in French Polynesia and Hawaii, including the candlenut tree (*Aleurites moluccana,* Euphorbiaceae) and the bamboo (*Schizostachyum glaucifolium,* Poaceae) where they are considered as either invasive (Smith, 1985) or part of the Polynesian social heritage (Larrue, et al., 2010).

Our survey indicates that the date of species introduction in the islands of French Polynesia, more particularly in Tahiti (Baas Becking, 1950; Jacquier, 1960), seems to be an important factor explaining attitudinal differences, as old introduced species seem to be more widely accepted or positively considered by people, because of their long co-existence (more than one century). This is the case of the small tree *Leucaena leucocephala* and the shrub *Lantana camara* which were introduced by Europeans in French Polynesia in 1845 as a fodder plant and 1853 as an ornamental garden plant respectively, and often still considered as beneficial species (Table 4). This phenomenon is sometimes, but incorrectly, called “indigenisation”, as these naturalised species (“naturalisation” is defined as an ecological proces where the alien plant species establishes and becomes incorporated within the natural flora, Richardson, et al., 2000) are not becoming indigenous or native but part of the human culture or natural heritage. It should be refered to as “heritagisation” (“patrimonialisation” in French) which describes a socio-cultural, legal or political process where an area, a good or a species is transformed into an object of the natural, cultural or religious heritage with conservation or restoration value.

One of the crucial challenges in invasive species management is the active involvement, engagement and support of local communities (Hart & Larson, 2014), as well as resolving or at least avoiding potential conflicts of interest between different stakeholders. The small Pacific islands, including French Polynesia, provide an excellent ground for testing new methodologies and initiatives in complex insular societies. Based on the results of this survey, we propose that an “invasive species perception index" should be included in feasibility studies to manage biological invasions in isolated inhabited islands.

A first step to integrate the local socio-economic and cultural dimensions of invasive species in the islands of French Polynesia was the creation of a network during and following this survey (called "Te Rau Mata Arai" in Tahitian, literally the "numerous watchful eyes"). Its aims are the prevention, detection, surveillance and control of invasive alien species by identifiying local, key people in each island (a total of 36 on the 19 surveyed islands) including local government and city council representatives, members of nature protection groups, small entrepreneurs, and other civil society actors.

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REFERENCES

Baas Becking, L.G.M. (1950). *Liste Préliminaire de Plantes Introduites à Tahiti*. Nouméa, Nouvelle-Calédonie: Commission du Pacifique Sud, Document Technique No. 7.

Bhattacharya, A. and Kumar, P. (2010). 'Water hyacinth as a potential biofuel crop'. *Electronic Journal of Environmental, Agricultural and Food Chemistry* 9(1): 112–122.

Blanvillain, C., Salducci, J.-M., Tuturai, G. and Maeura, M. (2003). 'Impacts of introduced birds on the recovery of the Tahiti flycatcher (*Pomarea nigra*)'. *Biological Conservation* 109(2): 197–205.

Bremner, A. and Park, K. (2007). 'Public attitudes to the management of invasive non-native species in Scotland'. *Biological Conservation* 139: 306–314.

Dalla Bernardina, S. (2010). 'Les Invasions Biologiques sous le Regard des Sciences de l'Homme'. In: R. Barbault and M. Atramentowicz (coord.) *Les Invasions Biologiques, une Question de Natures et de Sociétés*, pp. 65–108. Versailles, France: Editions Quae.

Estévez, R.A., Anderson, C.B., Pizzaro, J.C. and Burgman, M.A. (2014). 'Claryfying values, risk perceptions, and attitudes to resolve or avoid social conflicts in invasive species management'. *Conservation Biology* 29(1): 19–30.

Fischer, A., Selge, S., van der Wal, R. and Larson, B.M.H. (2014). 'The public and professionals reason similarly about the management of non-native invasive species: A quantitative investigation of the relationship between beliefs and attitudes'. *PLOS One* 9(8): e105495.

Fitzgerald, G., Fitzgerald, N. and Davidson, C. (2007). *Public Attitudes Towards Invasive Animals and Their Impacts*. Canberra, Australia: Invasive Animals Cooperative Research Centre.

Fourdrigniez, M. and Meyer, J.-Y. (2008). *Liste et Caractéristiques des Plantes Introduites Naturalisées et Envahissantes en Polynésie Française*. Papeete, Tahiti: Contribution à la Biodiversité de Polynésie française No. 17, Délégation à la Recherche, Papeete (Unpublished report).

Fourdrigniez, M., Taputuarai, R., Vivier, R. and Homo, D. (2014). *Guide de Protection des Îles contre les Espèces Envahissantes en Polynésie française*. Papeete, Tahiti: STP Multipress.

Fraser, W. (2001). *Introduced Wildlife in New Zealand: A Survey of General Public Views*. Landcare Research Science Series No. 23. Canterbury, New Zealand: Manaaki Whenua Press.

Garcia-Lorente, M., Martin-Lopez, B., Gonzalez, J.A., Alcorlo, P. and Montes, C. (2008). 'Social perceptions of the impacts and benefits of invasive alien species: Implications for management'. *Biological Conservation* 141: 2969–2983.

Haami, B. (1994). 'The Kiore Rat in Aotearoa: a Maori Perspective'. In: J. Morrison, P. Geraghty and L. Crowl (eds.) *Fauna, Flora, Food and Medecine, Science of Pacific Island People, Volume 3*, pp. 65–76. Suva, Fidji: Institute of Pacific Studies.

Hart, P.S. and Larson, B.M.H. (2014). 'Communicating about invasive species: How "driver" and "passenger" model influence public willingness to take action?' *Conservation Letters* 7(6): 545–552.

Hill, R.L. (1989). 'Environmental Protection Procedures and the Biological Control Programme against Gorse in New Zealand'. In: E.S. Delfosse (ed.) *Proceedings of the VIIth International Symposium on Biological Control of Weeds, 6–11 March 1988*, pp. 127–133. Rome, Italy.

IEOM (Institut d'Emission d'Outre-Mer) (2017). *Polynésie française. Rapport Annuel 2016.* Papeete, Tahiti: STP Multipress.

ISPF (Institut de la Statistique de la Polynésie Française) (2016). *Polynésie Française en Bref 2016. French Polynesia at a Glance* <www.ispf.pf> Accessed 10 June 2017.

Jacquier, H. (1960). 'Enumération des plantes introduites à Tahiti depuis la découverte jusqu'en 1885'. *Bulletin de la Société des Etudes Océaniennes* 130: 117–146.

Johnston, M.J. and Marks, C.A. (1997). *Attitudinal Survey on Vertebrate Pest Management in Victoria*. Melbourne, Australia: Report Series No. 3.

Kueffer, C., Daehler, C.C., Torres-Santana, C.W., Lavergne, C., Meyer, J.-Y., Otto, R. and Silva, L. (2010). 'A global comparison of plant invasions on oceanic islands'. *Perspectives in Plant Ecology, Evolution and Systematics* 12: 145–161.

Larrue, S., Meyer, J.-Y. and Chiron, T. (2010). 'Anthropogenic vegetation contributions to Polynesia's social heritage: the legacy of candlenut tree (*Aleurites moluccana*) forests and bamboo (*Schizostachyum glaucifolium*) groves on the island of Tahiti'. *Economic Botany* 64(4): 329–339.

Lorence, D.H., Wood, K.R., Perlman, S.P. and Meyer, J.-Y. (2016). 'Flore Vasculaire et Végétation des Îles Marquises: Caractéristiques, Originalités et Vulnérabilité'. In: R. Galzin, S.-D. Duron and J.-Y. Meyer (eds.) *Biodiversité terrestre et marine des îles Marquises, Polynésie française*, pp. 311–336. Paris, France: Société Française d'Ichtyologie.

Lowe, S., Browne, M., Boudjelas, S., and De Poorter, M. (2000). *100 of the World's Worst Invasive Alien Species. A Selection from the Global Invasive Species Database*. The Invasive Specialist Group (ISSG) of the Species Survival Commission (SSC) of the World Conservation Union. IUCN.

Marshall, N.A., Friedel, M., van Klinken, R.D. and Grice, A.C. (2011). 'Considering the social dimension of invasive species: The case of buffel grass'. *Environmental Management* 14: 327–338.

McNeely, J.A. (ed.). (2000). *The Great Reshuffling: Human Dimensions of Invasive Alien Species*. Gland, Switzerland: IUCN.

Meyer, J.-Y. (2003). 'French Polynesia'. In: C. Shine, J.K. Reaser and A.T. Gutierrez (eds.) *Invasive Alien Species in the Austral-Pacific Region*, pp. 22–34. Cape Town, South Africa: National Reports and Directory of Resources. The Global Invasive Species Programme.

Richardson, D.M., Pysek, P., Rejmanek, M., Barbour, M.G., Panetta, F.D. and West, C.J. (2000). 'Naturalization and invasion by alien plants: Concepts and definitions'. *Diversity and Distribution* 6: 93–107.

Russell, J.C. (2014). 'A comparison of attitudes towards introduced wildlife in New Zealand in 1994 and 2012'. *Journal of the Royal Society of New Zealand* 44(4): 136–151.

Selge, S., Fischer, A. and van der Wal, R. (2011). 'Public and professional views on invasive non-native species – a qualitative social scientific investigation'. *Biological Conservation* 144(12): 3089–3097.

Smith, C.W. (1985). 'Impacts of Alien Plants on Hawaii's Native Biota'. In: C.P. Stone and J.J. Scott (eds.) *Hawaii's Terrestrial Ecosystems: Preservation and Management*, pp. 180–250. Honolulu, Hawaii, USA: University of Hawaii Press.

Van Driesche, R.G. and Simberloff, D. (2016). 'Benefit-Risk Assessment of Biological Control in Wildlands'. In: R.G. Van Driesche, D. Simberloff, B. Blossey, C. Causton, M.S. Hoddle, D.L. Wagner, C.O. Marks, K.M. Heinz and K.D. Warne (eds.) *Integrating Biological Control into Conservation Practice*, pp. 80–104. Chichester, UK: John Wiley and Sons, Ltd.

Veitch, C.R. and Clout, M.N. (2000). 'Human dimensions in the management of invasive species in New Zealand'. In: J.A. McNeely (ed.) 2000. *The Great Reshuffling: Human Dimensions of Invasive Alien Species*, pp. 63–71. Gland, Switzerland: IUCN.

Warner K.D. and Kinslow, F. (2013). 'Manipulating risk communication: Value predispositions shape public understanding of invasive species science in Hawaii'. *Public Understanding of Science* 22: 203–216.

Whistler, W.A. (2009). *Plants of the Canoe People: An Ethnobotanical Voyage through Polynesia*. Honolulu, Hawaii, USA: Island Press.

Married bliss and shotgun weddings: effective partnerships for island restoration

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Abstract Island restoration is expanding as a tool for enhancing conservation outcomes. The ability of conservation managers to eradicate multiple invasive species over increasingly complex or large areas is steadily improving, but progress inevitably presents new challenges. There is always a larger, even more complex operation ahead, and islands with human populations present their own suite of problems and opportunities. The majority of the large and/or complex island restoration projects to-date have been carried out with a high level of government commitment including funding support (e.g. New Zealand, Australia, USA). However, there is increasing interest in applying this methodology to islands in jurisdictions where there is less central government support. This can be further complicated by regulatory systems and implementation logistics. Non-governmental organisations are now taking lead roles in many projects to restore islands worldwide, working collaboratively to share the financial, logistical and regulatory challenges and share in the outcomes. If we are to succeed in truly "turning the tide" on invasive species it will be necessary for governmental and non-governmental organisations to partner even more effectively in order to expand the capacity for such conservation actions worldwide. Choosing the right partners, clarifying shared values, programme goals, responsibilities and definitions of success is needed for NGOs, governments and other partners to work effectively and make the progress that is necessary to continue achieving good conservation outcomes in the future.

Keywords: collaboration, eradication, organisational structure, planning, project management

INTRODUCTION

"Cross sector partnerships ... are far from commonplace. True partnerships are the stuff of legends. Think of the Fellowship of the Ring..." Tennyson (2011).

The eradication of invasive vertebrates, especially rodents, as a key component of island restoration has an extensive history spanning more than 50 years (e.g. Towns & Broome, 2003; Howald, et al., 2007; Towns, et al., 2013; Russell & Broome, 2016). Techniques for carrying out this work have developed over this time, and eradication projects are now often highly complex and specialised operations using equipment and people from all over the world.

Until the last fifteen years, the agencies with the resources to undertake the largest projects were generally government conservation agencies (GCAs) in developed countries such as New Zealand, Australia, the USA and Canada. Projects included the consultation of stakeholders, but early operations were often led and managed by single organisations on government land (Towns & Broome, 2003).

As the case for carrying out island restoration projects has become more established (Courchamp, et al., 2003; Bellingham, et al., 2010; Jones, et al., 2016), new organisations, especially non-governmental organisations (NGOs) and for-profit enterprises (FPEs) have become increasingly involved in island restoration. There are now several NGOs and initiatives worldwide that are entirely dedicated to the restoration of islands through the removal of invasive species (e.g. Island Conservation (USA); Grupo de Ecología y Conservación de Islas (Mexico); Predator Free New Zealand (New Zealand)), and this work is gaining in prominence within the wider conservation NGO community (e.g. recent work by the Royal Society for the Protection of Birds (UK) and the South Georgia Heritage Trust). In addition, new commitments to carry out this work have been made by international organisations and through international agencies, e.g. the International Union for the Conservation of Nature (IUCN)'s Honolulu Challenge (IUCN, 2016), BirdLife International's Invasive Species Programme (BirdLife International, 2017).

The increasing challenge of partnerships

In order to continue to obtain the conservation benefits available from the eradication of invasive species on islands, projects will need to be carried out in even more complex conditions. In light of current technology and experience, we could describe a project where area is less than 10,000 ha; there is a single owner; country jurisdiction is clear; and a single funding source is available as "simple". In some countries all, or the majority of these projects have now been tackled, or the need for them has not arisen (Howald, et al., 2007; Dawson, et al., 2015; Parkes, et al., 2017; Stanbury, et al., 2017). In less developed countries where conservation funding is much scarcer, the idea of national governments supporting island restoration projects is often not well established and finding funds for any project of this sort is difficult. In many countries, islands without human habitation or regular use are extremely uncommon.

Islands with significant human populations, complex and challenging topography, and/or located in extremely remote parts of the world are thus becoming a higher priority (Oppel, et al., 2011, Dawson, et al., 2015; Parkes, et al., 2017; Stanbury, et al., 2017). For example, the Royal Society for the Protection of Birds (RSPB) is currently developing a project to restore Gough Island (Tristan da Cunha) through the eradication of house mice (*Mus musculus*).

The island is extremely remote and is located in a UK Overseas Territory with a small human population and insufficient financial resources to support the operation. The project partnership will include no fewer than six project partners from at least three countries, including government agencies, NGOs, and the local community.

The DIISE (2017) records 25 rodent eradication attempts that have been made on islands greater than one square kilometre in area since 2010. Of these, the majority (15 of the 25) were not undertaken solely by government agencies, and even where operations were government-led, some sort of partnership was needed (e.g. between State and Federal government agencies).

Howald, et al. (2011) explored the advantages and challenges of different organisational structures conducting island restoration projects. The authors found that there were clear advantages and disadvantages attached to GCAs, NGOs and FPEs in conducting eradication campaigns, but concluded that the potential advantages of collaboration were often greater than the challenges. In this paper, we consider local community groups separately from NGOs as another type of organisation which is increasingly proposing and supporting new island restoration projects. As well as type, the size and culture of organisations also has significant importance and impacts on internal bureaucracy, speed of decision-making and level of tolerance for risk.

Whilst it is apparent that partnerships provide opportunities to capitalise on the strengths and compensate for the weaknesses of different types of organisations, partnerships can be complicated to establish and maintain. The same people who have significant strengths and experience in designing and implementing island eradication projects do not always have a similar level of experience or expertise in developing or maintaining organisational partnerships, especially when organisations' cultural aspects can be highly variable. Staff turnover can be an issue, as partnerships are effectively formed between individuals as well as organisations, and some organisations have higher turnover than others. It is important for the organisations that are planning and managing eradication projects to recognise the importance of consistency of staffing in these projects, and endeavour to provide this, as well as supporting training in partnership-working for technical staff wherever possible.

This paper assesses some of the factors that may be influential in making partnerships work. There is no way to carry out a scientific analysis of how to create a strong partnership that will lead to a successful project outcome: partnerships (like marriages) are not a scientific construct. However, the authors of this paper have been involved in a wide range of projects with partners from government, NGOs, and local communities. From our combined experience, the main elements needed, in our opinion, are presented, along with some of the common pitfalls. Sharing our experiences may enable other project managers to analyse their own potential partnerships, and hopefully use these principles to enhance the likelihood of project success.

What partnerships are, and when they should be established

According to Wilcox (1998) a partnership is an agreement between two or more individuals or groups to work together to achieve common aims. Sterne, et al. (2009) identified nineteen characteristics of partnerships, including mutual trust and respect, clearly identified roles and responsibilities, transparency of decision-making, and a process for adjudicating disputes. The Nature Conservancy (2017) suggests there are six stages to most partnerships, and Tennyson (2011) defines twelve stages. The main points are:

- Prepare: define the need for partners.
- Select: choose the best partner(s) to work with.
- Negotiate: create agreement to inspire action and reduce the potential for conflict.
- Manage: implement joint work.
- Measure: monitor and evaluate the partnership.
- Conclude: adapt, improve or conclude the partnership.

During the first stage, it is important for project managers to consider carefully whether forming a partnership is the best choice in their individual situation. Reasons to establish a project partnership include the desire to increase capacity amongst other organisations and stakeholders; the need to access a new decision-making authority or constituents; the opportunity to share costs; and the ability to make projects more sustainable and resilient. One method of comparing these potential benefits with the potential costs would be a SWOT (Strengths, Weaknesses, Opportunities, Threats) analysis.

It is advisable to establish partnerships early in the project planning process, if indeed it is considered that a partnership would be beneficial. The Nature Conservancy (TNC) recommends a partner scoping exercise at the start of planning any conservation project (The Nature Conservancy, 2017). However, this step is not currently included in resources specific to island restoration, such as the Pacific Invasives Initiative (PII) Resource Kit (Pacific Invasives Initiative, 2011). In particular, there appear to be significant benefits from involving community partners, including landowners, at the initial stages of project planning (Varnham, 2011; McClelland, et al., 2011), not least because their local knowledge can add value to planning and their drive to succeed can assist in motivating the rest of the partnership.

Potential partners could be identified in a stakeholder analysis which may be carried out as part of the feasibility study for an operation, e.g. step 2.1 of the PII Resource Kit (Pacific Invasives Initiative, 2011), or through a scoping exercise (The Nature Conservancy, 2017; Flora and Fauna International, 2009). However, it is likely that if external funds are to be sought for an eradication, partners may need to be identified even sooner than this. As part of considering the composition of a partnership, it can be useful to consider the implications of excluding a particular organisation or group and how this could affect the outcome and the other partners. For example, excluding local people from a partnership could lead to mistrust from funders and external agencies as well as the community themselves, or even prevent the project from going ahead.

Often, there may be little or no choice about who to work with, for example it is often necessary to work with a local government agency, or the island owner. In some cases, they could be reluctant partners at first, but may become more engaged when they see the benefits of the relationship. This engagement could take a long time to achieve, and in some cases may never be possible. In other cases, partners may be willing, but there may be high costs connected to their involvement. Thinking ahead about the costs and benefits will help in considering whether a partnership is appropriate, and in minimising the costs and maximising the benefits (Flora and Fauna International, 2009).

The level of intensity of partnership that is desired should also be considered. Some partnerships are short-term, and relatively informal relationships, whereas others may develop into strategic long-term, organisation-wide relationships. It is also possible for any partnership to break down before its objectives have been achieved. A plan for how to deal with such a break should be included in the partnership agreement or memorandum of understanding (see discussion below).

Key elements of strong partnerships

Although there is a diverse range of organisations involved in island restoration work worldwide, the elements required for partnerships are the same. It has been suggested that the key principles of equity, transparency, and mutual benefit should apply to most partnerships (Tennyson, 2011). These principles provide a foundation on which the partnership can operate. In order to build this foundation, those forming new partnerships should consider the following in particular: good communication; clearly defined roles; appropriate leadership, staffing and personalities; clear, shared vision and expectations; and clarity over funding and resource issues (Wilson, 2005; Tennyson, 2011; Ozarski, 2015).

One basic tool that most partnerships use is a written partnership agreement (also referred to in some cases as a memorandum of understanding or memorandum of partnership). Partnership agreements can vary widely in their level of detail depending on the complexity and aims of the partnership and the degree of formality but should, at a bare minimum, define the boundaries of the partnership. Partnership agreements should cover the areas set out below and may include others, depending on the specific needs of the project.

As previously noted, one of the main characteristics of partnerships is that the partners are working towards a common aim. Where a group or individual has an interest in delivery of an island restoration project but is not as committed to the same goal as others, a partnership should not be formed – this could lead to confusion and frustration. The project team should seek to maintain a good relationship with such stakeholders but should not force an inappropriate partnership.

Good communication

When partnerships do not work, poor communication is often blamed. Effective communication is essential to move projects forward, especially in partnerships where partners may have different motivations and expectations. Setting out a shared communication strategy is recommended within partnerships. General principles of communication for partnerships (after The Nature Conservancy, 2017) include:

- be timely in communication;
- brainstorm new issues;
- be consultative, not dictatorial;
- be flexible;
- document agreements and plans, and revisit, adjust and adapt as the situation changes;
- a policy of "no surprises".

In addition to these principles, it is important to respect cultural and organisational differences and challenges when communicating. Communication methods should be adapted to suit each partner organisation's strengths and weaknesses, e.g. emailing high resolution newsletters to communities with limited internet access is not effective communication.

Partnerships may be formed for many reasons. These may include the development of fundraising support, advocacy, avoiding bureaucracy, the need for landowner and resident buy-in and support, resource sharing, research, provision of expertise, and to enable different organisations to gain project experience, perhaps building towards their own projects in the future. All of these reasons are legitimate; however, it is important that all partners are clear about their own and other partners' motivation, and the scope of each partner's involvement. It is only ongoing communication that will enable this clarity. At times, some partners may also need to operate transparently outside the scope of a partnership, for example, government agencies which may also have a regulatory role. If the scope of each partnership has been clearly established and communication is clear, this should be possible.

It is important to include positives in communication. Even though project planning and implementation is challenging, project teams benefit from celebrating successes, recognising achievements, and saying "thanks". It is important to ensure dispersed partners are all able to take part in celebrations and reflect on the achievements and progress being made.

It is also important to consider the way in which a project will communicate itself externally. Publicity and "branding" can often be stumbling blocks in partnerships, and many projects have developed their own brands, independent of the partner organisations (e.g. the Isles of Scilly Seabird Recovery Project). Early decisions on shared messages and how to acknowledge partners and supporters can help to avoid issues later on.

Clearly defined roles and responsibilities

It is extremely important that the partners in any island restoration project understand their respective roles and what is expected of them. The roles and responsibilities of different organisations will vary over the course of the project and it is important that all partners understand how their roles and those of other partners will change over time. This is particularly important in the planning and post-operational phases of the project where roles may be less obvious.

Communities, local and non-local NGOs, and local and non-local government agencies may all have a role. In federal systems, it may also be necessary to involve different levels of government (e.g. the Macquarie Island pest eradication project involved both the Tasmanian State Government and the Australian Federal Government, or in the UK Overseas Territories where local and UK governments may play a role). In the Macquarie Island pest eradication project, the funding agreement between the Tasmanian and Australian Federal governments outlined that funding was joint, but that implementation was a Tasmanian government responsibility. Based on that agreement there was no confusion over operational roles. Without such clarity, multiple partners and stakeholders may perceive themselves to have decision-making authority leading to confusion and potentially to operational difficulties.

In order to minimise this sort of confusion, most projects develop some sort of partnership agreement or Memorandum of Understanding. This may be legally binding in some partnerships. However, even if roles are clearly set out in writing, it should not be assumed that all project partner staff who are participating in meetings or in project teams are aware of these roles, and they may need to be reiterated and revisited many times. Templates for developing project partnership agreements are available from Flora and Fauna International (2009), The Nature Conservancy (2017) and in the Partnering toolbook (Tennyson, 2011). None of these are specific to island restoration projects, and it would be useful if practitioners could develop and share resources in this area in the future.

Sometimes the project plan can be effective as a partnership agreement. All projects should have a clear plan which clearly describes the agreed roles and responsibilities of all partners. A good reason to have a partnership agreement is when a partnership is likely to go beyond the scope of a single project.

It is also important to set out the roles and responsibilities for the various advisory and steering groups that will be developed within most partnership projects, each of which may involve a subset of the project partners. Terms of reference for these groups have been developed by many projects but, as with partnership agreements, they are not yet generally shared. Developing such a resource would be of great use to future projects, and this could be hosted on websites such as those of the Pacific Invasives Initiative (section now in development) or the Great Britain Non-native Species Secretariat.

Leadership, staffing and personalities

As with any endeavour, the people involved are a major factor that will lead to the work being enjoyable and effective, and the success of any island eradication project will largely depend on the skills, dedication, and attitude of the team involved. If a project is being carried out by a single agency, then that agency can recruit a team made up of the most suitable/experienced people available and although there may be some personality issues during the project implementation period, it should be possible to manage these as part of normal business practices.

However, when working in partnership, there can be pressure to include representatives from different organisations in teams despite substantial differences in experience and culture. In addition, the representatives of the partner organisations who should meet regularly to discuss project planning and implementation may have differing views and experiences. If one organisation/individual has been leading/driving the project for some time prior to implementation, they may have particularly strong opinions on how things should be done, and this could lead to clashes with others.

Personality clashes and power struggles can derail an operation, and lead to breakdowns in organisational relationships if left unaddressed. Whilst large organisations may have the capacity to replace staff that are causing difficulties within a partnership, smaller organisations may not have this ability. Individuals who may be skilled at motivating others and providing leadership within a project may not always be best suited to developing and maintaining complex project partnerships and vice versa.

Most partnerships require more management time than anticipated. If a particularly wide partnership is necessary, it may be necessary to bring in new staff whose job is primarily to service the partnership in terms of communications and logistics. Organisations that are planning large projects with complex partnerships should consider recruiting personnel who have skills and experience in this area and can complement the technical skills of the operational management team.

If possible, organisations partnering in island restoration projects should aim to involve more than one staff member in each project so that there is a chance to review decisions and assess how the partnership is developing, and to assist continuity in case of staff turnover. If problems arise, each partner organisation should have a clear understanding of how they can raise concerns and address them at an early enough stage to avoid a complete breakdown of the partnership and potentially of the whole project.

Clear, shared expectations

As discussed above, organisations may enter into partnerships for a variety of reasons and with a variety of expected benefits. Additionally, motivations for wanting to be involved may be very different, even if the ultimate goal is shared. At the start of the partnership relationship, organisations should work to establish their shared goals and vision for the work to be undertaken. They should develop a project plan or agree the process that will be used to develop this. They should set out guidelines for decision-making and what will happen in the event of disagreements. It is also helpful to agree on a formal grievance process before a dispute emerges.

One area where there seems to be particular potential for a mis-match of expectations is in relation to pre-eradication preparation. It is important to make clear plans regarding who will make an island "ready" for eradication, e.g. track cutting, removal of waste, informing residents, leading on any research, etc., as well as who will fund this work. Sometimes preparation can take a long time, and it can be difficult maintaining enthusiasm and energy throughout this phase.

There must also be clear expectations about how the partnership will move on, in the event of either success or failure of the planned eradication operation. Partners should be clear on: who will assume responsibility for reviewing the project, and for a repeat attempt if necessary; how failure will be communicated and who will lead on this and whether the partnership will be expected to remain in place until a repeat operation is planned and concluded. It is good practice to build in a review point for partnerships at a key milestone (e.g. once an eradication operation is completed) to assess how well the partnership has progressed, whether all partners have met their commitments, etc. The outputs of this process could inform the organisations if they are considering extending the partnership to cover further projects.

Project plans for eradication projects often conclude two years after on-island operations cease, or when the island is declared officially 'pest-free'. However, in many situations, site managers or residents will need to remain engaged with projects in the longer-term, for monitoring, biosecurity, or to begin more intensive restoration efforts such as the reintroduction of threatened species. Partners should make their plans clear as soon as they can, as if some partners plan to withdraw from working on the island post-eradication, it may be necessary to introduce new partners to assist in post-project site management. In particular, it is extremely important to be clear about who will be responsible for maintaining monitoring and biosecurity arrangements after the eradication project is complete, and who will respond in the event of a pest incursion (either of the pest that was eradicated, or something entirely new).

Clarity over funding and resource issues

It is good practice for partners to share information on their planned contribution to a project, including cash and non-cash (in-kind) contributions. Some contributions may be invisible to those partners who are not directly involved. For example, a partner who is taking the lead on drafting legal contracts may be spending time and money on expensive services, but another partner may be completely unaware of this activity. Project budgets should ideally take account of in-kind contributions of time from all project partners, as well as cash contributions from donors. A clear project plan (as discussed above), and a designated project manager are important to ensure all partners know what contribution they are expected to make. Clear governance of a partnership is also important so that the project manager is given clear accountability and responsibility for deliverables and is not told to do different things by different partners.

Joint fundraising can be a problematic and difficult area for many partnerships. Partnerships should aim to develop a collaborative fundraising agenda with mutually agreed messaging. There should be clarity over which partners have responsibility for donor cultivation and management, and how donors will be managed after the project has concluded to avoid perceptions of "donor poaching" (The Nature Conservancy, 2017).

Partners need to be clear on what financial disbursement or opportunities they expect from the project. For example, if local communities wish for project supplies to be purchased from local outlets, this should be flagged early so that any increased costs associated with this can be dealt with. If government agencies expect to charge fees to the project (or conversely will waive standard fees) these expectations should be raised and addressed early on to avoid budgetary shocks.

CONCLUSIONS

The days of single agencies implementing island restoration projects may be waning, as islands with human populations, mixed tenure, complex legal status and multiple stakeholders are now high on international priority lists for future operations (Oppel, et al., 2011; Dawson, et al., 2015; Stanbury, et al., 2017). Operational managers and organisations committing to carry out these operations in future need to be aware of the skills and the level of time needed to maintain partnerships, and the potential pitfalls. Recent experiences have served to illustrate that partnerships can deliver immense gains for conservation – without them, we would not have seen recent operations on Vahanga, Antipodes, Palmyra, Desecheo and the Isles of Scilly (as just a few examples).

Lessons from particular partnerships for island restoration projects may be captured in grey literature such as project review documents, but there is little openly accessible material available about best practice in this area. It would be very helpful to future project managers if review documents and templates for partnership agreements could be shared openly with others. Websites such as those for the Pacific Invasives Initiative or the Great Britain Non-Native Species Secretariat could usefully host this sort of information.

Although working in partnerships can be very challenging at times, it is apparent that island restoration will only continue to deliver benefits worldwide if its practitioners are able to work together and draw in new organisations. By considering the elements of partnerships early in the process, we hope that more operations will be “matches made in heaven” and a shotgun will seldom need to be drawn from the figurative cupboard.

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REFERENCES

Bellingham, P.J., Towns, D.R., Cameron, E.K., Davis, J.J., Wardle, D.A., Wilmshurst, J.M. and Mulder, C.P.H. (2010). ‘New Zealand island restoration: Seabirds, predators, and the importance of history’. *New Zealand Journal of Ecology* 34: 115–136.

BirdLife International. (2017).’ Invasive Alien Species’. <www.birdlife.org/worldwide/programme-additional-info/invasive-alien-species-programme-more-info>. Accessed 2 July 2017.

Courchamp, F., Chapuis, J.L. and Pascal, M. (2003). ‘Mammal invaders on islands: Impact, control and control impact’. *Biological Reviews* 78: 347–383.

Dawson, J., Oppel, S., Cuthbert, R.J., Holmes, N., Bird, J.P., Butchart, S.H.M., Spatz, D.R. and Tershy, B. (2015). ‘Prioritizing islands for the eradication of invasive vertebrates in the United Kingdom overseas territories’. *Conservation Biology* 29: 143–153.

DIISE. (2017). ‘The Database of Island Invasive Species Eradications’. Developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand, Auckland. <http://diise.islandconservation.org>. Accessed 26 June 2017.

Flora and Fauna International. (2009). *Guidance for Working with Other Organisations.* Report produced as part of the USAID-funded project: ‘Life on the Edge – Conserving Biodiversity and Rebuilding Livelihoods in the Wake of Natural Disaster and Human Conflict’. <www.fauna-flora.org/wp-content/uploads/Partnership_guidance_September2009.pdf>. Accessed 3 July 2017.

Howald, G., Donlan, C.J., Galván, J.P., Russell, J., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). ‘Invasive rodent eradication on islands’. *Conservation Biology* 21: 1258–1268.

Howald, G.R., Donlan, C.J., McClelland, P., Macdonald, N. and Campbell, K.J. (2011). ‘Advantages and challenges of government, non-profit and for-profit approaches to eradications: leveraging synergies by working together’. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 432–436. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

IUCN. (2016). ‘The Honolulu Challenge on Invasive Alien Species’. <www.iucn.org/theme/species/our-work/invasive-species/honolulu-challenge-invasive-alien-species>. Accessed 2 July 2017.

Jones, H.P., Holmes, N.D., Butchart, S.H.M., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Munoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P.E., Cuthbert, R.J., Ebbert, S., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R. and Croll, D.A. (2016). ‘Invasive mammal eradication on islands results in substantial conservation gains’. *Proceedings of the National Academy of Sciences of the United States of America 113.* 15: 4033–4038.

McClelland, P.J., Coote, R., Trow, M., Hutchins, P., Nevins, H.M., Adams, J., Newman, J. and Moller, H. (2011). ‘The Rakiura Titi Islands restoration project: community action to eradicate *Rattus rattus* and *Rattus exulans* for ecological restoration and cultural wellbeing’. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 451–454. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Oppel, S., Beaven, B.M., Bolton, M., Vickery, J. and Bodey, T.W. (2011). ‘Eradication of invasive mammals on islands inhabited by humans and domestic animals’. *Conservation Biology* 25: 232–240.

Ozarski, J. (2015). ‘Cooperation for Mutual Benefit: Opportunities for Primary Industry and the New Zealand Department of Conservation’. Fulbright New Zealand. <www.fulbright.org.nz/wp-content/uploads/2015/08/OZARSKI-Jill-final-report-new-PDF.pdf>. Accessed 2 July 2017.

Pacific Invasives Initiative. (2011). ‘Resource Kit for Rodent and Cat Eradication’. <www.pacificinvasivesinitiative.org/rk/index.html>. Accessed 2 July 2017.

Parkes, J.P., Byrom, A.E. and Edge K-A. (2017). ‘Eradicating mammals on New Zealand island reserves: What is left to do?’ *New Zealand Journal of Ecology* 41: 263–270.

Russell, J.C. and Broome, K.G. (2016). ‘Fifty years of rodent eradications in New Zealand: Another decade of advances’. *New Zealand Journal of Ecology* 40: 197–204.

Stanbury, A., Thomas, S., Aegerter, J., Brown, A., Bullock, D., Eaton, M., Lock, L., Luxmoore, R., Roy, S., Whitaker, S., and Oppel, S. (2017). ‘Prioritising islands in the United Kingdom and crown dependencies for the eradication of invasive alien vertebrates and rodent biosecurity’. *European Journal of Wildlife Research* 63: 31.

Sterne, R., Heaney, D. and Britton, B. (2009). ‘The Partnership Toolbox’. World Wildlife Fund. <http://assets.wwf.org.uk/downloads/wwf_parthershiptoolboxartweb.pdf>. Accessed 2 July 2017.

Tennyson, R. (2011). ‘The Partnering Toolbook.’ The Partnering Initiative. <https://thepartneringinitiative.org/publications/toolbook-series/the-partnering-toolbook/>. Accessed 22 April 2018.

The Nature Conservancy (TNC). (2017). ‘Partnering for Conservation’. <www.conservationgateway.org/ConservationPlanning/partnering/Pages/default.aspx>. Accessed 2 July 2017.

Towns, D.R. and Broome K.G. (2003). ‘From small Maria to massive Campbell: Forty years of rat eradications from New Zealand islands’. *New Zealand Journal of Zoology* 30: 377–398.

Towns, D.R., West, C.J., and Broome, K.G. (2013). ‘Purposes, outcomes and challenges of eradicating invasive mammals from New Zealand islands: An historical perspective’. *Wildlife Research*. 40(2): 94–107.

Varnham, K. (2011). ‘Involving the community in rodent eradication on Tristan da Cunha’. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 504–507. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Wilcox, D. (1998). ‘Guide to Development Trusts and Partnerships, The Partnerships Handbook’. <www.partnerships.org.uk>. Accessed 29 June 2017.

Wilson, C. (2005). ‘Developing Effective Partnerships Between the Department of Conservation and Community Groups’. *Science for Conservation* 248. Wellington, New Zealand: Department of Conservation.

Winning the hearts and minds – proceeding to implementation of the Lord Howe Island rodent eradication project: a case study

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Abstract Lord Howe Island (LHI) is a World Heritage listed volcanic remnant island, 600 km from mainland Australia. Home to many threatened endemic species, and an important nesting site for many migratory seabirds, LHI is also home to an established community of 350 people. The island economy relies heavily on tourism, based around the World Heritage landscapes, and terrestrial and marine biodiversity. The impact of rats and mice on the world heritage values of the island are well documented. A feasibility study in 2001 considered the eradication of both rats and mice on the Lord Howe Island Group (LHIG) in a single operation as feasible and achievable. Since then, numerous studies and community consultations have been undertaken; and the methodology, risks and benefits of an eradication effort have been carefully considered and evaluated. The project has long been topical in the community, with both ardent support and opposition. Though funding for the project was secured in 2012, implementation of the project has had repeated delays, and consequently, additional consultation efforts, studies and assessments have been required to address lingering community concerns. This paper describes a wide range of techniques that were used in an effort to gain community support and acceptance of the project, and ultimately, to enable its implementation. The paper also outlines those aspects of the social process that have led to greater support and those which led to less support and, in some cases, opposition. Within these lessons learnt, the paper presents some insights around how communities engage with and respond to scientific information and pest control (including perceived influence on livelihoods, conservation, health and the legacy to be left for future generations). These insights are explained in the context of psychological processes such as emotional responses (fear/trust) and personal values (economic, environmental, social). Given that the eradication of invasive predators on larger inhabited islands is the next logical step for island conservation, projects such as the LHI Rodent Eradication Project are of particular value as real life hard-earned lessons. We hope that these findings can help facilitate future successful island eradication programmes that work with the community.

Keywords: community, economic, engagement, human health, inhabited, tourism

INTRODUCTION

Purpose

The purpose of this paper is to present a case study of eradication planning on an inhabited island, illustrating the importance of social impact considerations and presenting some guiding principles and lessons learnt.

The island

Lord Howe Island (LHI) is located 570 km east of Australia (Fig. 1). It covers 1,455 ha, is 12 km long, and 1.0–2.8 km wide. The LHIG was listed as a World Heritage Area in 1982 and is located within the Lord Howe Island Marine Park (NSW).

LHI is part of the State of New South Wales and is administered by the Lord Howe Island Board (LHIB), which comprises four locally elected islanders and three ministerially appointed mainland members. The LHIB (and its administrative arm) are directly responsible for the care, control and management of the island's natural values and the affairs and trade of the island and carry out all local government functions on behalf of approximately 350 island residents.

The settlement area covers about 15% of the island (400 ha) and is used predominantly for residential, pastoral/agricultural and commercial purposes. Tourism is the most significant industry and major source of income on the island and is heavily focused around the world heritage values of both the marine and terrestrial environments (Lord Howe Island Tourism Association, 2015).

Current impacts of rodents on LHI

Ecological impacts

The devastating ecological impacts of introduced rodents on offshore islands around the world are well documented (Groombridge, 1992; Towns, et al., 2006; Jones, et al., 2008). Similar impacts on LHI have been observed since the arrival of mice (*Mus musculus*) in approximately the 1860s and ship rats (*Rattus rattus*) via a shipwreck in 1918. The Lord Howe Island Biodiversity Management Plan (DECC, 2007a) summarises the immediate and ongoing impact the introduction of rodents has had on flora and fauna species on LHI, including extinction of several species of birds, plants and invertebrates.

Socio-economic impacts

From the perspective of the human population, rats and mice are major domestic pests. They infest residences, destroy foodstuffs and vegetable gardens; and contaminate homes with excrement. They are also a known health risk to humans as they harbour and transmit diseases and parasites such leptospirosis and rat lungworm disease (Shiels, et al., 2014).

From an economic perspective, rats cause considerable economic loss to the island's kentia palm industry, with predation of seed as high as 30% (Parkes, et al., 2004) severely reducing seed production (Pickard, 1983; Billing, 1999; further detail in Wilkinson & Priddell, 2011).

Tourism, the LHIG's main industry, is based on the islands' unique biodiversity and World Heritage values. These values are significantly threatened by rodents (IUCN, 2017) therefore reducing the visitor experience offered by the island.

History of rodent control on LHI

Islanders and the LHIB have been involved in the control of rodents (rats and mice) on Lord Howe Island since about

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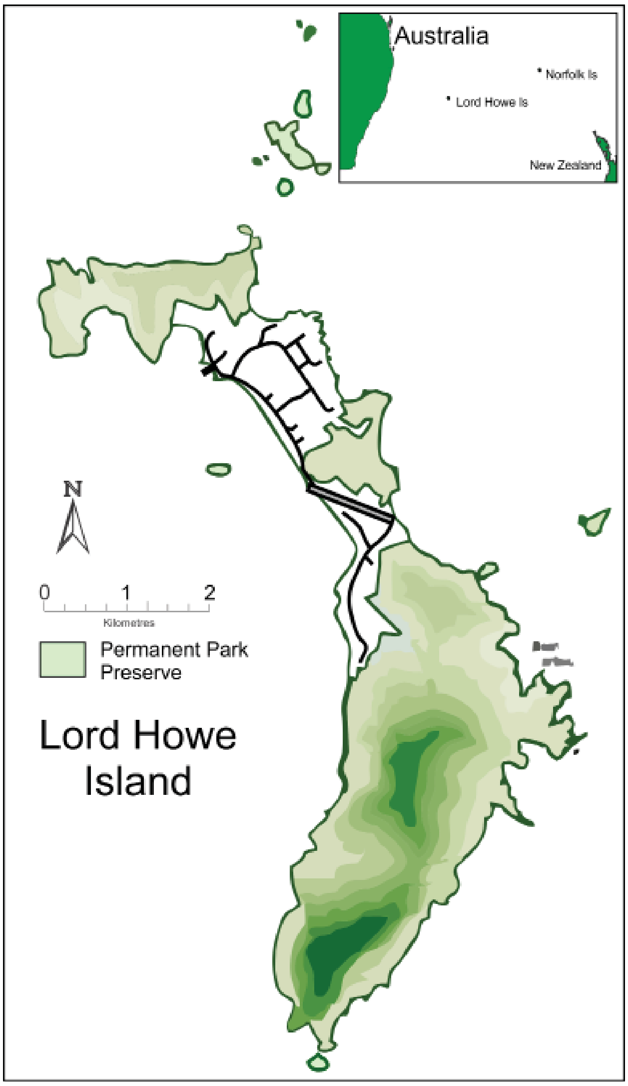

Fig. 1 Lord Howe Island showing the Permanent Park Preserve, airstrip and roads.

1920, highlighting both the long-recognised impacts of rodents and difficulty in achieving meaningful outcomes through ongoing control on the island (Saunders & Brown 2001). Previous control methods included a bounty on rat tails, hunting with dogs and shooting, introduction of owls and the use of various poisons including barium chloride, diphacinone, warfarin and brodifacoum (ibid). Control baiting on the island has undergone several reviews over time with Billing (1999) the most recent, resulting in a current pulse baiting schedule delivering over 4.5 tonnes a year of Ratex (coumatetralyl) and Roban (difenacoum) over approximately 10% of the island. While the use of brodifacoum has been discontinued by the LHIB in the lead up to the eradication, many Island residents continue to use brodifacoum-based rodenticides such as Talon™ and Tomcat™ to control rats and mice around their properties and inside dwellings. The above is intended to highlight the long-term and extensive use of rodenticides on the island, which could be avoided through a one off eradication, but also to highlight that the use of poison on the island is not new and residents are quite familiar with it.

The project

The LHIB is proposing to undertake the Lord Howe Island Rodent Eradication Project (LHI REP) to eradicate introduced rodents from LHI. A secondary outcome would be eradication of the masked owl (*Tyto novaehollandiae),* which was deliberately introduced to LHI in the 1920s and 1930s to control rats. The species now also preys on on many island birds (Milledge, 2010).

The one-off eradication proposes to distribute a cereal-based bait pellet (Pestoff 20R) containing the toxicant brodifacoum across the LHIG via dispersal from helicopters to the uninhabited parts of the island, and by a combination of hand broadcasting and the placement of bait in bait stations in the settlement area (for more information, see LHIB, 2016). Post eradication, rodent prevention and surveillance monitoring would be ongoing to prevent reinvasion and would need a high level of community vigilance.

The LHIB received significant funding (AUS$9.5M) in 2012 for planning and implementation of the REP from the Federal Government's former Caring for Our Country programme (now National Landcare program) AU$4,500,000 and the NSW Environment Trust AU$4,542,442.

PROJECT DEVELOPMENT AND EARLY CONSULTATION

During the turn of the century, successes in island eradications undertaken primarily in New Zealand (summarised in Russell & Broome (2016)) were gaining international attention. At the same time there was growing recognition by government in Australia (state and federal) of the impacts of rodents on LHI (summarised in NSW Scientific Committee, 2000; TSSC, 2006 and DECC, 2007a).

In 2001, a LHIB staff member and a community member attended an international eradication conference and subsequently submitted a proposal to the LHIB seeking support for research into feasibility of eradicating rodents on LHI. This led to commissioning of a feasibility study (Saunders & Brown, 2001) jointly funded by the LHIB and WWF. Saunders and Brown concluded that eradication on LHI was feasible using a combination of aerial broadcast, hand broadcast and bait stations with a brodifacoum based product. Based on some initial consultation undertaken with community on the island during the study, it was considered that the socio-political environment on LHI was conducive to supporting a possible eradication. The study identified potential ecological and social risks and gaps that needed to be further explored and recommended key next steps including:

- a cost benefit analysis
- additional field trials on rodent densities, bait uptake and non-target species impacts
- that a process be established to allow the community to be kept informed and be able to influence decisions
- establishment of a Taskforce to drive implementation
- the feasibility report be made available to the community and briefing sessions provided.

Following the feasibility study, a Rodent Eradication Taskforce was established by the LHIB through an "expression of interest" process in the community, meeting initially in 2002.

Receipt of a donation (ca. AUD$34,000) from the Foundation for National Parks and Wildlife allowed commissioning of a Cost Benefit Analysis (Parkes, et al., 2004). The study looked at additional feasibility, risks and benefits of eradication on LHI and again confirmed that eradication was feasible and highly beneficial, provided risks could be appropriately managed and funding and approvals obtained. The analysis identified social constraints suggesting that people may oppose any proposed eradication attempt at two levels: some may not see the need to attempt it, while others may disagree with the methods required. A process to involve the LHI community in considering options and an active process to seek agreement for baiting on properties and in houses was recommended in the study, but not detailed.

For several more years the LHIB sought funding to deliver facilitated community engagement and on island trials prior to implementation of the REP. Further funding (ca. AUD$ 200,000) was secured in 2008 from the Australian Government Caring for our Country Program to continue planning and trials, community consultation and engagement of a project manager. Consultation in 2008 included several rounds of public hall meetings including:

- specific sessions for livestock, poultry and dog owners
- for residents on the fringe of the settlement area where special consideration are required
- follow up sessions to address concerns previously raised.

A theme emerged that the meetings quickly disintegrated as some community members become vocal and dominating, particularly those opposed. Over several meetings, this led to a reduction in numbers of attendees. It was also unfortunately stated by a scientist at one of the public hall meetings at the time that “the eradication would not go ahead unless there was 100% support”. This created an undeliverable promise still haunting the project today.

Ad-hoc consultation also occurred through development of fact sheets, individual meetings and through briefing papers and updates at Board meeting open sessions.

Based on recommendation in Saunders & Brown (2001) and Parkes, et al. (2004), additional studies were also conducted on the LHI currawong (Carlile & Priddel, 2006) and non-toxic field trials (DECC, 2007b) that examined rodent and non-target species uptake of the bait pellets, bait breakdown in the environment and spread of the bait using helicopter.

In July 2009, locally elected Board members went door to door surveying residents on their views and concerns regarding the REP based on set questions. In total 125 residents were interviewed and detailed responses showed that there was sufficient community support to proceed. The survey results were used together with results of field trials to develop the Draft LHI Rodent Eradication Plan (LHIB, 2009) which then underwent external peer review. The Draft Plan was peer reviewed by the Island Eradication Advisory Group (IEAG) of the New Zealand Department of Conservation; the Invasive Species Specialist Group of the Species Survival Commission of the World Conservation Union; the Worldwide Fund for Nature (WWF), Australia; Birds Australia; Landcare Research, New Zealand; CSIRO and Professor Tim Flannery. All peer reviews of the Plan were supportive.

Public comment on the Draft Plan was sought in November 2009. Of the 83 submissions received, 39 submissions opposed the Plan, 33 supported it, four gave in-principle support, while one submission was undecided. All 39 submissions opposing the Plan were from LHI residents, organisations acting on their behalf, or had strong links with LHI community members. Of the 37 submissions supporting the proposal, 11 originated from LHI, and four came directly from scientists or scientific groups with experience in rodent eradication. All four submissions giving in-principle support originated from LHI as did the single undecided submission. It should be noted that 84 submissions is considered a very high level of response on LHI.

The most frequently raised issue (25 submissions) was concern about non-target impacts during the proposed eradication operation. Other dominant issues included concerns about possible impacts on the health of the community, the tourism industry and the marine environment, as well as the need for improved consultation. The most commonly mentioned issue supporting the Draft Plan was that eradication would deliver clear environmental benefits to LHI (13 submissions) (see Table 1).

The submissions provided a valuable snapshot of opinion, setting the direction for future studies and consultation for the project once additional funding could be secured. A submission analysis report was prepared but unfortunately due to not being able to secure funding for consultation, was not released to the community until 2013. A revised Draft Plan was also prepared addressing submissions but was never released, for reasons unknown.

One study that was progressed immediately was a Human Health Risk Assessment (HHRA) undertaken by a toxicology consultant Toxikos (2010). The HHRA considered all potential pathways related to direct and indirect contact with the poison from the REP (e.g. ingestion, inhalation, skin contact, ingestion of contaminated water and food) and found there to be no significant risk associated with any pathway with the proposed mitigation in place. The HHRA was presented to community and helped to satisfy some community concerns. However, although the LHIB undertook a competitive tender process to select the consultant Toxikos, some members of the community criticised the independence of the study and therefore disregarded the results. Subsequent third-party reviews of Toxikos’ HRRA by New South Wales Health, South Australian Health and Pacific Environment Pty Ltd (outlined in LHIB, 2016) did little to change the perception of some community members.

From 2010 to 2012 the LHIB continued to seek staged funding for progression of the REP. In May of 2012 it was recognised by the Board, that despite information provided to date it was clear that community concerns remained, and further work was required to address these. To enhance community awareness of the benefits that eradication would deliver for the environment, tourism and public health, it was recommended that a professional facilitator be engaged to consult with the LHI community. It was also recommended that a Community Liaison Group (CLG) was created. In June 2012 the project manager at the time resigned.

In July 2012, the LHIB received funding of AU$9.5M to implement the REP in full from the New South Wales Government’s Environment Trust and the Australian Government’s Caring for Our Country programme. The project was divided into three stages:

- Stage 1 – to complete all planning and preparations for the eradication operation
- Stage 2 – to implement the baiting strategy including captive management and post baiting monitoring
- Stage 3 – to monitor the environmental outcomes of the baiting operation.

Again, unfortunately, receipt of the full funding at once led to some perception in the community that the REP was a *fait accompli* and no longer open for community discussion.

In late 2012, a selection process for engaging a community consultation facilitator was undertaken. This included involvement of Board members and 13 community panel members to choose an applicant that would be able to "connect" with islanders. Two shortlisted consultants travelled to the island and were interviewed individually by the 13-member community panel. Consultants 'Make Stuff Happen' were selected and contracted to establish the CLG and, together with the CLG, to develop a draft Communication and Community Engagement Strategy. The CLG (12 members from 17 nominations) held their first meeting on Friday 8 February 2013. Terms of reference were to:

- Review REP information so it is clear, correct and relevant
- Identify ways to communicate with the community about implementing the REP
- Discuss issues and concerns about the REP

Additional meetings were held in March, April and June 2013, facilitated by the consultants. These meetings, whilst unpicking community issues also brought to light that community support: opposition was approximately 50:50. A Community Engagement Report and Plan (Make Stuff Happen, 2013) was developed recommending:

- Providing sufficient resources to restore trust and information flow
- Maintaining the momentum of the CLG and relationship with the LHIB
- Creating a compelling case by focussing on key drivers of change
- Providing content information at different levels
- Providing a variety of options for *how* information is received focusing on small scale approaches
- Demonstrating respect for community concerns and local knowledge.

Further detail can be found in the report (Make Stuff Happen, 2013).

As a recommendation of the report was to try and build support, the consultant facilitated an "open house" bringing experts relevant to the eradication to LHI in Aug 2013. This included a toxicologist, a medical doctor who has worked on eradications (Macquarie and South Georgia), an animal husbandry expert and project staff. Experts were available separately at small tables where community members could sit and ask questions of them over two days. This was held in the neutral ground of the museum and was attended by about 65 residents. Some individuals, however were known to actively boycott the event.

Additional activities in 2013 included:

- Meeting with Tourism Association to discuss risk analysis
- Targeted discussions with specific businesses related to tourism issues
- Key messages refined and communicated through a variety of means.

Following the open house, 'Make Stuff Happen' provided additional recommendations and proposed a Stage 2 engagement strategy to the LHIB. However, given the value of the contract, it had to be retendered on the open market.

Table 1 Key issues raised in submissions to the 2009 Draft Rodent Eradication Plan.

Issue	Number of submissions	% of submissions
Non-target impacts	25	30.1
Human health concerns	18	21.7
More consultation required	18	21.7
Tourism impacts	16	19.3
Marine impacts	14	16.9
Economic impacts	13	15.7
Eradication will deliver environmental benefits	13	15.7
Proposed eradication too risky	10	12.0
Children's health concerns	9	10.8
Threat posed by negative media associated with eradication	9	10.8
Question rodent impacts	8	9.6
Feasibility – it won't work!	8	9.6
Captive management issues	7	8.4
High cost of operation	7	8.4
Use of divers to remove bait	7	8.4
Rodents have significant impacts	7	8.4
Don't support aerial baiting	6	7.2
Peer review process flawed	6	7.2
Quarantine efficacy – new protocols to prevent reinvasion	6	7.2
Expand current control programme	5	6.0
The eradication is an experiment	5	6.0
Need to work with community to gain support	5	6.0
Distrust of Board	5	6.0

In recognition of the differing views within the community putting successful implementation at risk, the LHIB decided in early 2014 to put the proposed eradication on hold, and to go back to the community and to discuss the available options. The Board made the decision to divide the project into two separate but linked stages.

Stage One: community engagement and consultation which would go back to basics and ask what the community wants in relation to the eradication of rodents so that they can make an informed decision on the future of the project. This included the consequences of not doing the eradication. At the completion of that process an assessment was to be made of the level of support to gauge whether it is sufficient to progress to Stage Two.

Stage Two: operational implementation, which would commence in June 2015, but would only take place if there was sufficient community support for the project following the consultation process.

The tender process to select a consultant for the additional community engagement was undertaken with the assistance of the Department of Premier and Cabinet in early 2014 with Elton Consulting selected. Between July 2014 and February 2015, Elton Consulting undertook a series of community consultation visits to Lord Howe Island. They spoke on a one-on-one basis, through personal visits or open sessions at the public hall, to many island residents, (on a number of occasions) concerning the issue of rodent control and potential eradication on the island. They implemented an incremental approach to consultation to unpack the complexity of the community response to the previous rodent eradication process, and to identify what it would take for the community to actively engage in the evaluation of alternatives and options, with the aim of obtaining community support or endorsement of one of the options.

A Community Working Group (CWG) was established, based on residents who indicated a willingness to participate, along with an open call for nomination / involvement, put out through a newsletter to community residents. In working towards a solution, the CWG identified many issues (particularly regarding human health, potential impacts to business and tourism and potential impact to the environment) and considered a range of options. The option to "do nothing" was generally not considered as an alternative, as there was broad agreement that rats and mice are a problem, and that Lord Howe Island would be better off with no rodents.

Two scenarios were therefore further investigated and discussed:

1. Ongoing control through the existing baiting program, and the potential to expand this.
2. An eradication programme as previously proposed or modified where possible to address island residents' concerns.

The CWG agreed to develop and implement a community survey to test community support for these scenarios, whilst recognising that many of the community concerns with the proposed eradication could be addressed during the Planning and Approvals Phase. The CWG also agreed that an additional independent HHRA was needed and should be progressed.

In May of 2015, an options paper (Elton Consulting, 2015a), providing detailed explanation of options and answers to key questions, was disseminated to all people registered on the electoral roll for Lord Howe Island, together with an anonymous survey (Elton Consulting, 2015b) to allow the community to choose between:

Option 1 – Retain and expand the current management programme

Option 2 – moving to the planning and approvals stage of an eradication programme.

The survey also asked for level of agreement on whether the rodent problem needed to be addressed and ranking of areas concerns for both options.

A total of 212 respondents (71% of the 299 people on the electoral roll) participated in the survey. 208 survey responses were received before the closing time. A consensus was reached that the rodent problem on Lord Howe Island needs to be addressed with the majority of respondents (91%) agreeing (38%) or strongly agreeing (53%). A marginal majority 52 of the respondents expressed a preference for Option 2, while 48% of respondents expressed a preference for Option 1.

In line with the agreed Process for Resolution (Fig. 2), the LHIB responded to the majority view and on 19 May 2015 made the decision to proceed to the Planning and Approvals Phase (Option 2).

PROGRESSING TO IMPLEMENTATION

Since 2015, the project team focussed jointly on progressing the necessary approvals and operational planning; and continuing to increase community support for and/or acceptance of the REP, recognising that some people who may not support the REP would however accept it. The latter is critical to ensuring that baiting can be conducted on every property on the island. Residual community issues and how we are attempting to resolve them are detailed below.

Human health

Safety of people has always been a priority for the LHIB and the community when considering the LHI REP. Given the criticism of the independence of the original HHRA described previously, the community suggested that a further additional study be undertaken.

The NSW Office of the Chief Scientist and Engineer (OCSE) was identified by the community / CWG as an agency with a high level of independence and credibility and was subsequently requested to oversee an additional

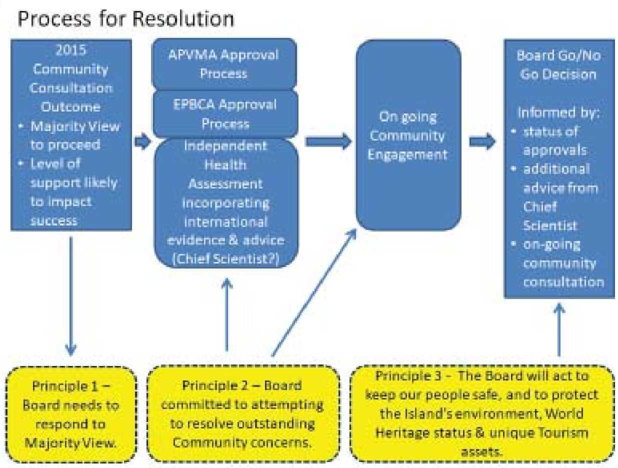

Fig. 2 The agreed process for resolution.

independent HHRA for the project in line with the agreed Process for Resolution (Fig. 2). The OCSE was requested by the NSW Minister for the Environment to convene an Expert Panel in 2016 to:

- Provide advice to the Board on processes for commissioning the HHRA including identification of suitable experts and scope of the request for proposal. (The CWG endorsed the Scope.)
- Review proposals to undertake the HHRA and select a preferred candidate; review project plans and methodologies, and review draft and final reports of the HHRA as required. (Proposed membership of the panel was endorsed by the CWG.)
- Provide advice to the Minister for the Environment on the HHRA.
- Respond to media enquires as they relate to the Terms of Reference for the Expert Panel.

The Expert Panel (with the assistance of two nominated CWG members) selected Ramboll Environ Pty Ltd. to undertake the HHRA. This work, overseen by the OCSE, concluded that estimates of exposure from all potential sources associated with the REP are below those likely to result in adverse health effects for residents and visitors. (Ramboll Environ, 2017).

The outcomes from this additional HHRA and expert panel review concur with the results of previous HHRA undertaken by Toxikos Pty Ltd (2010), which found that, with the proposed mitigation in place, the REP would be safe for the community and visitors. Whilst the outcome was the same as the previous study, it should be noted the process undertaken by the OCSE to select Ramboll was very different to how Toxikos were originally selected, giving the report substantially more credibility within the community.

Economics, tourism and livelihoods

Ongoing concern from some elements of the community regarding potential impacts to tourism (specifically reduction in visitors) before, during or post eradication led the LHIB to commission an economic evaluation of the project in 2016. Community input was sought for the development of the scope of the evaluation and a prominent local business owner was included on the tender selection panel.

A study by Gillespie & Bennett (2017) looked at the costs and benefits (market and non-market services) of not proceeding with the REP compared to the costs and benefits of proceeding (i.e. a Cost-Benefit Analysis) as well as the distribution of said costs and benefits. Though all costs and benefits were considered, particular focus was placed on the potential impacts or benefits to biodiversity (non-market services) and tourism (market service) as the major contributors. This is based on the fact that a key motivation for visiting LHI is to experience the natural, undeveloped and unspoilt surroundings (Lord Howe Island Tourism Association, 2015), some of which are under threat from rodents.

Using choice modelling undertaken for other relevant studies, Gillespie & Bennett (2017) applied the benefit transfer technique to provide an economic value estimate for the biodiversity value of protecting species from extinction. Considering the economic importance of tourism to LHI, tourism impacts or benefits were modelled using supply and demand data in peak and off-peak tourist periods, before, during and after the REP. The study showed that accommodation providers on the island would be the biggest beneficiaries during the REP as the workforce required for the project would more than offset any temporary reduction in tourism. It also showed that tour and accommodation providers would be the major beneficiaries of increased tourism after the eradication. The REP was demonstrated to have a Benefit to Cost ratio of 17:1, resulting in an estimated net social benefit of AU$142M, with AU$58M of that returning directly to LHI residents. Hence, the REP was justified on economic efficiency grounds.

Overall, the cost-benefit analysis was considered an important tool for the REP in overcoming some residents' concerns about tourism and the economy. Others, however, will not be convinced until they actually see the visitors on island during or after the REP.

The eradication method

There has been considerable debate in the community about the method proposed for the eradication and, in particular, the aerial distribution delivery method.

A range of alternatives for eradicating rodents were considered for LHI including alternative techniques and mortality agents. Many were considered to have fatal flaws and were unsuitable for use for eradication on LHI because: a) the technique was not suited to the terrain or size of the island, b) it did not ensure that all individuals would be killed or c) was too experimental. However, early in the project these flaws in alternative methods were not well communicated with the community. The only method identified as capable of removing every rat and mouse on LHI was aerial distribution, in conjunction with minimal hand broadcast and bait stations where required (i.e. the settlement area), of highly palatable bait containing an effective toxicant.

To overcome concerns in the community relating to the method, the project team has recently taken the community through the process of looking at all the options and ruling out options with fatal flaws and therefore not suitable for deployment on LHI (see Table 2). In addition, the project team have undertaken one on one property management plans with all residents to agree the particular baiting method to be used on their property considering their concerns. These two tools have led to a greater understanding in the community of the methods proposed and why other methods were unsuitable.

Environmental and non-target impacts

Some members of the community have been concerned about environmental and non-target impacts. We have used a combination of methods to help allay concerns including:

- Undertaking monitoring on the island to provide evidence of rodent damage to a range of species on LHI and communicated those results back to the community regularly. We have found that a single photo of a rat taking a seabird chick or egg at a local nesting ground to be much more powerful than scientific reports of images of rodent damage locally or images from other locations around the world.
- Conducting a range of studies on the island to determine locally at-risk species and those not at risk (detail in Wilkinson & Priddell, 2011) and repeatedly communicating the results to community.
- Engaging world experts in captive management to manage the captive management programmes on LHI for the two high risk species and conducted trials on-island to show the community how the species can be managed without harm (Taronga Conservation Society, 2014).

Table 2 Assessment of eradication options.

Eradication technique	Suitable for eradication	Feasible for eradication on LHI	Justification
Disease	No	No	No suitable pathogen yet developed that could eliminate all individuals.
Trapping	Yes	No	May be feasible for eradication on small islands, however may cause individuals to become trap shy. Size and inaccessible terrain of LHI makes this option unfeasible.
Biological	No	No	Currently experimental. Likely to fail to completely eradicate the target species.
Fertility Control	No	No	No suitable fertility control yet developed that could eliminate all individuals.
Toxicant – bait station / hand broadcast only	Yes	No	May be feasible for eradication on small islands. Size and inaccessible terrain of LHI makes this option unfeasible.
Toxicant – aerial broadcast only	Yes	No	Highly successful on uninhabited islands. Socially unacceptable on LHI. Problematic with the number and nature of buildings.
Toxicant – combination of aerial and hand broadcast / bait stations	Yes	Yes	Allows for bait to be made readily available to all individual rodents. Brodifacoum in the form of Pestoff 20R has been selected as the preferred toxicant on LHI considering proven success, efficacy and non-target impacts.

- Talking to the community about the differences (particularly in species) between LHI and other islands where non-target impacts have been observed. This included talking through differences in ecology and feeding behaviour leading to different risk profiles.
- Being open and up front with people about our expectations of non-target impacts, how we have formed those expectations and how those impacts compare to current impacts from rodents.
- Conveying the thoroughness, scientific rigour, and independence behind the environmental assessments undertaken by the various regulatory agencies that have assessment and approval roles on the project.
- Highlighting recovery of species and ecosystems from eradications around the world, through sharing recovery stories, science and media on other eradications.
- Developing appropriate mitigation plans for domestic animals and livestock at an individual level.

Lack of trust

Lack of trust of new people, new technologies, and the LHIB in general, was perhaps the most difficult issue to address. It manifests as suspicion of non-island experts and scientific reports, unwillingness to accept change, spreading of misinformation and criticism of LHIB decisions and communications. It stems partially from a sense of resentment by islanders of Government control of the island. It also stems partially from a history of poor communication and follow through by the LHIB on many issues unrelated directly to the REP. Trust is essential to being able to communicate all aspects of the REP, including new information risk, benefits, mitigation and for people to feel comfortable expressing their true concerns. Trust is not easily given and has to be earnt over a long time through listening, demonstrating genuine interest in all aspects of the community (not just those related to rodent eradication), doing what you say you will and following through on commitments.

The most important mechanism we have found to build trust is to have our core project staff living on the island and living in the community for as long as possible. Our Project Manager and Assistant Project Manager have both been resident on LHI for at least two years. Wherever possible we engaged locals in the project including in communications roles, in advocacy, in adding local knowledge, in brainstorming and in any other aspect where they are willing and able.

CURRENT STATUS

The Planning and Approval phase was completed in 2017. At the Sept 2017 Board meeting, the LHIB made the final decision to proceed with the eradication based on the technical, social and financial feasibility of the project as per the agreed process for resolution that was an outcome of ongoing community consultation in 2015 including:

- the status of approvals
- level of community support
- recommendations from an additional independent HHRA.

In March 2018, a decision was made to delay implementation of the REP until winter 2019 due to not having received one of the permits (previously received and surrendered due to a technical administrative flaw).

LESSONS LEARNT

Planning a rodent eradication project on an inhabited island over a long period of time has given us many chances to reflect on what has worked well and what hasn't and to adapt our strategies over time.

Least effective tools

On reflection, the tools we have found least effective were:

- Having a predetermined solution (eradication by the methods proposed) with little opportunity for genuine community input and influence on the decision-making process at the start. This has left some members of the community disenfranchised.
- Flooding the community with more and more technical, scientific information. While science plays a critical role in providing information and answers, it needs to be provided in a way that key concepts and results can be easily understood, are relatable and align with people's values. Many scientific reports are too technical and too detailed for many people to understand, easily, or relate to. More information is not always better.
- Public Town Hall meetings. We have found that these have generally been dominated by a select minority of people (either supporters or opposition), which leaves many people unheard or losing interest.
- Scientists and ecologists are not always the best at community engagement. Often pure science does not address the emotional issues particularly when they concern people's children or livelihoods.
- Mainland consultants and experts. We have found generally (although not always) poor engagement outcomes where we have flown specialists from the mainland to help with various technical aspects. Often the consultants are there for very short timeframes and may not return. As a result, the required depth of understanding of community issues is often overlooked and, consequently, there is little opportunity to build rapport and trust. While we recognise that subject matter and consultation experts are required, they may be best deployed behind the scenes providing the right advice to the core project team.
- Presenting information from other sites – each community considers their island to be different from all others.

Most effective tools

We have found the most effective tools for communicating with our community to be:

- Having key members of the project team based on the island for as long as possible before implementation. This gives the community the opportunity to get to know the team and start to build trust. An open-door policy allows people to come to the project team at any time to discuss any concerns they have with the project. Being based on LHI also allowed the team to understand the broader issues that face the community and to interact with the community outside of work.
- One-on-one consultation with every resident, repeatedly and as many times as necessary on their properties. This has allowed us to identify individual concerns (and underlying motivations) and work with residents to address them. Working through with residents about exactly how they would like baiting undertaken on their properties considering their concerns (i.e. vegetable gardens, pets, children, etc.) has been critical for getting people comfortable with the project.
- The economic evaluation was an important piece of work as it converted biodiversity values (negative outcomes without eradication and positive outcomes with eradication) into economic (tourism) terms that our community could relate to in a meaningful way (livelihoods) and understand key concepts and implications.
- Independence and credibility of the OCSE in undertaking the HHRA.
- Engaging locals on as many aspects of the project as possible where skills allowed.
- Being patient, passionate, resilient and willing to go the extra mile to succeed. These personality traits of individuals in the core project team are essential to gaining trust in the community.

DISCUSSION

The LHI REP has been a long time in planning and community consultation. Implementation of the project was likely drawn out most due to the fact that early planning and consultation was not done as effectively as it could have been. Additionally, funding was received in full which could have sent the message to the community that the project was a *fait accompli*. This did not engender the initial community support that was essential and has led to a long road to recovery. Though many of the tools from the emerging Strategic Environmental Assessment/ Social Impact Assessment toolbox (Russell, *et al*, 2018) were eventually employed on the project, these were often reactive, not integrated and used too late in the process.

The project would have likely encountered much less community resistance if these tools were used much earlier and in an integrated and methodological fashion. It is likely that the plan that was taken to community was too far developed down a particular path and was therefore considered not open to community input. This meant that it wouldn't have allowed sufficient opportunity for the community to adapt the plan to their needs or to feel a sense of ownership of the plan. Although extensive community engagement has since identified issues, and these were addressed, it would have been much more effective if this was done at the start of the planning process. Early community engagement (not information sharing) to gain support needs to be the top-priority for future eradications.

ACKNOWLEDGEMENTS

The LHI REP is funded by the Australian Government's National Landcare programme and the New South Wales Environmental Trust. This paper provides an update to a previous paper published by Wilkinson & Priddell (2011) and builds on a paper by Hutton, et al. (2016) at the Island Arks 2016 conference.

Thanks to previous and current staff involved in the project. This paper has been prepared with the benefit of hindsight and is by no means criticism of those previously involved.

REFERENCES

Billing, J. (1999). *The Management of Introduced Rodents on Lord Howe Island*. Unpublished report. Lord Howe Island. Lord Howe Island Board.

Carlile, N. and Priddel, D. (2006). *Population Size and Distribution of the Lord Howe Currawong* Stepera graulina crissalis. Unpublished Report prepared for the Lord Howe Island Board. NSW: Department of Environment and Conservation.

DECC (2007a) *Lord Howe Island Biodiversity Management Plan*. Hurstville: Department of Environment and Climate Change.

DECC (2007b). *Report on Non-toxic Bait Trials Lord Howe Island – August 2007.* Unpublished report for the Lord Howe Island Board. Hurstville: Department of Environment and Climate Change.

Elton Consulting. 2015a. *Rodent Control on Lord Howe Island – Towards a Solution*. Unpublished report for the Lord Howe Island Board. Elton Consulting Pty Ltd.

Elton Consulting. 2015b. *Lord Howe Island Rodent Consultation Community Survey*. Unpublished report for the Lord Howe Island Board. Elton Consulting Pty Ltd.

Gillespie, R. and Bennet, J. (2017). 'Costs and benefits of rodent eradication on Lord Howe Island, Australia'. *Ecological Economics* 140: 215–224.

Groombridge, B. (1992). *Global Biodiversity: Status of the Earth's Living Resources: A Report*. Report prepared for the World Conservation Monitoring Centre, British Museum (Natural History), and International Union for Conservation of Nature and Natural Resources. London: Chapman and Hall.

Hutton, I., Brice, L., Kelly, D. and Murray, C. (2016) *The Social Aspect to Successful Island Conservation Programs: Lessons Learnt from the Proposed Rodent Eradication on Lord Howe Island*. Paper Presented at Island Arks IV Conference, Norfolk Island, 2016.

IUCN (2017). 'World Heritage Outlook Lord Howe Island Group'. <https://www.worldheritageoutlook.iucn.org/node/990>.

Jones, H.P., Tershy, B.R., Zavaleta, E.S., Croll, D.A., Keitt, B.S., Finkelstein, M.E. and Howald, G.R. (2008). 'Severity of the effects of invasive rats on seabirds: a global review'. *Conservation Biology* 22: 16–26.

LHIB. (2009). *Draft Lord Howe Island Rodent Eradication Plan.* NSW: Lord Howe Island Board.

LHIB. (2016). *Lord Howe Island Rodent Eradication Project: Public Environment Report. EPBC 2016/7703*. Prepared by Lord Howe Island Board. NSW: Lord Howe Island Board. <http://www.lhib.nsw.gov.au/sites/lordhowe/files/public/images/documents/lhib/Environment/Rodent%20Eradication/LHI%20REP%20Final%20Public%20Environment%20Report_21Dec2016.pdf>.

Lord Howe Island Tourism Association, 2015. *Lord Howe Island Destination Management Plan 2014–2017*. Unpublished report for Destination NSW. Lord Howe Island: Lord Howe Island Tourism Association.

Make Stuff Happen. 2013. *Community Engagement Report and Plan*. Unpublished report for the Lord Howe Island Board. Annandale, New South Wales: Make Stuff Happen Pty Ltd.

Milledge, D. (2010). *Research to Inform the Eradication of the Introduced Masked Owl Population on Lord Howe Island*. Unpublished report. Landmark Ecological Services.

NSW Scientific Committee, (2000). *Final Determination to List the Predation by the Ship Rat* Rattus rattus *on Lord Howe Island as a KEY THREATENING PROCESS.* <http://www.environment.nsw.gov.au/determinations/ShipRatLordHoweKTPListing.htm>. Accessed 8 June 2016.

Parkes, J., Ruscoe, W., Fisher, P. and Thomas, B. (2004). *Benefits, Constraints, Risks and Costs of Rodent Control Options on Lord Howe Island*. Unpublished report to the Lord Howe Island Board. Lincoln, New Zealand: Landcare Research.

Pickard, J. (1983). 'Vegetation of Lord Howe Island'. *Cunninghamia* 1: 133–265.

Ramboll Environ (2017), *Human Health Risk Assessment – Proposed Lord Howe Island Rodent Eradication Program*. Unpublished report for the NSW Office of the Chief Scientist. The Junction, NSW, Australia: Ramboll Environ Pty Ltd.

Russell, J.C. and Broome, K.G. (2016) 'Fifty years of rodent eradications in New Zealand: Another decade of advances'. *New Zealand Journal of Ecology* 40(2): 197–204.

Russell, J., Taylor, C.N. and Aley, J. (2018). 'Social assessment of inhabited islands for wildlife management and eradication'. *Australasian Journal of Environmental Management* 25(1): 24–42.

Saunders, A. and Brown, D. (2001). *An Assessment of the Feasibility of Eradicating Rodents from the Lord Howe Island Group*. Unpublished report to the Lord Howe Island Board. Hamilton, New Zealand: Endangered Species Recovery Council.

Shiels, A.B., Pitt W.C., Sugihara R.T., Witmer G.W. (2014). 'Biology and impacts of Pacific island invasive species 11. The black rat, *Rattus rattus* (Rodentia, Muridae)'. *Pacific Science* 68(2): 145–184.

Taronga Conservation Society Australia, (2014). *Captive Management for Woodhen and LHI Currawong Associated with the Lord Howe Island Rodent Eradication Project.* Unpublished report for the Lord Howe Island Board. Sydney: Taronga Conservation Society.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?' *Biological Invasions* 8: 863–891.

Toxikos (2010). *Human Health Risk Assessment for the use of Brodifacoum for the Lord Howe Island Rodent Eradication Plan*. Unpublished Report for the Lord Howe Island Board. Caulfield East, Vic., Australia: Toxikos Pty Ltd.

TSSC (2006) *Commonwealth Listing Advice on Predation by Exotic Rats on Australian Offshore Islands of Less than 1000 km² (100,000 ha).* Threatened Species Scientific Committee. Canberra: Australian Government Department of the Environment.

Wilkinson, I.S. and Priddel, D. (2011). 'Rodent eradication on Lord Howe Island: challenges posed by people, livestock, and threatened endemics'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 508–514. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Recovery and current status of seabirds on the Baja California Pacific Islands, Mexico, following restoration actions

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Abstract The Baja California Pacific Islands, Mexico, are globally important breeding sites for 22 seabird species and subspecies. In the past, several populations were extirpated or reduced due to invasive mammals, human disturbance, and contaminants. Over the past two decades, we have removed invasive predators and, for the last decade, we have been implementing a Seabird Restoration Programme on eight groups of islands: Coronado, Todos Santos, San Martín, San Jerónimo, San Benito, Natividad, San Roque, and Asunción. This programme includes monitoring; social attraction techniques; removal of invasive vegetation; reducing human disturbance; and an environmental learning and biosecurity programme. Here, we summarise historical extirpations and recolonisations during the last two decades of restoration actions, and we update the status of breeding species after more than a decade. To date, from 27 historically extirpated populations, 80% have returned since the first eradication in 1995. Social attraction techniques were key in recolonisations of Cassin's auklet (*Ptychoramphus aleuticus*), royal tern (*Thalasseus maximus*), and elegant tern (*T. elegans*). A total of 19 species breed on these islands, four more species than a decade ago, including 12 new records. The most abundant seabirds, black-vented shearwater (*Puffinus opisthomelas*), Cassin's auklet, western gull (*Larus occidentalis*), and Brandt's cormorant (*Phalacrocorax penicillatus*), have shown a remarkable population increase. Current threats include the potential reintroduction of invasive mammals, guano mining, recreational activities, pollution, and commercial fisheries. To maintain these conservation gains in the long-term it is necessary to continue implementing restoration actions and reinforcing protection on these important natural protected areas.

Keywords: conservation, invasive mammal eradications, population status, seabird recovery, social attraction techniques, threats

INTRODUCTION

Mexican islands and their surrounding waters are key breeding and foraging sites for one-third of all seabird species worldwide, placing Mexico as the third most diverse country and the second in terms of endemism (Croxall, et al., 2012). In particular, the Baja California Pacific Islands (Fig. 1), influenced by the productive waters of the California Current System, support more than a million breeding pairs of 22 seabird species and subspecies (Wolf, et al., 2006). Unfortunately, on these islands at least 18 seabird populations were extirpated, several more diminished from their former abundances, and the Guadalupe storm-petrel (*Oceanodroma macrodactyla*) is presumed extinct due to the presence of invasive mammals, human disturbance, and contaminants that affected their breeding grounds during the last two centuries (Everett & Anderson, 1991; McChesney & Tershy, 1998; Wolf, et al., 2006).

Over the past two decades, we have removed 60 populations of invasive mammals from 39 islands in Mexico, in collaboration with government agencies, academic institutions, fishing cooperatives and a donor network (Aguirre-Muñoz, et al., 2011, 2016, 2018). In the Baja California Pacific, 12 islands smaller than 1,000 ha are now free of invasive predators; 24 populations of cats (*Felis catus*), goats (*Capra hircus*), rabbits (*Oryctolagus cuniculus*), donkeys (*Equus asinus*), dogs (*Canis lupus familiaris*), ship rats (*Rattus rattus*), and deer mice (*Peromyscus eremicus cedrosensis*) were eradicated between 1995–2004 and 2013 (Aguirre-Muñoz, et al., 2011, 2016). Only a small population of house mice (*Mus musculus*) remains on Coronado Sur Island, and white-tailed antelope squirrels (*Ammospermophilus leucurus*) on Natividad Island (Aguirre-Muñoz, et al., 2016).

Seabird surveys, carried out on some of these islands a few years after the eradications, recorded a low number of natural recolonisations (Palacios pers. comm., 2003, Wolf, et al., 2006, Whitworth pers. comm., 2007). In order to attract birds back and improve recolonisation rates, we initiated, in 2008, a Seabird Restoration Programme that includes monitoring, implementing social attraction techniques used successfully elsewhere (Jones & Kress 2012), removal of introduced vegetation for habitat enhancement, and an environmental learning and biosecurity programme with local communities (Aguirre-Muñoz, et al., 2011, 2016). Over the last decade, we have recorded several positive

Fig. 1 The Baja California Pacific islands where seabird restoration actions have been done.

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outcomes after the implementation of these restoration actions that have not been documented yet. Moreover, the last comprehensive compilation of the status of seabird breeding populations on these islands was made more than 10 years ago and needs to be updated (Wolf, et al., 2006).

Here, we summarise historical seabird extirpations for each island and subsequent recolonisations after the implementation of restoration actions during the last two decades and update the status of all breeding species on eight islands groups: Coronado, Todos Santos, San Martín, San Jerónimo, San Benito, Natividad, San Roque, and Asunción.

METHODS

We used historical records of breeding seabirds from published and grey literature to determine the number of extirpated populations and compare the status of seabird populations after the implementation of restoration actions (invasive mammal eradication, social attraction techniques). Current information derives from our own seabird censuses and estimations conducted in 2008–2017 on San Roque and Asunción islands, in 2013–2017 on Coronado, Todos Santos, San Martín, San Jerónimo, and Natividad, and in 2016–2017 on San Benito islands. For surface-nesting species, we surveyed active nests from land-based vantage points, complemented with boat counts and searches around the islands, every 15 days during the whole breeding season. All colonies were mapped and divided into sub-colony sites to increase count accuracy. For burrow-nesting species, we conducted a continuous exhaustive and intensive search of active nests in all potential breeding sites. On islands with accessible nesting sites, we conducted a census all around and across the whole island and checked nest content using a hand-lamp or a borescope. For those species with high nest density such as western gull (*Larus occidentalis*) on Todos Santos Islands, Cassin's auklet (*Ptychoramphus aleuticus*) on San Jerónimo Island, and black-vented shearwater (*Puffinus opisthomelas*) on Natividad Island, we estimated nest densities during peak incubation, counting nests within circular and square plots randomly distributed and georeferenced. Burrow occupancy was determined by recording apparently occupied burrows, i.e. with signs of activity such as guano, feathers, clear entrances, and footprints (Walsh, et al., 1995). Population size was calculated through Bayesian statistics using the total number of nests and occupied burrows (McCarthy, 2007). We included in our counts pairs nesting within artificial colonies installed on all the islands (Table 1).

We analysed the number of recolonisations of extirpated colonies by island and seabird group (surface-nesting species and burrow-nesting species). A recolonisation rate was not possible to estimate as post-eradication surveys were not systematic on many islands until we started our monitoring in 2008. We also present a brief account for each currently breeding species, where we include the maximum number of breeding pairs estimated during our own survey period on each island, except for storm-petrels on San Benito Islands.

Table 1 Social attraction techniques implemented on seabird populations on the Baja California islands, Mexico, from 2008 to 2017. Y = Yes, N = No. *Mirrors were used from 2008–2011.

Species	Island	Year	Social attraction techniques	Successful
Heermann's gull	San Roque	2008–2017	Decoys, acoustic playbacks	Y
Elegant tern	San Roque	2008–2017	Decoys, acoustic playbacks, mirrors*	Y
	Asunción	2008–2017	Decoys, acoustic playbacks, mirrors*	N
Brandt's cormorant	Coronado Norte	2014–2017	Decoys, acoustic playbacks	N
	Coronado Sur	2014–2017	Decoys, acoustic playbacks	N
	Todos Santos Sur	2014–2017	Decoys, acoustic playbacks	Y
Double-crested cormorant	Coronado Norte	2014–2017	Decoys, acoustic playbacks	N
	Coronado Sur	2014–2017	Decoys, acoustic playbacks	N
	Todos Santos Sur	2014–2017	Decoys, acoustic playbacks	Y
Pelagic cormorant	Todos Santos Sur	2016–2017	Decoys	N
Cassin's auklet	Coronado Norte	2015–2017	Artificial burrows, acoustic playbacks	N
	Coronado Sur	2014–2017	Artificial burrows, acoustic playbacks	Y
	Todos Santos Sur	2014–2017	Artificial burrows, acoustic playbacks	Y
	Todos Santos Norte	2016–2017	Artificial burrows, acoustic playbacks	Y
	San Martín	2014–2017	Artificial burrows, acoustic playbacks	N
	San Jerónimo	2014–2017	Artificial burrows, acoustic playbacks	Y
	Natividad	2014–2017	Artificial burrows, acoustic playbacks	Y
	San Roque	2014–2017	Artificial burrows, acoustic playbacks	Y
	Asunción	2014–2017	Artificial burrows, acoustic playbacks	Y
Scripps´s murrelet	Todos Santos	2016–2017	Artificial burrows, acoustic playbacks	Y
Black storm-petrel	Coronado Norte	2015–2017	Artificial burrows, acoustic playbacks	N
	Coronado Sur	2015–2017	Artificial burrows, acoustic playbacks	N
Ashy storm-petrel	Coronado Norte	2017–2017	Artificial burrows, acoustic playbacks	N
	Coronado Sur	2015–2017	Acoustic playbacks	N
	Todos Santos Sur	2016–2017	Artificial burrows, acoustic playbacks	N

Study area

The eight island groups are located on the continental shelf off the west coast of the Baja California Peninsula, Mexico within 66 km of the coast (Fig. 1). Their climate is Mediterranean to desert-like. The northern islands are characterised by subarctic waters throughout the year while a tropical-subtropical domain persists during summer and autumn in the southern islands (Durazo & Baumgartner, 2002; Durazo 2009, 2015). Natividad (736 ha), San Roque (35 ha), and Asunción (41 ha) islands were designated as part of the El Vizcaíno Biosphere Reserve in 1988 (CONANP, 2000) while Coronado (173 ha), Todos Santos (118 ha), San Martín (265 ha), San Jerónimo (48 ha) and San Benito (541 ha) were recently included in the Islas del Pacífico de la Península de Baja California Biosphere Reserve in 2016 (DOF, 2016).

RESULTS AND DISSCUSION

Recovery and status

In total, according to historical records, 27 seabird populations were extirpated from the 12 coastal Baja California Pacific islands where restoration actions were conducted; Todos Santos, San Martín, and San Roque islands were the most affected islands, with between five to six taxa extirpated on each island. In contrast, San Benito Islands have no historical record of any extirpation (Table 2, Fig. 2). Extirpated species included five burrow-nesting species: Leach's storm-petrel (*Oceanodroma leucorhoa*), black storm-petrel (*O. melania*), Scripps's murrelet (*Synthliboramphus scrippsi*), Craveri's murrelet (S. craveri) and Cassin's auklet; and five surface-nesting species: brown pelican (*Pelecanus occidentalis*), double-crested cormorant (*Phalacrocorax auritus*), Brandt's cormorant (*Phalacrocorax penicillatus*), royal tern (*Thalasseus maximus*), and elegant tern (*T. elegans*) (Table 2). Burrow-nesting species lost 15 breeding populations of which 9 colonies corresponded to Cassin's auklet and Scripps's murrelet that were extirpated from almost all their historical breeding sites in Mexico (Table 2, Fig. 3). Similarly, 12 colonies of surface-nesting species were extirpated, with brown pelican and double-crested cormorant being the most impacted species (Fig. 3).

After two decades of restoration actions, in total, 22 colonies of extirpated seabirds have returned to breed to these islands, which represent 80% of all extirpated colonies. San Martín and San Roque islands are the islands that have benefited the most as currently all extirpated species are breeding again on these islands. Likewise, the species with more colonies extirpated are now breeding on almost all their historic sites (Table 2, Fig. 3). Social attraction techniques were key in recolonisations of Cassin's auklet on Natividad Island; and royal tern and elegant tern on San Roque Island (Table 1).

Moreover, we have recorded 12 new colonisations during the last decade that have never been recorded before, five of them on San Jerónimo Island (Table 2). These new records together with recolonisations have increased considerably the number of breeding taxa on many islands in comparison with the last comprehensive compilation (Wolf, et al., 2006). For instance, San Jerónimo Island with only four species recorded last decade now supports 12 breeding species (Fig. 4). Currently, breeding seabirds on these 12 islands comprise 19 species, four more species than the last record (Wolf, et al., 2006).

The most abundant seabird is the black-vented shearwater, which has a total population an order of magnitude higher than all other species, but is restricted to three breeding sites (Natividad, San Benito and Guadalupe islands). Cassin's auklet, western gull, Brandt's cormorant, and double-crested cormorant are relatively abundant and have a wide distribution. In contrast, elegant tern and blue-footed booby (*Sula nebouxii*) only have one pair nesting in one site (Table 2).

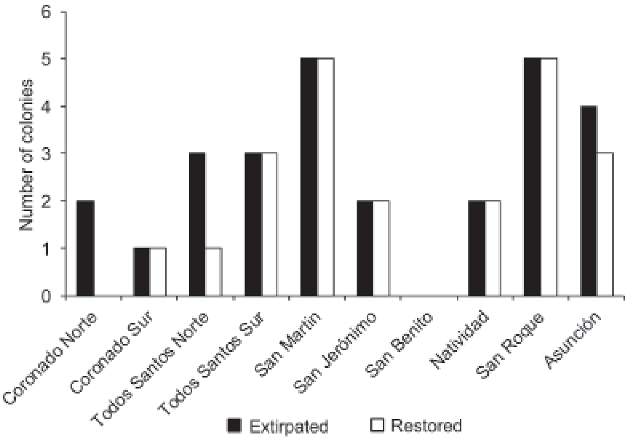

Fig. 2 Number of seabird colonies historically extirpated and restored after two decades of restoration actions on the Baja California Pacific islands, Mexico.

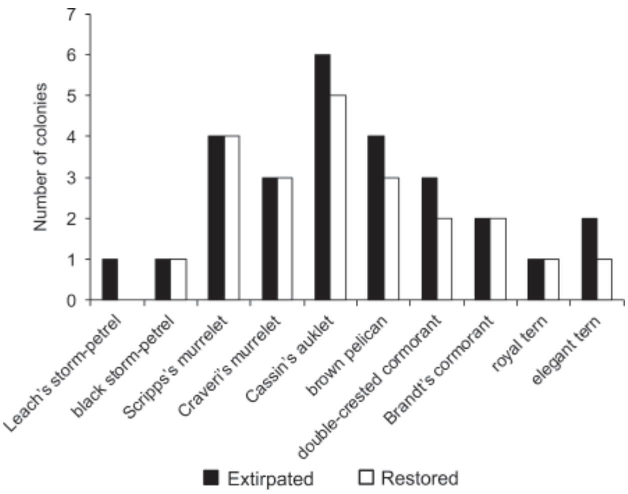

Fig. 3 Number of colonies historically extirpated and restored after two decades of restoration actions of each seabird species on the Baja California Pacific islands, Mexico.

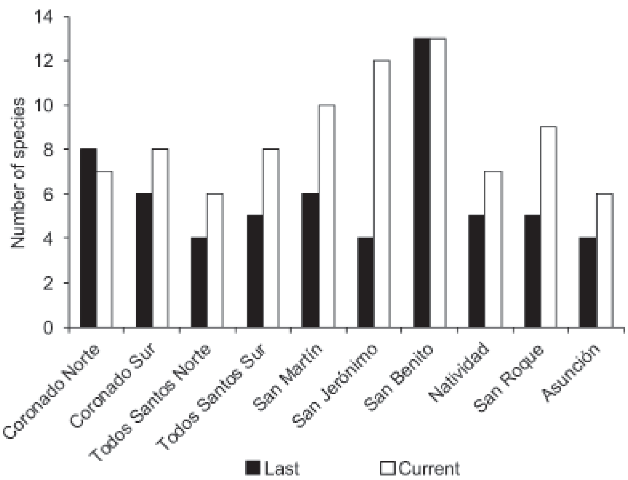

Fig. 4 Number of seabird species recorded a decade ago in the last comprehensive compilation (Wolf, et al., 2006) and during our monitoring on the Baja California Pacific islands, Mexico.

Table 2 Last and current status of the seabird breeding populations on the Baja California Pacific Islands, Mexico. CN: Coronado Norte, CS: Coronado Sur, TSN: Todos Santos Norte, TSS: Todos Santos Sur, SM: San Martín, SJ: San Jerónimo, SB: San Benito, NA: Natividad, SR: San Roque, AS: Asunción. L = mostly estimates from Wolf, et al., 2006; C = our own surveys: maximum numbers recorded during the monitoring period on each island. Estimates are number of breeding pairs. Historical extirpations are indicated. B: Breeder, PB: Probable breeder, E: Extirpated, PE: Possibly extirpated, U: Unknown. *New record, #Intensive nest search on the island, $Estimation for the whole archipelago with spotlight survey, %Adults with fully developed brood patch were captured with mist-nest, &Scripps's and Guadalupe murrelets were considered together, a) Van Rossem, 1915; b) Carter, et al., pers. comm. 1996; c) Ainley & Everett, 2001; d) Whitworth, et al., pers. comm.,2007; e) Palacios, et al., pers. comm. 2003; f) Wolf, et al., 2006; g) Jehl, 1977; h) Van Denburgh, 1924; i) Jehl & Bond, 1975; j) Everett, 1989; k) Anderson & Keitt, 1980; l) Everett & Anderson, 1991; m) Whitworth et al. 2018. n) Keitt, et al., 2003; o) McChesney & Tershy, 1998.

Species	CN		CS		TSN		TSS		SM		SJ		SB		NA		SR		AS	
	L	C	L	C	L	C	L	C	L	C	L	C	L	C	L	C	L	C	L	C
Black-vented shearwater													125–600[f]	124[#]	76,570[n]	110,000-120,000				
Leach's storm-petrel	E[a]	PE										PB*	510,000[f]	B						
Ashy storm-petrel	PB[b]	PB				55*		20*		PB*%										
Black storm-petrel	B[c]	78	E[c]	1						PB*%			260,000[f]	B						
Least storm-petrel													135,000[f]	B						
Brown pelican	1,818[d]	203	33[d]	328	E[e]	E	E[e]	1,200	204[Eek]	376		299*	197[f]	101	37–75[f]	366	<10[f]	99	E[f]	270
Blue-footed booby												1*								
Double-crested cormorant	168[d]	30	230[d]	187	E[e]	E	93[e]	238	520[Eel]	791	20[f]	150	63[f]	40	57[f]	434	E[f]	113	10[f]	131
Brandt's cormorant	100[d]	32	150[d]	164	0[Beh]	53	336[e]	732	E[e]	335	E[f]	833	79[f]	18	750[f]	3,504	<100[f]	5,802	25–50[f]	5,200
Pelagic cormorant	7[d]		2[d]	3	20[e]	15		13				1*								
Heermann's gull													100[f]	109			34[f]	42		
Western gull	225[e]	133	135[e]	364	150[e]	2,248–3,691	1500[e]	5,846–9,598	300[e]	1382	250[f]	2,442	575[f]	1,010	2,500–5,000[f]	2,746	B[f]	1,749	B[f]	1,373
Caspian tern										186*		49*								
Royal tern												80*					E[l]	870		
Elegant tern																	E[l]	1	E[l]	E
Scripps's murrelet	750-1,250[$f]	15[#]	B[e]	22	B[PEfi]	19	25-125[$PEfi]	90[#]	25–125[PEfi]	PB	50–250[PEfi]	9[#]	125-500[f&]	174[&]						
Guadalupe murrelet												PB								
Craveri's murrelet									1[m]	PB			PB[f]	B	PE[m]	1*	B[PEm]	4	B[PEm]	1
Cassin's auklet	E[g]	E		14*	B[f]	12	B[PEfj]	20	500–2,500[Efl]	136[#]	30,000[f]	50,000-110,000	37,667[f]	B	E[o]	10	E[o]	1,659	E[o]	2,128
Total breeding taxa	8	7	6	8	4	6	5	8	6	10	4	12	13	13	5	7	5	9	4	6
Total extirpated taxa	2	2	1	0	2	2	1	0	1	0	1	0	0	0	2	0	4	0	3	1

Black-vented shearwater (*Puffinus opisthomelas*)

In the past, the breeding population of black-vented shearwater declined on Natividad Island, its main breeding colony worldwide due to predation by feral cats and habitat destruction (Keitt, et al., 2002, 2003). At present, we estimate a population of 110,000–120,000 breeding pairs, which indicates almost a twofold increase in relation to the last estimation two decades ago (Keitt, et al., 2003). The small population on San Benito Islands of around 100 pairs remains almost unchanged since its last record (Wolf, et al., 2006, Table 1).

Leach's storm-petrel (*Oceanodroma leucorhoa*)

Leach's storm-petrel, considered the most abundant species in the region, currently breeds only on Islote Medio in Coronado Islands and on San Benito Islands (Wolf, et al., 2006), however, its population estimate has not been updated yet. In the last century, it was extirpated from Coronado Norte Island by feral cats (Grinnell & Daggett, 1903, van Rossem, 1915), our surveys indicate that the species is possibly extirpated. In 2016, we captured adults with brood patches using mist nets on San Jerónimo Island, thus, we consider this species as a probable breeder but we have not found active nests.

Ashy storm-petrel (*Oceanodroma homochroa*)

The Coronado Islands are considered the southernmost breeding range of the ashy storm-petrel and the only breeding site in Mexico (Ainley, 1995). This species was considered as a probable breeder on Coronado Norte Island (Jehl, 1977) and at present, our surveys indicate the same, although we have not found an active nest. In the last decade, Islote Medio, an islet historically pest-free, was the only confirmed site with a small breeding population (Wolf et al., 2006, Carter, pers. comm. 2006). However, we recently confirmed this species breeding on Todos Santos Islands. In 2014, we found the first active nest on Todos Santos Sur Island, and we corroborated species identification by measuring adults captured using mist-nets at night; broadcasting responses in the nest; and carrying out genetic analyses (GECI unpubl. data). We captured five adults with brood patches on San Martín Island, which indicates that the breeding range of the ashy storm-petrel is probably expanding or is wider that was recorded before (Table 2).

Black storm-petrel (*Oceanodroma melania*)

In the Mexican Pacific, black storm-petrels nest exclusively on Coronado and San Benito islands (Ainley & Everett, 2001). On Coronado Islands, this species is recorded as breeding on Coronado Norte, Coronado Medio, and Islote Medio, and as extirpated on Coronado Sur, with a total estimated population of 100–150 breeding pairs (Grinnell & Daggett, 1903; Osburn, 1909; Sowls, et al., 1980; Ainley & Everett, 2001; Carter, pers. comm. 2006). In 2016, we found 120 breeding pairs breeding on Coronado Norte and Islote Medio, and one nest on Coronado Sur, which indicates this species recolonisation of this island (Table 2). On San Martín Island, we captured adults with fully developed brood patches in 2017, thus, we consider this species as a probable breeder, but we have not found an active nest yet. Its breeding population size on San Benito Islands has not been updated yet since the last estimate in 1999 (Wolf, et al., 2006).

Brown pelican (*Pelecanus occidentalis)*

Currently, the brown pelican breeds on all its historical breeding sites on these islands, except on Todos Santos Norte Island, and 40% of its breeding population is concentrated on Todos Santos Sur Island (Table 2). San Jerónimo Island, previously not recorded as a breeding site (Wolf et al., 2006), recently supports a population of around 300 pairs (Table 2). This species was one of the most affected by organochlorines and human disturbance; colonies of thousands or hundreds of pairs recorded in the last century were dramatically reduced on Coronado (ca. 5,000 pairs; Jehl, 1973, Gress, 1995), and San Benito islands (~1,000 pairs, Everett & Anderson, 1991; Wolf, et al., 2006), and was extirpated on Todos Santos (Everett & Anderson, 1991; Palacios & Mellink, 2000; Palacios, pers. comm. 2003), San Martín (ca. 1000 pairs; Jehl, 1973; Anderson & Keith 1980), and Asunción islands (Anthony, 1925; Wolf, et al., 2006). All colonies, except on Todos Santos Islands, recovered considerably after these threats were mitigated (Palacios & Mellink, 2000; Wolf, et al., 2006; Whitworth, pers. comm. 2007). At present, no declined colony has reached its historical numbers as the species' population size remains in the hundreds of pairs, except on Todos Santos Sur Island that supports a population of more than 1,000 pairs (Table 2).

Blue-footed booby (*Sula nebouxii*)

We recorded one nest of blue-footed booby with two chicks on San Jerónimo Island in September 2016. This record represents the first on the Baja California Pacific islands and the northernmost breeding range for this species that was previously considered on Midriff Islands, in the Gulf of California (Hernández Díaz & Salazar Gómez, 2011).

Double-crested cormorant (*Phalacrocorax auritus*)

During the last century, the double-crested cormorant was extirpated from Todos Santos Norte (Van Denburgh, 1924; Palacios pers. comm. 2002), San Martín (Everett & Anderson, 1991), and San Roque islands (Wolf, et al., 2006). Currently, this species breeds on all 12 islands, except on Todos Santos Norte Island where it remains extirpated (Table 2). In the past, San Martín Island supported the largest colony in North America with hundreds of thousands of pairs (Wright, 1913; Gress, et al., 1973; Carter, et al., 1995); after the main threats were removed, the colony increased from zero to around 600 pairs (Palacios & Mellink, 2000, Palacios, pers. comm. 2003) and, at present, this island sustains the biggest population in the region with about 800 pairs (Table 2).

The breeding colony on the Coronado Islands declined from thousands to hundreds of pairs (Howell, 1917; Gress, et al., 1973, Carter, et al., 1995), and on San Roque Island from thousands to zero pairs (Townsend, 1923; Huey, 1927; Wolf, et al., 2006). We recorded double-crested cormorant recolonisation on San Roque in 2008. These colonies have remained in the hundreds of pairs during the last two decades (Carter, pers. comm. 1996; Palacios, pers. comm. 2003; Whitworth, comm. pers. 2007; Table 2).

The small colony recorded on Natividad in 2000 of around 60 nests (Wolf, et al., 2006), at present, has a sevenfold increment in population size (Table 2). On Todos Santos Sur Island the colony doubled its size and social attraction techniques were successful with a record of eight nests within an artificial colony.

Brandt's cormorant (*Phalacrocorax penicillatus*)

Brandt's cormorant returned to nest at all its historical breeding sites on the Baja California Pacific Islands (Table 2). In the past, it was extirpated from San Martín Island, the main breeding site with several thousand nests (Wright, 1913; Everett & Anderson, 1991; Palacios & Mellink, 2000; Palacios, pers. comm. 2003), and from San Jerónimo

Island (Wolf, 2002). At present, both islands maintain colonies of hundreds of pairs, and the largest colonies with > 3,000 pairs are located on the southernmost islands, Natividad, San Roque, and Asunción that before had low numbers of pairs (Table 2; Wolf, et al. 2006). In total, the breeding population of Brandt's cormorant has increased nine times more than the last decade from around >1,000 pairs to > 10,000 pairs (Table 2; Wolf, et al. 2006).

Pelagic cormorant (*Phalacrocorax pelagicus*)

Coronado and Todos Santos islands are considered the southernmost breeding range for the pelagic cormorant (Hobson, 2013). On these islands, small breeding populations have been previously reported on all four Coronado Islands and Todos Santos Norte Island (Palacios, pers. comm. 2003; Carter, pers. comm. 2006, Whitworth, pers. comm. 2007). During our monitoring in 2013–2017, we have recorded nests on Coronado Sur, Coronado Medio, both Todos Santos islands, and, for the first time, one nest on San Jerónimo Island in 2017, which represents an expansion of its breeding range to the south (Table 2).

Heermann's gull (*Larus heermanni*)

Heermann's gull breeds in small colonies (ca. 50–100 pairs) on San Benito Medio and San Roque islands (Table 2). On San Benito, the colony has increased from nine nests (Jehl, 1976) to more than 100 pairs that has not changed during the last decade (Wolf, et al*.,* 2006, Table 2). On San Roque, previous surveys showed a population of 35–42 pairs (Huey, 1927; Mellink, 2001). In 2008, when we started implementing social attraction techniques, we recorded 23 nests within the artificial colony and also during all subsequent years. The colony has reached its maximum number in 2017 with 42 nests (Table 2).

Western gull (*Larus occidentalis*)

The western gull is a species widely distributed on the Baja California Pacific islands. There are no historical records of extirpated colonies. It breeds on all 12 islands and is one of the most abundant species, with a total population estimate of approximately 20,000 breeding pairs (Table 2). The Todos Santos Islands concentrate around 50% of the current population. Estimates 10 years ago, showed a total population three times smaller than today. This increase has been very remarkable mainly on Todos Santos Sur, but also on Todos Santos Norte, San Martín, and San Jerónimo where colony sizes range from about 1,000 to 8,000 pairs (Table 2).

Caspian tern (*Hydroprogne caspia*)

In 2013, we recorded 15 nests of Caspian tern on San Jerónimo Island, the first record on the Baja California Pacific islands. In 2014, the population increased to 49 pairs and we also recorded nests on San Martín Island (89 pairs). The colony on San Jerónimo decreased to 11 pairs the last year because it was established on a California sea lion (*Zalophus californianus*) resting area. In contrast, the colony on San Martín found suitable habitat on the island and has increased to almost 200 pairs. (Table 2).

Royal tern (*Thalasseus maximus*)

This species was extirpated from San Roque Island and its last breeding record was 90 years ago (Bancroft, 1927; Everett & Anderson, 1991). In 2017, after eight years of the implementation of social attraction techniques, 870 breeding pairs were recorded nesting within the artificial colony installed for elegant tern (Table 2). For the first time, we found 80 nests of royal tern on San Jerónimo Island in 2013 but their numbers decreased rapidly to seven in 2017 (Table 2).

Elegant tern (*Thalasseus elegans*)

In the past, the elegant tern bred on San Roque and Asunción islands (Anthony, 1925) but was extirpated from both islands (Everett & Anderson, 1991). Currently, in 2017, we found one pair nesting on San Roque Island within the colony of royal tern associated with the artificial colony. On Asunción Island, this species remains extirpated.

Scripps's murrelet (*Synthliboramphus scrippsi*)

Scripps's murrelet was presumably extirpated from all historical breeding sites, except Coronado and San Benito islands (Table 2; Jehl & Bond, 1975; Everett & Anderson, 1991; Drost & Lewis, 1995; Wolf et al., 2006). At present, it has returned to breed on all the islands where was extirpated (Table 2). Our nest census on the islands shows lower population sizes from Coronado to San Benito in comparison to nocturnal surveys at-sea conducted a decade ago (Wolf, et al., 2006; Carter, pers. comm. 2015). We considered the last monitoring overestimated the population size. We found four breeding pairs nesting in artificial burrows on Todos Santos Sur Island (Table 1).

Guadalupe murrelet (*Synthliboramphus hypoleucus*)

Guadalupe murrelet breeds on the San Benito islands in a low proportion in comparison with Scripps's murrelet, but we do not have a precise number as identification in the nest is complicated. Colonies extirpated from Natividad, San Roque and Asunción islands mentioned by Wolf, et al. (2006) may have been a misidentification of *Synthliboramphus craveri* (Keitt, 2005, Carter pers. comm. 2015; Whiworth, et al., 2018). We recorded a pair (one individual was Guadalupe and the other Scripps's murrelet) on a trap-camera on San Jerónimo Island but the species' breeding status is not confirmed yet.

Craveri's murrelet (*Synthliboramphus craveri*)

Craveri's murrelet is currently breeding in low numbers on San Benito, Natividad, San Roque, and Asunción islands (Table 2). Whitworth, et al. (2018) previously found breeding evidence on San Martín, San Roque, and Asunción islands in 2007.

Cassin's auklet (*Ptychoramphus aleuticus*)

Cassin's auklet was also extirpated from almost all of its historical breeding sites, except from San Jerónimo and San Benito islands (Table 2). Currently, it is breeding again on all islands but Coronado Norte, an island that supported a population of thousands of breeding pairs (Osborn, 1909) extirpated by cat predation (Jehl, 1977). Extirpated colonies from San Roque and Asunción islands have been growing from about 100 pairs in 2008, when we recorded their recolonisation, to around 2,000 pairs in 2017 (Table 2).

Social attraction systems were key for the colonisation on Coronado Sur and recolonisation on Natividad Island. In 2017, we recorded 20 breeding pairs nesting in artificial burrows on Coronado Sur Island, which represents the first record for this island, and for the archipelago, as the last record was on Islote Medio, a pest-free islet, three decades ago (Everett & Anderson, 1991). After more than a century since the last breeding record (Kaeding, 1905), in 2016, we found five breeding pairs on Natividad Island, including one pair inside an artificial burrow close to a sound system (Table 1). We recorded one nest on Todos Santos Sur in 2014, and currently, more than 40 pairs are nesting on the archipelago, 17 of them in artificial burrows (Tables 1 and 2).

THREATS

Major historic threats to seabird colonies included predation by invasive mammals, habitat modification, and direct disturbance (Everett & Anderson, 1991; Wolf, et al., 2006). Current threats are similar to a decade ago but less extensive: 1) potential reintroduction of invasive mammals; 2) invasive plants that reduce nesting habitat, 3) habitat modification by guano mining, 4) exploitation of eggs, 5) disturbance by human activities including recreation and inadequate waste management; 6) fisheries impacts, and 7) pollution.

All these islands are now free of invasive predators (Aguirre-Muñoz, et al., 2016), however, potential reintroduction is high on all islands due to constant movement from the continent, as temporal and permanent fishermen's camps are established on all of them, except on the Coronado Islands. Natividad Island, inhabited by a fishing community of about 300 people, is the most susceptible to this threat: the reintroduction of a few pets has been recently recorded (CONANP, comm. pers.). The impacts of house mice that remain on Coronado Sur Island and white-tailed antelope squirrels on Natividad Island need to be evaluated.

Invasive plants such as ice-plant (*Mesembryanthemum crystallinum*) are widely distributed on many islands (Rebman et al., 2016), and are displacing native flora and affecting their associated fauna. Brandt's cormorants that nest on clean areas along the coast line, and Cassin's auklets and black-vented shearwaters that breed in subterranean burrows might be the most affected species.

Guano mining caused severe damage on San Jerónimo, San Roque, and Asunción islands during the last century (Everett & Anderson, 1991; Wolf, et al., 2006). At present, this activity continues on San Jerónimo Island at least since 2015 and is causing disturbance and destruction of the Cassin's auklet colony (GECI unplub. data). Human exploitation of western gull eggs persists on San Benito and Natividad islands. However, the impact on these populations is unknown. In 2016, the harvesting of Heermann's gull eggs on the small colony of San Benito Medio Island caused low productivity (GECI unpubl. data).

Recreation activities (surfing, kayaking, and fishing) are a threat on Todos Santos, San Benito Oeste, and Natividad islands. We have recorded tourists and residents walking close to breeding colonies which could cause temporal abandonment of nests and increase gull predation, and also damage to the burrows of nocturnal species. On Natividad Island, a metal fence around the town landfill built in front of the breeding colony in 2014, severely impacted the population. In 2016, the structure was removed by the fishing cooperative, however, inadequate waste management is a serious problem for the black-vented shearwater.

Commercial fishing is one of the most important economic activities along the Mexican Pacific coast (CONAPESCA, 2014). Although no fisheries bycatch impacts have yet been evaluated, information from the Gulf of California, shows that 17 species of seabirds and aquatic birds, mainly brown pelicans and blue-footed boobies, are incidentally caught in nets during fishing operations (Comunidad y Biodiversidad, A.C. pers. comm.). The most abundant seabird species in the region forage on small pelagic fish, thus, probable competition for food with commercial fisheries represents a potential threat that should be studied.

Light pollution in fishermen's camps created an important impact on nocturnal species decades ago (Wolf, et al., 2006) – currently this threat has been mitigated (GECI unpub. data) but still it is necessary to evaluate it on fishing boats around the islands. Current information about pollution-related threats is scarce. Oil spill hazards are a potential threat due to the region being an important transportation route and having fuel reception facilities. However, there is no action plan or personnel trained to manage oiled seabirds, and fauna in general.

Plastic consumption for variety of seabird species is increasing worldwide (Wilcox, et al., 2015). We have found evidence of plastic consumption in black-vented shearwater breeding on Guadalupe Island (GECI unpubl. data), thus, the impact of microplastics on seabirds in the region requires evaluation.

CONSERVATION

Although 65% of all seabirds recorded breeding on the islands are listed in the IUCN Red List of threatened species and are protected in Mexico in the Norma Oficial Mexicana NOM-059-SEMARNAT-2010, it is essential to update the status of several species. The majority of protected natural areas fully protect breeding and foraging areas for coastal species, however, foraging sites of pelagic seabirds are not included as most are located hundreds of kilometres from colonies. To address this issue, we are developing a Marine Important Bird Areas (Marine IBAs) proposal in collaboration with government agencies, national and international NGOs and seabird experts. We are also in the process of publishing an action plan for endemic seabirds that delineates the next actions to improve their conservation status. In the short-term this plan will incorporate all breeding seabirds in Mexico.

Regulations and surveillance enforcement to prevent introduction of invasive species, mitigate disturbance to colonies and impacts of fisheries are primordial. An important step is the National Island Biosecurity Programme, recently initiated on several islands including San Benito Islands, which aims to involve all key stakeholders in the protection of island environments from invasive alien species (Latofski-Robles et al., 2019). Moreover, it is necessary to continue working together with local communities to raise awareness about the threats seabirds are facing.

We consider research priorities to improve management decisions are: 1) to continue monitoring of populations and obtain data on productivity; 2) to obtain accurate population estimates for nocturnal seabirds, especially for storm-petrels; 3) to evaluate the impact of threats such as fisheries interactions and microplastics.

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REFERENCES

Aguirre-Muñoz, A., Samaniego-Herrera, L., Luna-Mendoza, L., Ortiz-Alcaraz, A., Rodríguez-Malagón, M., Méndez-Sánchez, F., Félix-Lizárraga, M., Hernández-Montoya, J.C., González-Gómez, R., Torres-García, F., Barredo-Barberena, J.M. and Latofski-Robles, M. (2011). 'Island restoration in Mexico: ecological outcomes after systematic eradications of invasive mammals'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 250–258. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Aguirre Muñoz, A., Samaniego Herrera, A., Luna Mendoza, L., Ortiz Alcaraz, A., Méndez Sánchez, F. and Hernández Montoya, J. (2016). 'La Restauración Ambiental Exitosa de las Islas de México: Una Reflexión sobre los Avances a la Fecha y los Retos por Venir'. In: E. Ceccon and C. Martínez-Garza (eds.) *Experiencias Mexicanas en la Restauración de los Ecosistemas*, pp. 487–514. Ciudad de México: CRIM-UNAM, Universidad Autónoma del Estado de Morelos, CONABIO.

Aguirre-Muñoz, A., Bedolla-Guzmán, Y., Hernández-Montoya, J., Latofski-Robles, M., Luna-Mendoza, L., Méndez-Sánchez, F., Ortiz-Alcaraz, A., Rojas-Mayoral and Samaniego-Herrera, A. (2018). 'The Conservation and Restoration of the Mexican Islands, a Successful Comprehensive and Collaborative Approach Relevant for Global Biodiversity'. In: A. Ortega-Rubio (ed.) *Mexican Natural Resources Management and Biodiversity Conservation*. pp.177–192. Springer International Publishing AG, part of Springer Nature.

Ainley, D.G. (1995). 'Ashy Storm-petrel (*Oceanodroma homochroa*)'. In: A. Poole and F. Gill (eds.) *The Birds of North America*, No. 185. Philadelphia, PA: Academy of Natural Sciences and Washington, DC: American Ornithologists' Union.

Ainley, D.G. and Everett, W.T. (2001). 'Black Storm-Petrel (*Oceanodroma melania*), version 2.0'. In: A.F. Poole and F.B. Gill (Eds.). *The Birds of North America*. Ithaca, NY, USA: Cornell Lab of Ornithology.

Anderson, D.W. and Keith, J.O. (1980). 'The human influence on seabird nesting success: Conservation implications'. *Biological Conservation* 18: 65–80.

Anthony, A.W. (1925). 'The expedition to Guadalupe Island, Mexico, in 1922. The birds and mammals'. *Proceedings of the California Academy of Sciences* 14(4): 277–320.

Bancroft, G. (1927). 'Notes on the breeding coastal and insular birds of coastal Lower California'. *Condor* 29: 188–195.

Carter, H.R., Sowls, A.L., Rodway, M.S., Wilson, U.W., Lowe, R.W., McChesney, G.J., Gress, F. and Anderson, D.W. (1995). 'Population size, trends, and conservation problems of the double-crested cormorant on the Pacific Coast of North America'. *Colonial Waterbirds* 18: 189–207.

CONANP (Comisión Nacional de Áreas Naturales Protegidas). (2000). *Programa de Manejo de la Reserva de la Biósfera El Vizcaíno*. Mexico City: SEMARNAP.

CONAPESCA (Comisión Nacional de Acuacultura y Pesca). (2014). 'Anuario estadístico de acuacultura y pesca 2014'. <https://www.gob.mx/conapesca/documentos/anuario-estadistico-de-acuacultura-y-pesca>.

Croxall, J.P., Butchart, S.H.M., Lascelles, B., Stattersfield, A.J., Sullivan, B., Symes, A. and Taylor, P. (2012). 'Seabird conservation status, threats ad priority actions: a global assessment'. *Bird Conservation International* 22(1): 1–34.

Van Denburgh, J. (1924). 'The birds of Todos Santos Islands'. *Condor* 26: 67–71.

DOF. Diario Oficial de la Federación. (2016). *Decreto por el que se declara Área Natural Protegida, con el carácter de reserva de la biosfera, la región conocida como Islas del Pacífico de la Península de Baja California*, *Diario Oficial de la Federación*. <http://www.dof.gob.mx/nota_detalle.php?codigo=5464451&fecha=07/12/2016>.

Drost, C.A. and Lewis, D.B. (1995). 'Xantu's Murrelet (*Synthliboramphus hypoleucus*)'. In: A.F. Poole and F.B. Gill (eds.) *The Birds of North America*, No. 164. Philadelphia, PA: Academy of Natural Sciences and Washington, DC: American Ornithologists' Union.

Everett, W.T. and Anderson, D.W. (1991). 'Status and conservation of the breeding seabirds on offshore Pacific islands of Baja California and the Gulf of California'. *ICBP Technical Plublication* 11: 115–139.

Gress, F. (1995). 'Organochlorines, Eggshell Thinning, and Productivity Relationships in Brown Pelicans Breeding in the Southern California Bight'. PhD thesis. Davis, California: University of California.

Gress, F., Risebrough, R.W., Anderson, D.W., Kiff, L.F. and Jehl, J.R. (1973). 'Reproductive failures of double-crested cormorants in southern California and Baja California'. *Wilson Bulletin* 85: 197–208.

Grinnell, J. and Daggett, F. (1903). 'An ornithological visit to Los Coronados Islands, Lower California'. *Auk* 20: 27–37.

Hernández Díaz, J.A. and Salazar Gómez, E.N. (2011). 'Blue-footed Booby (*Sula nebouxii*), Version 1.0'. In: T.S. Schulenberg (ed.) *Neotropical Birds Online*. Ithaca: Cornell Lab of Ornithology. <https://doi.org/10.2173/nb.bfoboo.01>.

Hobson, K.A. (2013). 'Pelagic Cormorant (*Phalacrocorax pelagicus*)'. In: P.G. Rodewald (ed.) *The Birds of North America*. Ithaca: Cornell Lab of Ornithology. < https://birdsna.org/Species-Account/bna/species/pelcor/introduction>.

Howell, A.B. (1917). 'Birds of the islands off the coast of southern California'. *Pacific Coast Avifauna* 12: 1–127.

Huey, L.M. (1927). 'Northernmost breeding station of the Heermann Gull on the Pacific Ocean, and other notes from San Roque Island, Lower California'. *Condor* 29: 205–206.

Jehl, J.R. (1973). 'Studies of a declining population of brown pelicans in northwestern Baja California'. *Condor* 75(1): 69–79.

Jehl, J.R. (1976). 'The northernmost colony of Heermann's gull'. *Western Birds* 7: 25–26.

Jehl, J.R. (1977). 'An annotated list of birds of Islas los Coronados, Baja California, and adjacent waters'. *Western Birds* 8: 91–101.

Jehl, J.R. and Bond, S.I. (1975). 'Morphological variation and species limits in murrelets of the genus *Endomychura'*. *Transactions of the San Diego Society of Natural History* 18: 9–23.

Jones, H.P. and Kress, S.W. (2012). 'A review of the world's active seabird restoration projects'. *Journal of Wildlife Management* 76(1): 2–9.

Kaeding, H.B. (1905). 'Birds from the west coast of Lower California and adjacent islands'. *Condor* 7: 105–111.

Keitt, B.S., Wilcox, C., Tershy, B.R, Croll, D.A. and Donlan, C.J. (2002). 'The effect of feral cats on the population viability of black-vented shearwaters (*Puffinus opisthomelas*) on Natividad Island, Mexico'. *Animal Conservation* 5: 217–223.

Keitt, B., Tershy, B.R. and Croll, D.A. (2003). 'Breeding biology and conservation of the black-vented shearwater *Puffinus opisthomelas'*. *Ibis,* 145: 673-680.

Keitt, B. (2005). 'Status of Xantus's murrelet and its nesting habitat in Baja California, Mexico'. *Marine Ornithology* 33: 105–114.

Latofski-Robles, M., Méndez-Sánchez, F., Aguirre-Muñoz, A., Jáuregui-García, C., Koleff-Osorio, P., González-Martínez, A.I., Born-Schmidt, G., Bernal-Stoopen, J. and Rendón-Hernández, E. (2019). 'Mexico's island biosecurity programme: collaborative formulation and implementation*'*. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 484–488. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

McCarthy, M. A. (2007). *Bayesian Methods for Ecology*. New York, USA: Cambridge University Press.

McChesney, G. and Tershy, B. (1998). 'History and status of introduced mammals and impacts to breeding seabirds on the California channel and northwestern Baja California Islands'. *Colonial Waterbirds* 21(3): 335–347.

Mellink, E. (2001). 'History and status of colonies of Heermann's gull in Mexico'. *Waterbirds: The International Journal of Waterbird Biology* 24(2): 188–194.

Osburn, P.I. (1909). 'Notes on the birds of Los Coronados Islands, Lower California'. *Condor* 11: 134–138.

Palacios, E. and Mellink, E. (2000). 'Nesting waterbirds on Islas San Martín and Todos Santos, Baja California'. *Western Birds* 31: 184–189.

Sowls, A.L., DeGange, A.R., Nelson, J.W. and Lester, G.S. (1980). *Catalog of California Seabird Colonies*, *Biological Services Program*. Washington, D.C.: U.S. Fish and Wildlife Service (FWS/OBS-80/37).

Townsend, C.H. (1923). 'Birds collected in Lower California'. *Bulletin of the American Museum of Natural History* 28: 1–26.

Van Rossem, A. (1915). 'Notes on murrelets and petrels'. *Condor* 17: 74–78.

Walsh, P. M., Halley, D. J., Harris, M. P., del Nevo, A., Sim, I. M. W. and Tasker, M. L. (1995). *Seabird Monitoring Handbook for Britain and Ireland*. Norfolk, United Kingdom: The Seabird Group.

Wilcox, C., Van Sebille, E. and Hardesty, B. D. (2015). 'Threat of plastic pollution to seabirds is global, pervasive, and increasing'. *Proceedings of the National Academy of Sciences* 112 (38): 11899–11904.

Wolf, S. (2002). 'The Relative Status and Conservation of Island Breeding Seabirds in California and Northwest Mexico'. MSc thesis. Santa Cruz: University of California.

Wolf, S., Keitt, B., Aguirre-Muñoz, A., Tershy, B., Palacios, E. and Croll, D. (2006). 'Transboundary seabird conservation in an important North American marine ecoregion', *Environmental Conservation*, 33(4): 294–305.

Whitworth, D.L., Carter, H.R., Palacios, E. and Gress, F. (2018). Breeding of Craveri's Murrelet *Synthliboramphus craveri* at four islands off west-central Baja California, México. *Marine Ornithology* 46: 117-124.

Wright, H.W. (1913). 'The birds of San Martín Island, Lower California'. *Condor* 15: 207–210.

The recovery of seabird populations on Ramsey Island, Pembrokeshire, Wales, following the 1999/2000 rat eradication

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Abstract Ramsey Island, 259 ha, is ca. 1 km off the Pembrokeshire coast, south-west Wales. The eradication of brown rats (*Rattus norvegicus*) was successfully completed in the winter 1999/2000 using a ground-based bait station operation. The pre-eradication survey using tape playback estimated the Manx shearwater (*Puffinus puffinus*) population to be 849 pairs. These surveys were repeated in 2007, 2012 and 2016. Each survey showed the Manx shearwater population had increased, reaching 4,796 pairs in 2016 with birds spreading from previously known breeding locations. European storm petrels (*Hydrobates pelagicus*) were first recorded breeding on Ramsey Island in 2008 with up to 12 pairs in 2016 (a minimum estimate based on accessible survey areas). Other species have also shown improvements to population estimates and range since the rat eradication. This evidence shows that there can be little doubt that the presence of brown rats on Ramsey played a significant role in suppressing breeding numbers and limiting the breeding range of seabirds on the island and the positive results following the successful eradication are now being seen.

Keywords: brown rat, European storm petrel, *Hydrobates pelagicus*, Manx shearwater, *Puffinus puffinus*, *Rattus norvegicus*, survey

INTRODUCTION

Rats are known to have devastating effects on seabird and land bird populations by predation of eggs, chicks and adults which reduces breeding success, recruitment, population size and distribution. They have caused extinctions of birds on numerous islands throughout the world (Moors & Atkinson, 1984, Atkinson, 1985, Towns, et al., 2006; Jones, et al., 2008). Smaller burrowing seabirds are recognised as the species most affected by invasive rats (Jones, et al., 2008; Towns, et al., 2011). The eradication of rats from seabird islands is recognised as one of the most important tools in avian conservation in recent times, with significant long-term restoration benefits such as increased productivity and populations sizes and establishment of new, or return of previously locally extinct, seabird species being achieved (Atkinson, 1985; Moors, et al.,1992; Lock, 2006; Ratcliffe, et al., 2009; Booker & Price, 2010, Bourgeois, et al., 2013; Le Corre, et al., 2015).The protection and enhancement of UK seabird breeding habitat has been recognised as an important conservation priority, including under international conservation agreements (Brooke, et al., 2007; Ratcliffe, et al., 2009; Dawson, et al., 2015; Thomas, et al., 2017). Over 400 islands around the world have been successfully cleared of rats, including twelve in the United Kingdom, with a subsequent increase in bird populations (Thomas & Taylor, 2002; Towns & Broome, 2003; Jones, et al., 2008; Howald, et al., 2007, DIISE, 2015, Thomas, et al., 2017).

A feasibility study of eradicating brown rats (*Rattus norvegicus*) from Ramsey Island was completed in 1998 and led to the ground-based eradication in autumn 1999. Documenting the recovery of bird species on islands that have had invasive mammals removed is becoming increasingly important. RSPB has been monitoring bird populations on Ramsey Island since 1992. Due to difficulty in accessing natural burrows, between 2013 and 2016, RSPB constructed a man-made seabird habitat using artificial burrows with the aim to establish a Manx shearwater (*Puffinus puffinus*) colony that could be used to monitor productivity, recruitment and adult survival. This paper details the changes to the Manx shearwater population on Ramsey Island, including within the man-made habitat, and the subsequent colonisation of the island by European storm petrel (*Hydrobates pelagicus*) following the eradication of brown rats.

STUDY AREA AND METHODS

Study site

Ramsey Island, 259 ha (5°20′W, 51°51′N), is located about 1 km off the Pembrokeshire coast, south-west Wales (Fig. 1). It is a nature reserve owned and managed by the RSPB. Ramsey Island is approximately 3.2 km long and 1.6 km across at its widest point and is surrounded by coastal cliffs which are particularly high and steep on the western side of the island. There is also a number of small islets (including a chain of islets from the southern end) and caves around the coastline. The coastline of the island is made up of exposed rocky shores with a small number of sandy coves. The top of the island is gently rolling and is dominated by two prominent peaks (Carn Ysgubor 101 m and Carn Llundian 136 m). The island supports three main habitats; acid grassland, bracken-dominated grassland and coastal heathland (Doncaster, 1981; Hurford & Evans, 2006; CCW, 2008). The heathland and maritime grassland communities are of conservation importance in Wales (JNCC, 2001; Hurdford & Evans, 2006; CCW, 2008). The rush-pasture fields are grazed by rabbits (*Oryctolgus cuniculus*), ponies (*Equus caballus*) and sheep (*Ovis aries*) as part of the management to support wildlife, particularly choughs (*Pyrrhocorax pyrrhocorax*) (Doncaster, 1981; Long, 2003). The bank vole (*Myodes glareolus*) and common shrew (*Sorex araneus*) are also present on the island.

Fig. 1 Location of Ramsey Island, Wales.

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The island is part of the Pembrokeshire Coast National Park and has a range of designations including as a Site of Special Scientific Interest (SSSI), National Nature Reserve (NNR), Important Bird Area (IBA), Special Protection Area (SPA) and Marine Special Area of Conservation (MSAC) (JNCC, 2001; CCW, 2008; Hayhow, et al., 2016). Ramsey holds important breeding populations of razorbill (*Alca torda*), guillemot (*Uria aalge*), kittiwake (*Rissa tridactyla*), Manx shearwater, chough and wheatear (*Oenanthe oenanthe*) (JNCC, 2001; Johnstone, et al., 2011). Ramsey Island is also an important breeding site for Atlantic grey seals (*Halichoerus grypus*).

The island is popular with visitors who are interested in the seabirds, land birds, flora and history. These visitors travel to the island on small passenger vessels from St David's between April and October. There is a jetty and several buildings on the island, including the warden's home and information shelter.

Manx shearwaters have been recorded on Ramsey Island since the 18th century (Mathew, 1894; Holloway, 2010; Lovegrove, et al., 2010). Historical reports and more recent seabird monitoring on Ramsey recorded declines in Manx shearwaters and other seabirds (Mathew, 1894; Humpridge & Bullock, 1999; Lovegrove, et al., 2010). These declines were attributed in part to the presence of brown rats and predation on eggs and chicks (Humpridge & Bullock, 1999; Lovegrove, et al., 2010).

It is not known when brown rats became established on Ramsey Island; but this was likely to have occurred more than two hundred years ago from an early shipwreck.

Brown rat eradication

The eradication of brown rats was completed as a ground-based operation using rodenticide cereal blocks in protective bait stations to reduce risk to non-target species. A 50 m ×50 m grid was established in autumn 1999. Bait stations were made from 500 mm lengths of corrugated plastic drainage pipe staked into position using wire. A total of 1,260 stations were placed on the main island and offshore stacks. The poisoning operation ran from 11 January 2000 to 10 March 2000. Two 24 g blocks of cereal-based rodenticide bait (Neosorexa™, active ingredient 0.005% difenacoum, manufactured by Sorex Ltd) were placed in each station on the main island and ten blocks on the offshore stack throughout the poisoning programme and replaced as required when eaten by rats, non-target species and/or damaged by weather.

The stations on the main island were checked daily, but stations on the offshore stacks were checked when sea conditions allowed. Bait take was recorded by bait station number and the species believed to have consumed or removed the bait.

From 1 March to 15 March 2000, monitoring stations were established around the island next to and in-between the bait stations. Chew sticks, chocolate blocks and small pieces of candle were used. Sand and mud areas on the island were checked for rat foot prints and burrows and rat runs were checked for fresh activity. All monitoring points were individually numbered and any evidence of activity (i.e. teeth marks or foot prints) was recorded by station number and the species believed to have consumed or marked the monitoring item. Each monitoring site was checked regularly, either separately or together with the poisoning bait station grid. Any rat and non-target species sign found on detection devices was recorded.

Manx shearwater breeding population survey

Earlier surveys on Ramsey Island had shown that Manx shearwaters only occur in a narrow strip around the coast of the island on hills Carn Llundain and Carn Ysgubor (I.D. Bullock, unpublished data; Perkins, et al., 2017). For burrow counting and sampling purposes, Ramsey Island was divided into 42 sub-areas by topographical features. A full count of suitable burrows (i.e. more than 0.7 m in length and not doubling back to the surface) in these 42 sub-areas was completed in 1999, 2007, 2012 and 2016.

Estimation of the numbers of Manx shearwaters on Ramsey Island was based on playback of recorded calls (Brooke, 1978; Smith, et al, 2001; Perkins, et al., 2017). This method relies on the fact that if a male Manx shearwater call is played down a sample of burrows during the incubation period, most incubating males, but no incubating females will respond to that call (Smith, et al., 2001). For a given number of breeding pairs, it is then possible to establish the number of males that respond to recorded calls. From this, using the following formula it is possible to estimate the number of breeding pairs in the burrows on the island.

$$\textit{Breeding pair} = \textit{No. of burrows} \times \frac{\textit{No. Responding}}{\textit{No. Sampled}} \times \frac{1}{\textit{Response Rate}}$$

The response rate for Manx shearwaters was calculated by Bullock in 1999 (0.409) and was based on a study set of 13 burrows (Humpridge & Bullock 1999). Alternative response rates were available from Skomer (0.43, Smith, et al., 2001) and Skokholm (0.505, Brooke, 1978) or the seabird monitoring handbook (0.505, Walsh, et al., 1995). The Ramsey response rate of 0.409 was used in 2007 as it allows direct comparison to the earlier survey on the island. The response rates were recalculated for the 2012 survey (to 0.4625) using methods developed by Murray, et al. (2003), Newton, et al. (2004) and Perrins, et al. (2012). The response rates were recalculated for the 2016 survey (to 0.845) which used dual-sex calls which had been shown to give a more reliable correction factor (Perkins, et al., 2017).

A recording of male Manx shearwater calls was played down 20% of burrows in each sub-area during the main incubation period unless the sub-area contained fewer than 50 burrows up to 2016 and then duetting male and female calls were used for 2016 (Perkins, et al., 2017). In those cases, the recording was played in all burrows. Recordings were played at natural volumes ('normal' Manx shearwater call volumes as heard from the burrows that were set 'by ear' before 2016 and by a decibel reader in 2016) within 30 cm of the burrow entrance for up to 25 seconds. Playback of calls was carried out in the day and responses, or lack thereof, were recorded. Playback was undertaken between the end of May and mid-June at a time when all eggs laid should be being incubated by one adult (Brooke, 1990).

Between 2013 and 2016 nearly 100 artificial nest boxes were established on the island. These burrows are the same design as those developed in New Zealand for fluttering shearwaters (*Puffinus gavia*) by Bell (1995) and recommended for burrow-nesting petrel and shearwater species (Gummer, et al., 2014). These artificial burrows were put in place to provide easily accessible study burrows for tracking studies and to determine productivity and population parameters such as survival and recruitment (Morgan, 2012; Kirk, et al., 2013).

European storm petrel breeding population survey

Surveys of suitable storm petrel habitat (i.e. stone walls, rock tumbles and scree) on Ramsey Island were undertaken using playback in 2008, 2010, 2012 and 2016. A recording of a male European storm petrel was played close to a suspected site and a reply listened for (Ratcliffe, et al., 1998; Gilbert, et al., 1999; Mayhew, et al., 2000). Burrow entrances that had responses were mapped using GPS.

RESULTS

Brown rat eradication

Bait acceptance was good, with rats accounting for 165 kg of bait consumed. As the LD_{50} for a 250-g brown rat is 9 g and the mean (± SE) bait take by rats was 81.6 ± 0.7 g (3.4 ± 0.02 blocks, range 0–30 blocks) the rat population on Ramsey Island was estimated to be between 1,850 and 5,400 rats). The bait take pattern was typical of other rat eradication operations; very high in the immediate five to ten days after original baiting and dropping to a relatively low level 21 days after original baiting. Bait take dropped to zero by day 41 after the original baiting (Fig. 2). The rats were widely distributed across the island, but the density was not even, as shown by the distribution of bait take (Fig. 3). Rats were present in all coastal areas and in highest numbers within the central and northern areas of the island.

Rabbits interfered with the bait stations between days 12 and 26 of the operation, with a number of carcasses being collected. Bait stations were modified by halving the entrance size to prevent access by rabbits and this greatly reduced their interference levels. Carrion crows (*Corvus corone*), ravens (*Corvus corax*) and herring gulls (*Larus argentatus*) also interfered with the stations from day 25 after the birds had learnt to reach into the stations to get access to the bait. Eight crows and three raven carcasses were located but no herring gull deaths were recorded. The bait stations were further modified by extending the length from 500 mm to 750 mm which reduced crow, raven and gull interference to almost nil. Crows were observed working in pairs to remove wires; one pulling the wire out while the other stood on top of the station to hold it in place, to get access to the bait. Access to the bait by the bank voles could not be prevented and 30 dead voles were found. A small captive population was maintained during the eradication as a precaution and was released after the poison had been removed. Voles and vole sign on monitoring tools were recorded throughout the eradication.

Monitoring for rat presence continued for two years after the end of the poisoning operation. No rats or sign were detected. The rat-free status for Ramsey Island was declared in March 2002.

Manx shearwater breeding population survey

The number of Manx shearwater burrows on Ramsey Island totalled 13,800 burrows in 1999, 14,970 burrows in 2007, 12,302 burrows in 2012 and 12,319 in 2016 (Humpridge & Bullock, 1999; Morgan & Morgan, 2008; Morgan & Morgan, 2013; Morgan & Morgan, 2017).

The Manx shearwater breeding population size increased 3-fold and 5-fold, 8 and 17 years after the rat eradication respectively (Table 1).

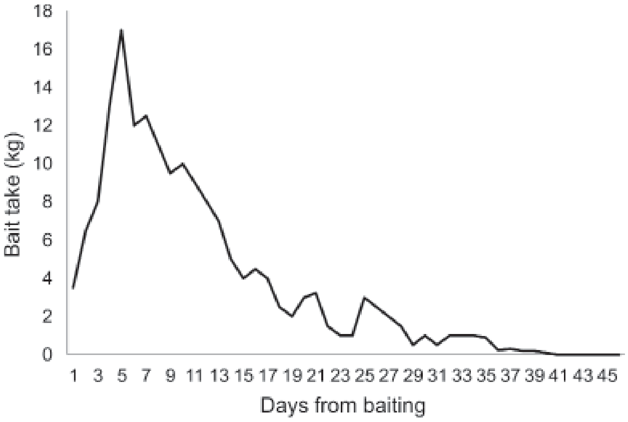

Fig. 2 Bait take by rats during the brown rat (*Rattus norvegicus*) eradication on Ramsey Island, Wales, 1999/2000.

The distribution of Manx shearwaters remained largely unchanged between 1999 and 2007 censuses, but the range spread between the 2007, 2012 and 2016 censuses (Fig. 4). There have also been significant increases in the population within the distribution with new areas recorded in 2007, 2102 and 2016 that previously had no responses recorded in 1999 (Fig. 4).

Burrow density is greatest along the west, north and north-east coasts and on the hills (Fig. 4). Interestingly in a section at the northern end of Ramsey Island there was no response to the recordings despite a high number of suitable burrows available for Manx shearwaters (n = 2,247) in 1999 or 2007. This area showed a low level of response in 2012 and higher in 2016.

A prospecting pair of Manx shearwaters was recorded in one of the artificial burrows in 2015. Two pairs nested successfully in the artificial burrows in 2016 and seven pairs were recorded incubating eggs in April 2017.

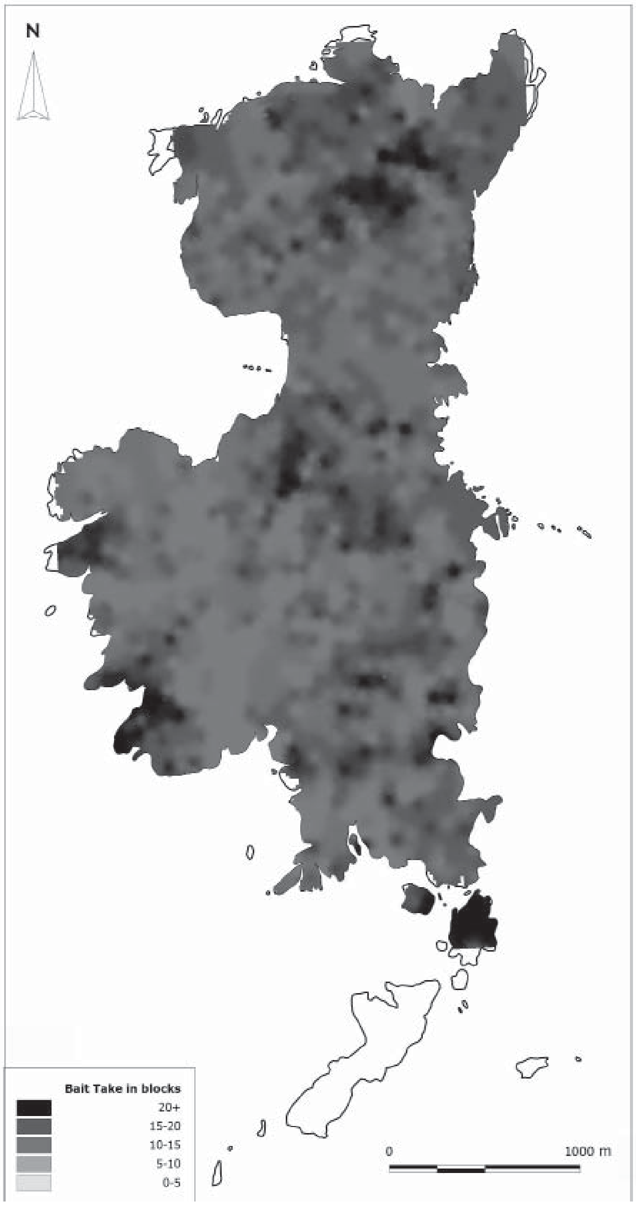

Fig. 3 Distribution of bait take during the brown rat (*Rattus norvegicus*) eradication on Ramsey Island, Wales, 1999/2000.

Table 1 The total number of burrows, response rate used, total number of burrows sampled using playback, total number of responses and total number of breeding pairs of Manx shearwater *Puffinus puffinus* on Ramsey Island between 1999 and 2016.

Year	Total number of burrows	Response rate used	Total number sampled using playback	Total number of responses	Total number of breeding pairs
1999	13,800	0.409	2,760	74	905
2007	14,970	0.409	3,190	208	2,387
2012	12,302	0.4625	2,788	402	3,835
2016	12,319	0.845	2,860	941	4,796

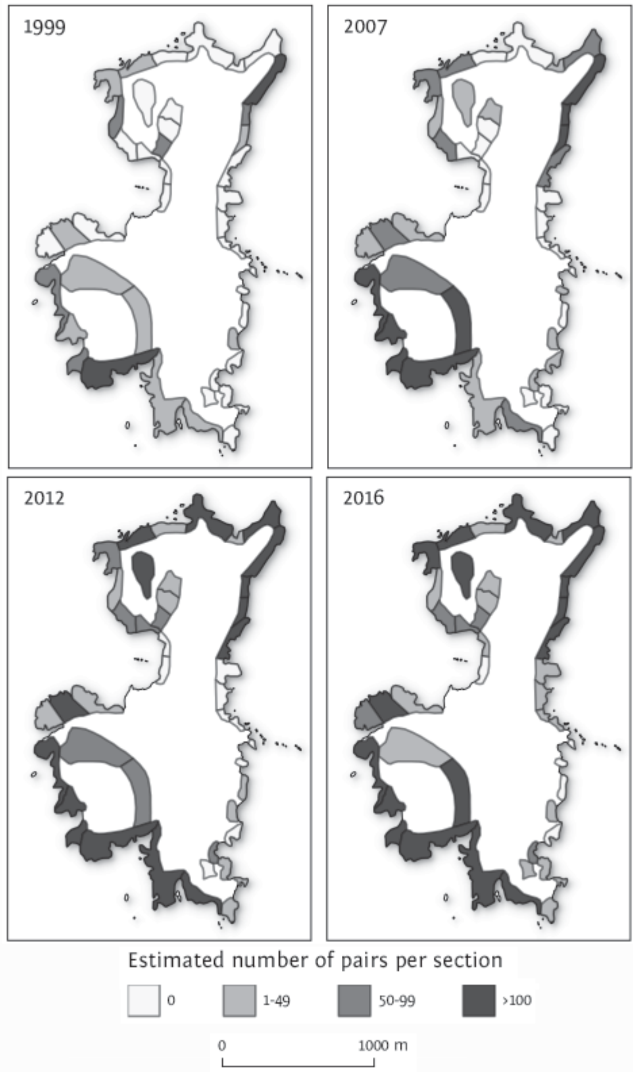

Fig. 4 Distribution and density of Manx shearwaters (*Puffinus puffinus*) from the full surveys in 1999, 2007, 2012 and 2016 on Ramsey Island, Wales.

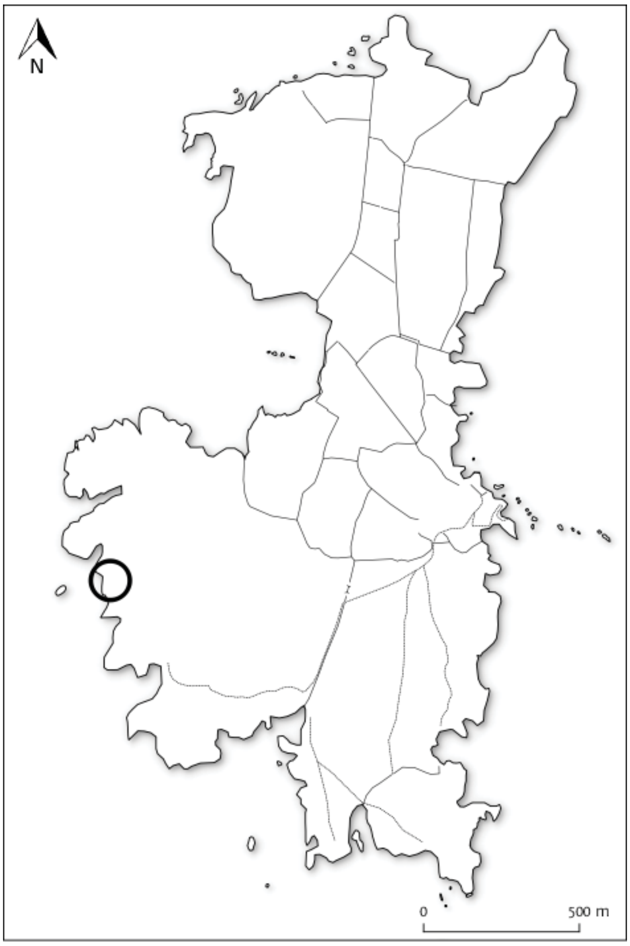

Fig. 5 Location of European storm petrel breeding sites (in black circle) on Ramsey Island, Wales, 2016.

European storm petrel breeding population survey

The first storm petrel breeding burrows were detected in 2008 (4 pairs). By 2016, the numbers had increased to 12 breeding burrows (Table 2).

DISCUSSION

The Ramsey Island brown rat eradication was one of the first eradications undertaken by the RSPB and a number of important lessons were learnt that helped with the planning and implementation of later eradications on Lundy Island, St Agnes and Gugh, Isles of Scilly and the Shiant Isles. Bait station design was adapted to suit local conditions and local non-target species.

The Ramsey Island operation was the first island-wide eradication that used a rodenticide containing the active ingredient difenacoum proving that this rodenticide could be used to successfully target all brown rats on an island. This toxin has since been used on a number of island eradication operations in the UK and around the world (Howald, et al., 2007; Jones, et al., 2008). Since the eradication of rats, there have been biosecurity protocols put in place to prevent a re-incursion on Ramsey Island and to outline how to respond if rats are ever detected on the island. It is important that these measures are maintained indefinitely.

Ramsey Island has seen dramatic changes since the removal of brown rats, not least the increase in the distribution and density of Manx shearwaters. The number of Manx shearwaters has multiplied by five times between

Table 2 The number of burrows of European storm petrels *Hydrobates pelagicus* on Ramsey Island between 2008 and 2016.

Year	Number of breeding burrows
2008	4
2009	Not surveyed
2010	6
2011	Not surveyed
2012	5
2013	5
2014	5
2015	8
2016	12

1998 and 2016 to almost 5,000 breeding pairs, representing a 560% increase. This proves that the brown rats were having a significant role in suppressing the number of breeding pairs on the island and their range across the island and provides more evidence that invasive rats have significant impacts on seabird populations on islands (Atkinson & Moors, 1984; Atkinson, 1985; Towns, et al., 2006; Jones, et al., 2008). A similar pattern was observed on Lundy Island following the rat eradication operation in 2004 (Brown, et al., 2011). It is suspected that although increased productivity will have occurred on Ramsey Island, given that the Manx shearwater does not breed until five or six years of age (Brooke, 1990), much of this increase may be due to immigration from the extremely large neighbouring colonies on Skomer and Skokholm. This theory was confirmed by the capture of an adult that had been ringed as a chick on Skomer in 1993 which was on its way to feed a chick on Ramsey Island in 2017 (GM, pers. obs.).

The greatest increases have occurred within the existing sub-colonies, but there has also been expansion into new areas. Nine sections that showed nil response in 1999 and 2007 contained breeding birds in 2012 and a further five new sections were occupied in 2016. There is limited habitat available on Ramsey Island away from the coastal areas. However, restoration of drystone walls, former rabbit warrens and artificial burrows have all provided more nest-sites. However, the presence of rabbits may affect the distribution of Manx shearwaters on Ramsey Island. Competition for burrows with a small number of birds may account for restricted range and densities in specific locations on the island. The development of an artificial study colony on Ramsey Island has proved successful with up to seven birds nesting in the man-made burrows in 2017, of which five successfully fledged chicks.

European storm petrels have also started breeding on the island for the first time since records began. Although storm petrels are known to breed on two offshore islands, the Bishops and Clerks (163 apparent occupied sites in 2017; G.M., pers. obs.), it was not until 2008 that they were confirmed on Ramsey Island itself. Six birds were recorded breeding in 2012 and this increased to 12 pairs in 2016. It is important to note that these estimates are the minimum number of storm petrels present on the island as not all adults may respond to the recorded calls. This has been shown to be the case in a number of other studies (Insley, et al., 2002; Brown, 2006; Hounsome, et al., 2006) and, as these studies have also shown that correction factors for storm petrels are known to be highly variable between sites and even between years, the use of recorded calls and corrections have not been used to estimate the current Ramsey Island storm petrel population. As the population increases, an island-specific correction factor will be calculated for Ramsey Island and used to estimate the population size in the future. Currently, the minimum estimate is used (i.e. the known response to taped calls).

However, the basic playback-response method is widely used, standardised and is comparable between years and across sites. It is also a low-impact method, completed during the day, and provides spatial information on breeding burrows. The storm petrel population on Ramsey Island is likely to increase into a range of available habitat including drystone walls, rabbit burrows and rock tumbles.

The success on Ramsey Island provided valuable information and techniques for later eradication operations in the UK, particularly those with important non-target species. It also showed that ground-based eradication techniques developed in New Zealand could be adapted and used on islands in the UK, and Ramsey Island serves as a good example of the significant long-term benefits that can be achieved through short-term investment.

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REFERENCES

Atkinson, I.A.E. (1985). 'The Spread of Commensal Species of *Rattus* to Oceanic Islands and their Effects on Island Avifaunas'. In: P.J. Moors (ed.) *Conservation of Island Birds*, pp. 35–81. ICBP Technical Publication 3. Cambridge, UK: International Council for Bird Preservation (ICBP).

Bell, B. (1995). 'Translocation of Fluttering Shearwaters: Developing a Method to Re–establish Seabird Populations'. In: M. Serena (ed.) *Reintroduction Biology of Australian and New Zealand Fauna,* pp. 143–156. Australia: Surrey Beatty and Sons.

Booker, H. and Price, D. (2010). 'Manx shearwaters on Lundy: A study of population and distribution change from 2001 to 2008'. *Journal of the Lundy Field Society* 2: 105–112.

Bourgeois, K., Ouni, R., Pascal, M., Dromzée, S., Fourcy, D., and Abiadh, A. (2013). 'Dramatic increase in the Zembretta Yelkouan shearwater breeding population following ship rat eradication spurs interest in managing a 1500-year old invasion'. *Biological Invasions* 15: 475–482.

Brooke, M. (1978). 'Sexual differences in the voice and individual vocal recognition in the Manx shearwater (*Puffinus puffinus*)'. *Animal Behaviour* 26: 622–629.

Brooke, M. (1990). *The Manx Shearwater*. London: Poyser.

Brooke, M., Hilton, G. and Martins, T. (2007). 'Prioritising the world's islands for vertebrate-eradication programmes'. *Animal Conservation* 10: 380–390.

Brown, A., Price, D., Slader, P., Booker, H., Lock, L. and Deveney, D. (2011). 'Seabirds on Lundy: their current status, recent history and prospects for the restoration of a once important bird area'. *British Birds* 104: 139–158.

Brown, J. (2006). 'Census of European storm-petrels *Hydrobates pelagicus* on Skomer Island'. *Atlantic Seabirds* 8: 21–30.

CCW (Countryside Council for Wales). (2008). *Core Management Plan Including Conservation Objectives for St David's Special Area of Conservation*. Bangor, UK: Countryside Council for Wales.

DIISE. (2015). 'The Database of Island Invasive Species Eradications, developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand'. <http://diise.islandconservation.org>.

Dawson, J. Oppel, S., Cuthbert, R., Homes, N., Bird, J., Butchart, S., Spatz, D. and Tershey, B. (2015). 'Prioritizing islands for the eradication of invasive vertebrates in the United Kingdom overseas territories'. *Conservation Biology* 29: 143–153.

Doncaster, C. (1981). ‘The spatial distribution of ants' nests on Ramsey Island, South Wales’. *Journal of Animal Ecology* 50: 195–218.

Gilbert, G., Gibbons, D. and Evans, J. (1999). *Bird Monitoring Methods B: A Manual of Techniques for Key UK Species*. Sandy, Bedfordshire, UK: RSPB, BTO, JNCC, ITE & The Seabird Group.

Gummer, H., Taylor, G. and Collen, R. (2014). *Field Guidelines for Burrow-nesting Petrel and Shearwater Translocations—a Companion Guide to the Seabird Translocation Best Practice Documents*. Wellington: Department of Conservation.

Hayhow, D., Burns, F., Eaton, M., Al Fulaij, N., August, T., Babey, L., Bacon, L., Bingham, C., Boswell, J., Boughey, K., Brereton, T., Brookman, E., Brooks, D., Bullock, D., Burke, O., Collis, M., Corbet, L., Cornish, N., De Massimi, S., Densham, J., Dunn, E., Elliott, S., Gent, T., Godber, J., Hamilton, S., Havery, S., Hawkins, S., Henney, J., Holmes, K., Hutchinson, N., Isaac, N., Johns, D., Macadam, C., Matthews, F., Nicolet, P., Noble, D., Outhwaite, C., Powney, G., Richardson, P., Roy, D., Sims, D., Smart, S., Stevenson, K., Stroud, R., Walker, K., Webb, J., Webb, T., Wynde, R. and Gregory, R. (2016). *State of Nature 2016*. UK: The State of Nature partnership.

Holloway, S. (2010). *The Historical Atlas of Breeding Birds in Britain and Ireland 1875–1900*. London, UK: Poyser Series, A&C Black.

Hounsome, M., Insley, H., Elliott, S., Graham, K. and Mayhew, P. (2006). ‘Monitoring European storm-petrels *Hydrobates pelagicus*: a comparison of the results provided by mark/recapture and tape response methods’. *Atlantic Seabirds* 8: 5–20.

Howald, G., Donlan, C., Galvan, J., Russell, J., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershey, B. (2007). ‘Invasive rodent eradication on islands’. *Conservation Biology* 21: 1258–1268.

Humpridge, R. and Bullock, I. (1999). ‘Estimation of the breeding Manx shearwater population on Ramsey Island’. *RSPB Ramsey Island Annual Report* 1999–2000. St Davids, Pembrokeshire, UK: RSPB.

Hurdford, C. and Evans, S. (2006). ‘Monitoring Coastal Heaths at St David’s’. In: C. Hurford and M. Schnieder (eds.) *Monitoring Nature Conservation in Cultural Habitats: A Practical Guide and Case Studies*, pp. 157–168. Netherlands: Springer.

Insley, H., Hounsome, M., Mayhew, P. and Chisholm, K. (2002). ‘A preliminary comparison of population estimation by tape response and mark/recapture methods for storm petrels *Hydrobates pelagicus* in two contrasting Scottish colonies’. *Ringing & Migration* 21: 75–79.

JNCC (Joint Nature Conservation Committee). (2001). *Ramsey and St David’s Peninsula Coast: SPA Description. Special Protection Areas under the EC Birds Directive*. <http://jncc.defra.gov.uk/default.aspx?page=2063>. Accessed 16 June 2017.

Johnstone, I., Thorpe, R. and Noble, D. (2011). *The State of Birds in Wales 2011*. Cardiff, Wales: RSPB Cymru.

Jones, H., Tershey, B., Zavaleta, E., Croll, D., Keitt, B., Finkelstein, M. and Howard, G. (2008). ‘Review of the global severity of the effects of invasive rats on seabirds’. *Biological Conservation* 22: 16–26.

Kirk, H., Morgan, G. and Guilford, T. (2013). ‘Ramsey Island Manx shearwater geolocation study: Field report 2012’. *Birds in Wales* 10: 42–47.

Le Corre, M., Danckwerts, D.K., Ringler, D., Bastien, M., Orlowski, S., Morey Rubio, C., Pinaud, D. and Micol, T. (2015). ‘Seabird recovery and vegetation dynamics after Norway rat eradication at Tromelin Island, western Indian Ocean’. *Biological Conservation* 185: 85–94.

Lock, J. (2006). ‘Eradication of brown rats *Rattus norvegicus* and black rats *Rattus rattus* to restore breeding seabird populations on Lundy Island, Devon, England’. *Conservation Evidence* 3: 111–113.

Long, J. (2003). *Introduced Mammals of the World: their History, Distribution and Influence*. Wallingford, UK: CABI.

Lovegrove, R., Williams, I. and Williams, G. (2010). *Birds in Wales*. London, UK: Bloomsbury Publishing.

Mathew, M. (1894). *The Birds of Pembrokeshire and its Islands*. London: R.H. Porter.

Mayhew, P., Chisholm, K., Insley, H. and Ratcliffe, N. (2000). ‘A survey of storm petrels on Priest Island in 1999’. *Scottish Birds* 21: 78–84.

Moors, P.J. and Atkinson, I.A.E. (1984). ‘Predation on Seabirds by Introduced Animals and Factors Affecting its Severity’. In: J.P. Croxall, P.G.H. Evans and R.W. Schreiber (eds.) *Status and Conservation of the World’s Seabirds,* pp. 667–690. ICBP Technical Publication 2. Cambridge, UK: International Council for Bird Preservation (ICBP).

Moors, P., Atkinson, I. and Sherley, G. (1992). ‘Reducing the rat threat to island birds’. *Bird Conservation International* 2: 93–114.

Morgan, G. and Morgan, L. (2008). ‘A census of the Manx shearwater *Puffinus puffinus* on Ramsey Island RSPB Nature Reserve, Pembrokeshire in 2007’. *RSPB Ramsey Island Annual Report* 2007–2008. St Davids, Pembrokeshire, UK: RSPB.

Morgan, G. (2012). ‘The bird populations of Ramsey and Grassholm’. *British Birds* 105: 716–732.

Morgan, G. and Morgan, L. (2013). ‘A census of the Manx shearwater *Puffinus puffinus* on Ramsey Island RSPB Nature Reserve, Pembrokeshire in 2012’. *RSPB Ramsey Island Annual Report* 2012–2013. St Davids, Pembrokeshire, UK: RSPB.

Morgan, G. and Morgan, L. (2017). ‘A census of the Manx shearwater *Puffinus puffinus* on Ramsey Island RSPB Nature Reserve, Pembrokeshire in 2016’. *RSPB Ramsey Island Annual Report* 2016–2017. St Davids, Pembrokeshire, UK: RSPB.

Murray, S., Shewry, M., Mudge, G. and Spray, S. (2003). ‘A survey of Manx shearwaters *Puffinus puffinus* on Rum, Inner Hebrides in 2001’. *Atlantic Seabirds* 5: 89–100.

Newton, S., Thompson, K. and Mitchell, P. (2004). ‘Manx Shearwater *Puffinus puffinus’*. In: P. Mitchell, S. Newton, N. Ratcliffe and T. Dunn, (eds.) *Seabird Populations of Britain and Ireland*, pp. 63–80. London: T & AD Poyser.

Perkins, A., Douse, A., Morgan, G., Cooper, A. and Bolton, M. (2017). ‘Using dual-sex call improves the playback census method for a nocturnal burrow-nesting seabird, the Manx shearwater *Puffinus puffinus*’. *Bird Study* 64: 146–158.

Perrins, C., Wood, M., Garroway, C., Boyle, D., Oakes, N., Revera, R., Collins, P. and Taylor, C. (2012). ‘A whole-island census of the Manx shearwater *Puffinus puffinus* breeding on Skomer Island in 2011’. *Seabird* 25: 1–13.

Ratcliffe, N., Vaughan, D., Whyte, C. and Shepherd, M. (1998). ‘Development of playback census methods for storm petrels *Hydrobates pelagicus*’. *Bird Study* 45: 302–312.

Ratcliffe, N., Mitchell, I., Varnham, K., Verboven, N. and Higson, P. (2009). ‘How to prioritize rat management for the benefit of petrels: a case study of the UK, Channel Islands and Isle of Man’. *Ibis* 151: 699–708

Smith, S., Thompson, G. and Perrins, C. (2001). ‘A census of the Manx shearwater (*Puffinus puffinus*) on Skomer, Skokholm and Middleholm, west Wales’. *Bird Study* 48: 330–340.

Thomas, B. and Taylor, R. (2002). ‘A history of ground-based rodent eradication techniques developed in New Zealand, 1959–1993’. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 301–310. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Thomas, S., Brown, A., Bullock, D., Lock, L., Luxmoore, R., Roy, S., Stanbury, A. and Varnham, K. (2017). ‘Island restoration in the UK – past present and future’. *British Wildlife* 28: 231–242.

Towns, D. and Broome, K. (2003). ‘From small Maria to massive Campbell: Forty years of rat eradications from New Zealand islands’. *New Zealand Journal of Zoology* 30: 377–398.

Towns, D., Atkinson, I. and Daugherty, C. (2006). ‘Have the harmful effects of introduced rats on islands been exaggerated?’. *Biological Invasions* 8: 863–891.

Towns, D., Byrd, G., Jones, H., Rauzon, M., Russell, J. and Wilcox, C. (2011) ‘Impacts of Introduced Predators on Seabirds’. In: C. Mulder, W. Anderson, D. Towns and P. Bellingham (eds.) *Seabird Islands: Ecology, Invasions and Restoration*, pp. 56–90. New York: Oxford University Press.

Walsh, P., Halley, D., Harris, M., del Nevo, A., Sim, I., and Tasker, M. (1995). *Seabird Monitoring Handbook for Britain and Ireland*. Peterborough: Joint Nature Conservation Committee, Royal Society for the Protection of Birds, Institute of Terrestrial Ecology and Seabird Group.

Practical considerations for monitoring invasive mammal eradication outcomes

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Abstract Monitoring provides important evidence to evaluate if invasive mammal eradications on islands achieved conservation goals and to inform future allocation of conservation funds. However, monitoring can be costly, and compete with management actions for resources, so it is important that efforts are targeted. It can be difficult to obtain results of monitoring from previous projects, which puts future projects at a disadvantage when projecting possible conservation outcomes. In this short paper we discuss practical considerations for designing monitoring associated with an eradication programme. We focus on the ecological outcomes of invasive species eradication, as opposed to specifically monitoring if an eradication was successful or not. We identify major motivations for undertaking monitoring and present a decision tree intended to improve the efficiency of monitoring by supporting project managers determining when and what to monitor, and how to incorporate monitoring into project planning.

Keywords: decision tree, monitoring, post-eradication, project planning

INTRODUCTION

There is a substantial shortfall between the investment in conservation worldwide and the amount required to tackle the current biodiversity crisis (McCarthy, et al., 2012). It is well established that this necessitates prioritised, efficient allocation of resources, with evidence-based management (Sutherland, et al., 2004; Wilson, et al., 2006; Kapos, et al., 2008; Underwood, et al., 2008). Monitoring, defined as the collection and analysis of repeated field-based empirical measurements (Lindenmayer & Likens, 2010), provides this evidence. Over the past fifty years the eradication of invasive mammals from islands has developed into a reliable and effective conservation tool, resulting in substantial conservation gains (Veitch, et al., 2011; Jones, et al., 2016). There has been noteworthy progress in determining why, what, when and where to eradicate mammals from islands. Prioritisation of management is more robust and evidence-based than ever. When and how much to invest, how to balance different outcomes, and dealing with uncertainty of outcomes have all been addressed in recent literature (Donlan & Wilcox, 2007; Dawson, et al., 2015; Donlan, et al., 2015; Helmstedt, et al., 2016). Yet why, what, when and where to *monitor* following eradication is not always apparent. Recent assessments have highlighted that data on native species' responses to eradication are rare, often not quantitative, and not readily available through published sources, suggesting that monitoring, or reporting on monitoring, following eradication is uncommon (Jones, et al., 2016; Brooke, et al., 2017; Towns, 2018). Both eradication, and monitoring the outcomes that result, can be costly (e.g. Helmstedt, et al., 2016; Springer, 2016). Assuming both activities are being funded from the same combined budget, there is a potential trade-off between spending on eradication versus spending on monitoring (Possingham, et al., 2012). This paper discusses what to consider when designing monitoring of eradication projects. We focus on monitoring the wider ecological impacts of invasive species eradication, rather than short-term post-eradication monitoring for signs of invasive species that determines whether an eradication project has succeeded or failed. The paper incorporates inputs from the Island Invasives 2017 workshop: "Effective monitoring of response to eradications" attended by 60 conference participants. We aim to outline the main considerations for practitioners assessing the monitoring needs for projects they are involved in.

WHY, WHEN AND WHAT SHOULD WE MONITOR?

Possingham, et al. (2012) identified five separate benefits of long-term monitoring. Three of them—auditing the outcomes of a project (Case study 1), detecting unanticipated outcomes and researching mechanisms for those outcomes—have ecological benefits. The other two

CASE STUDY 1: AUDITING THE OUTCOMES OF RAT ERADICATION AT ANACAPA, CALIFORNIA, USA

Black rats (*Rattus rattus*) were successfully eradicated from the three islands of Anacapa in the Channel Islands, California USA, in 2001–2002. The goal of the eradication project was to improve seabird nesting habitat, and aid recovery of Scripps's murrelet (*Synthliboramphus scrippsi*, formerly Xantus's murrelet) and ashy storm-petrel (*Oceanodroma homochroa*) (NPS, 2000). The project was funded via oil-spill restoration resources, and an additional goal was to offset impacts that had occurred to these two species during the 1990 American Trader spill (ATTC, 2001). Monitoring included tracking artificial eggs (mimicking Scripps's murrelet eggs) before the eradication to quantify rat predation on this life history stage, and after the eradication to confirm the expected outcome of removing that impact (Jones, et al., 2005). Long-term monitoring of focal seabird species ensued for a decade including the hatching success, distribution and abundance of Scripps's murrelet on the island, which saw a three-fold increase in hatching success and expansion of nesting (Whitworth, et al., 2013). The ashy storm-petrel was discovered breeding on the island 10 years post-eradication, highlighting the contribution of the project towards stated goals (Whitworth, et al., 2013; Newton, et al., 2016). The operation was also the first aerial broadcast of rodenticide in the USA, and short-term non-target monitoring was undertaken to follow expected impacts (Howald, et al., 2010), and improve knowledge for further planning of this activity in the USA. Surveillance monitoring of other taxa also occurred, including endemic deer mice, herpetofauna and inter-tidal communities, to understand the broader impacts that occurred as a consequence of the eradication.

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– informing stakeholders of outcomes and engaging the public – have social benefits. Whether ecological or social, several of these benefits involve measuring or reporting against targets, so clearly defining the target outcomes of eradications is often a prerequisite for designing monitoring.

Monitoring may yield diminishing returns in terms of advancing our ecological knowledge, when the same outcome is monitored repeatedly. It can, therefore, detract from investment in future management action. This is an important consideration for repeated monitoring of the same island or site (Possingham, et al., 2012), but also for monitoring across projects where islands share similar habitat types, and invasive mammal-native species interactions. The target outcome of an eradication of a particular invasive species e.g. population recovery of a threatened species, may be confidently predicted if it is driven by a simple mechanistic relationship or there is sufficient evidence from previous eradications that benefited the same or ecologically similar species. The decision whether to monitor should, therefore, be informed by the current state of evidence: what prior knowledge exists and is it sufficient to confidently predict outcomes? For invasive mammal eradications, the evidence-base for predicting different outcomes is mixed. Individual outcomes have been reported for several projects but not consistently or comprehensively.

A key recommendation made during the Island Invasives 2017 monitoring workshop was to compile a synthesis of monitoring efforts to date, to identify taxonomic or geographic gaps in coverage that will help target future monitoring efforts. Although no comprehensive synthesis exists currently, some studies have collated and synthesised monitoring, either at a regional level (e.g. Russell, et al., 2016; Towns, et al., 2016), or globally for a taxonomic group. Schweizer, et al., (2016) reviewed available evidence of vegetation responses to goat (*Capra hircus*) and European rabbit (*Oryctolagus cuniculus*) eradications. Although there was evidence that vegetation responded following herbivore removal, variation in monitoring methods, timeframe and accounting for native versus non-native vegetation response hindered the drawing of conclusions. Thus, the authors recommended further monitoring to develop a general model of expected vegetation responses. Brooke, et al., (2017) collated seabird demographic responses following invasive mammalian predator eradication, highlighting that, in general, seabird populations increase following invasive mammal eradications. However, not all populations grew, insufficient data were available to distinguish between threatened and non-threatened species, and variation in response among major seabird taxa was evident. Thus, while generally seabirds can be predicted to respond positively following invasive mammal eradication, we lack sufficient knowledge to predict how and why this circumstance occurs, hence the recommendation for systematic long-term monitoring to improve understanding of the mechanisms of seabird population recovery (Brooke, et al., 2018).

The social benefits that accrue from monitoring–stakeholder feedback and public engagement–are more linear because, while ecological knowledge grows cumulatively from all projects that monitor, the social returns are primarily project specific. Foreseeably, the ecological need for monitoring may be low but, if the operation had high public or stakeholder interest, monitoring will be necessary.

Beyond a theoretical framework for monitoring, The Nature Conservancy is one organisation looking at their motivations for monitoring at an institutional level (Montambault & Groves, 2009). They found monitoring was a tool for managing risk and securing future investment–the greater a project's risk or higher the likelihood it could lead to follow-on funding, the higher the investment that should be made in monitoring (Case study 2). Eradication operations with considerable ecological uncertainty, or reputational risk, and those whose success could leverage additional public, political or financial support for future operations therefore all warrant a significant investment in monitoring (Table 1).

Having identified the motivations for monitoring, and decided on that basis whether monitoring is needed, it becomes easier for a project team to decide what to monitor and how. When the aim is to confirm that an eradication achieved target outcomes, monitoring focusses on those target beneficiaries. When the risk of unexpected outcomes is high, broader surveillance monitoring is appropriate. Both rely on assessing *the state* of target or non-target species or habitats, whereas understanding broader ecosystem responses is likely to require more detailed research into ecological mechanisms.

The goal and audience for reporting ecological outcomes of an eradication can influence the type of monitoring undertaken. When there is a need to report outcomes in a peer-reviewed publication to a technical or scientific audience, a different approach such as a quantified before-after comparison, may be required than for projects reporting to non-technical audiences such as donors or local communities (Case study 3), for which qualitative approaches like photo-monitoring vegetation changes may be sufficient. Further, the stakeholders using Traditional Ecological Knowledge will require a different approach

CASE STUDY 2: LEVERAGING CONSERVATION GAINS THROUGH GOAT ERADICATIONS IN THE GALÁPAGOS, ECUADOR

The ultimate goal of Project Isabela, initiated in 1997 and completed in 2006, was to facilitate the restoration of Pinta (5,940 ha) and Santiago (58,465 ha) Islands and the larger, northern portion of Isabela Island (approximately 250,000 ha; the whole island encompasses 458,812 ha). The project began in response to the massive destruction by introduced goats of both native vegetation and terrain (Galápagos Conservancy, 2017). Long-term vegetation monitoring was established on six of the 12 islands in the Galápagos where goats had been introduced (Tye, 2006). Permanent plots and transects showed that eradication or reduction of goat populations led to regeneration of native vegetation (Hamann, 1993; 1979), with a return to a near natural state in most cases after 20 years (Tye, 2006). The monitoring programme successfully fulfilled a number of roles. It confirmed, overall, the success of goat eradication in facilitating recovery of native vegetation and it provided lessons for subsequent eradication operations. In doing this, monitoring helped to manage the risk associated with the operation. The programme highlighted cases where individual species did not recover following goat eradication or exclusion so additional conservation management was required, including the tree fern (*Cyathea weatherbyana*) on Alcedo, whose last two remnant populations were protected by fences in 1997 (Tye, 2006). Perhaps most importantly monitoring demonstrated to public, state and donor audiences the benefits of invasive species management helping to leverage future investment. This led to the Charles Darwin Foundation (CDF) and the Galápagos National Park Service (GNPS) convening a workshop in 2007, on the completion of Project Isabela, to develop an action plan for managing rodents within the Galápagos (Galápagos Conservancy, 2017).

Table 1 Motivations and conditions for monitoring biodiversity outcomes of invasive species eradications.

Why?	When?	What?
Confirm target outcomes[1]	Outcomes are complex and difficult to predict, or poorly studied	Target beneficiaries – quantitative studies
Detect non-target outcomes[1]	Large or complex systems where outcomes are unpredictable	Non-target surveillance – quantitative studies
Learn about whole ecosystem responses[1]	Ecosystem responses remain poorly studied.	Ecological mechanisms of change – quantitative question-driven research. Community ecology
Inform stakeholders[1]	If required, especially for larger operations	Target beneficiaries – qualitative studies (see Case Study 1)
Engage the public[1]	Inhabited islands, regularly visited islands, large operations, publicly funded operations. Projects involving beneficiary species with a high public profile	Target beneficiaries – qualitative participatory monitoring
Ecological risk and uncertainty[2]	Threatened species involved and outcomes uncertain e.g. complex systems	Target beneficiaries – empirical studies
Reputational risk[2]	Large operations funded by key donors, or receiving political and public backing	Target beneficiaries – qualitative studies?
Leverage[2]	Exemplars and trial operations in new geographies paving the way for subsequent repetition/scaling-up	Target beneficiaries – empirical studies

Sources: [1]Possingham, et al., 2012 and [2]Montambault & Groves, 2009.

than non-Western science frameworks. Thus, identifying early the key audiences and their needs is recommended as this will influence the cost and approach of monitoring.

Finally, there are a number of practical considerations which may predispose projects to monitor, namely: when existing baseline data are particularly good; when there are existing established monitoring programmes e.g. run by rangers, universities or participatory groups; and/or when funding for monitoring does not compete with management.

Integrating monitoring into project planning

Fundamentally, monitoring should be considered in the earliest stages of project planning. This allows for additional baseline data to be collected if existing data are insufficient for a robust before-after comparison, and for monitoring to be costed and potentially included in the project budget.

There is a wide spectrum of possible monitoring investment for invasive mammal eradication projects, ranging from not monitoring at all, through to comprehensive whole ecosystem monitoring. The few whole ecosystem studies that exist (e.g. Towns, et al., 2016; Griffiths, et al., 2019) provide detailed learning into how systems respond to the eradication of particular species and provide a model for planning equivalent exercises elsewhere. Although these excellent studies represent the optimum approach for eradication monitoring, they are not achievable for all projects, nor may they be necessary to achieve project goals. Here, we aim to provide general guidelines for deciding what level of monitoring is required.

Fig. 1 presents a decision tree outlining the key considerations which determine whether monitoring is necessary, what needs to be monitored and the type of monitoring needed. Although it is presented as a workflow, several steps are inter-related and feed into one another.

1. Defining the desired outcomes of eradication

The most common motivation for monitoring is to confirm the expected outcomes for native taxa after removing a pest species from an island. It is therefore essential that projects clearly define their objectives (Prior, et al., 2018): why is eradication proposed?; what is it expected to achieve? Outcomes should be explicitly split into proximal outcomes, which will typically include the removal of an invasive species and the undesirable interactions with native species (e.g. predation), and ultimate outcomes such as the recovery of a native species. These ultimate outcomes are sometimes referred to as impacts (Nam, et al., 2013). Conceptually, post-eradication outcomes like improved survival and recruitment can lead to impacts like population growth. Where possible, outcomes should be specific, measurable, agreed-upon by those involved in the project, realistic (i.e. ecologically viable), and time-bound (Doran, 1981).

CASE STUDY 3: MONITORING ON ST AGNES AND GUGH, ISLES OF SCILLY, UK

Brown rats (*Rattus norvegicus*) were successfully eradicated from the islands of St Agnes and Gugh in 2013 (Thomas, et al., 2017). The islands have a combined area of 142 ha and a population of 82 people, making it the largest community-led rat eradication project in the world to date. Engaging the community in all aspects of the project including monitoring – and keeping them engaged throughout the life span of the project – was key to the project's success. Community members, especially schoolchildren, were involved in the work, with many people volunteering to take part in monitoring of native shrews, invertebrates, plants and birdlife. The islands' seabirds are of particular value to the community, and islanders are involved in ongoing 'chick check' walks which monitor the breeding success of Manx shearwaters (*Puffinus puffinus*) and European storm-petrels (*Hydrobates pelagicus*) two species which have bred on the islands for the first time in living memory following the eradication of rats. The monitoring activities associated with the eradication project have therefore fulfilled several roles – they have provided ongoing scientific data on the wider ecological impacts of rat eradication and have provided powerful publicity and advocacy information regarding the immediate benefits of eradication on species preyed upon by rodents, such as shearwaters and storm-petrels. The monitoring has also galvanised and helped maintain ongoing community support for the project and ownership of its long-term outcomes.

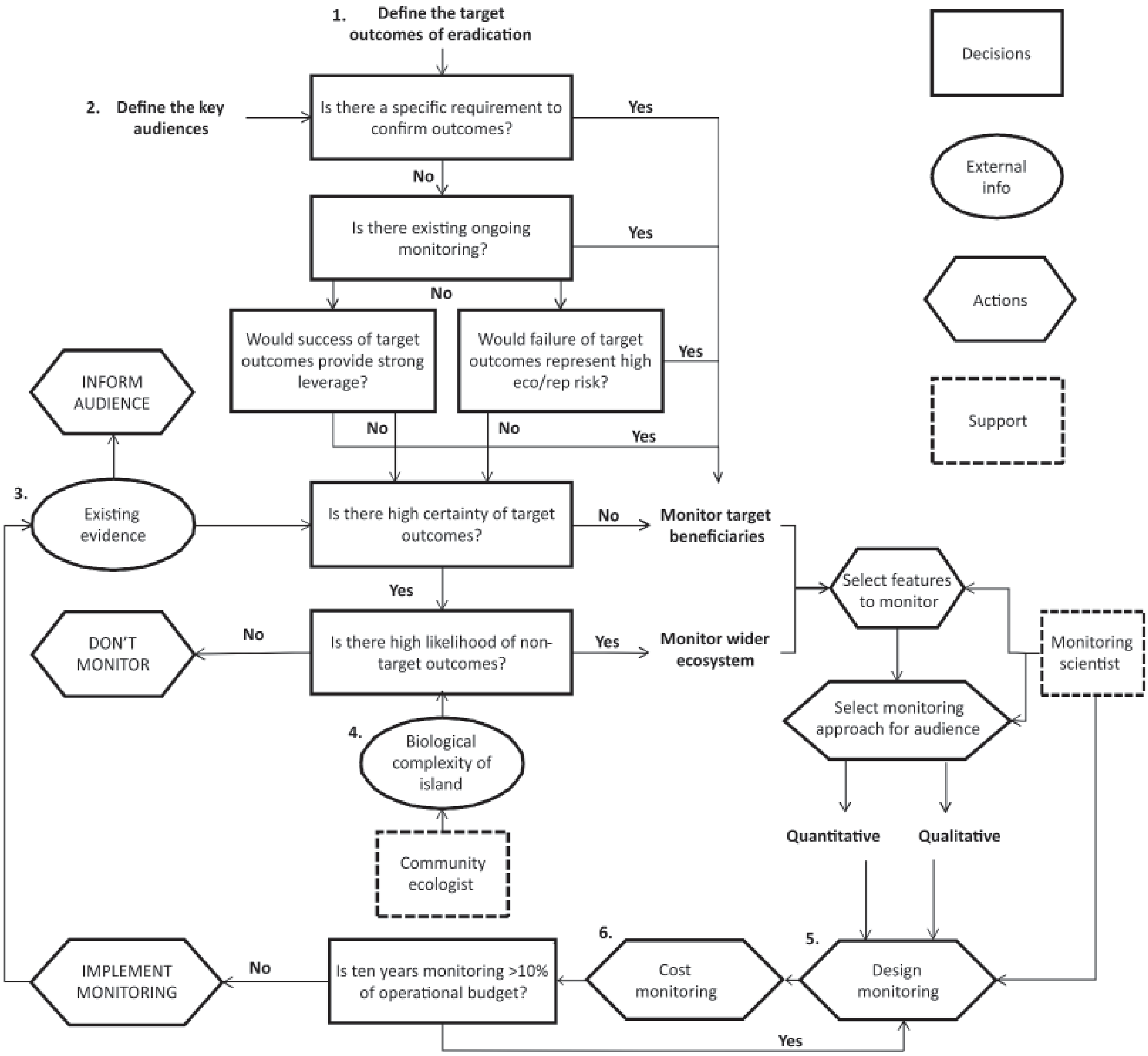

Fig. 1 A decision tree to assist with planning biodiversity monitoring in relation to eradication programmes. After the target outcomes of the eradication and the key audiences are defined, gathering existing evidence and answering a number of questions will inform the scope that monitoring needs to encompass, as well as guide selection of monitoring targets and the required approach to monitoring. Numbered points are discussed in the text.

2. Defining the key audiences

Just as *what* is monitored is driven by project goals (e.g. seabird protection), to *whom* monitoring results should be communicated should also be defined as a part of project planning. For each project, target audiences should be identified: a relevant group for whom the results of ecological monitoring will be of interest, some of whom may actively request the information, while others may be informed more for advocacy and education purposes. By defining these audiences, managers can prioritise and determine what messages and products (e.g. peer-reviewed publications, images, webpages, reports or public lectures) monitoring needs to inform and, in a feedback loop, can identify what to monitor. Key audiences may include the local community, especially island residents or communities close to the island; permit providers such as statutory bodies and island managers wanting to understand the wider ramifications of eradication; conservation scientists and technical communities wishing to use monitoring data to highlight ecological benefits of projects to advocate for similar work; donors vetting project outcomes and return on investment; and decision makers at local and regional levels. At a higher level, the data may also be used to lobby policy makers to enact or amend legislation relating to invasive species and their management. Finally, project managers may wish to engage the wider public with the results of their monitoring work, seeking to develop more broadly society's understanding of the issues posed by invasive species on islands.

3. Identifying existing resources

Determining the presence and suitability of existing baseline data for the target island is an important activity. Existing baseline information may satisfy pre-eradication information needs and can inform future monitoring to replicate the baseline methodology. This exercise may identify stakeholders already engaged in monitoring on the target island, or nearby control islands, whose ongoing work may be tailored to inform eradication outcomes.

It is also valuable to assess the outcomes of other eradication projects that benefited similar species or ecosystems, for example ground-nesting seabirds like terns (*Sterna* spp.) perform well after the removal of all invasive mammalian predators (Brooke, et al., 2017). The consistency with which previous eradications delivered particular outcomes will establish the level of confidence in achieving desired management goals. Syntheses have been undertaken for some taxa that outline broad-scale responses (e.g. Jones, et al., 2016; Schweizer, et al., 2016; Brooke, et al., 2017). Williams, et al., (2017) synthesised information from 16 before-and-after studies documenting seabird responses to predator removal and provide practitioners with effectiveness and certainty ratings for invasive mammal control as a conservation intervention. If the results among projects vary considerably, or there is a specific requirement for reporting to an audience on localised information, then monitoring is warranted.

4. Adopting a whole-ecosystem approach

When planning and designing monitoring, ecosystem processes and community structure should be considered (Zavaleta, et al., 2001; Prior, et al., 2018). By modelling the trophic interactions in a system, flow-on effects can be anticipated, reducing the likelihood of unexpected outcomes (e.g. Baker, et al., 2017). There is a gradient of approaches available to achieve this process, ranging from simple food web diagrams through to models with input from community/ecosystem ecologists. Generating sophisticated models is challenging for many sites owing to a lack of baseline information. However, even simple exercises capturing current and projected interactions within an ecosystem could aid planning. By considering the trophic interactions on an island, those component taxa of interest which are most likely to be impacted can be identified and elevated to monitoring targets. This will also clarify the complexity of the system which highlights the potential need for wider surveillance monitoring beyond anticipated outcomes.

5. Designing monitoring

There is a whole suite of taxa- and site-specific monitoring methods that projects can utilise—it is not our aim to discuss them here. Rather, we focus on three key elements of monitoring design: i) choosing between quantitative and qualitative monitoring methods; ii) determining what to monitor; and iii) allowing for pre- and post-eradication comparison.

The need for quantitative or qualitative monitoring is influenced by the audience to whom monitoring results will be communicated. As described above, a spectrum exists. At one end, are projects for which quantitative monitoring is required: for example, those with quantitative targets such as percentage population changes or reductions in negative trends; or those aiming to quantify outcomes to inform other eradication operations (e.g. by providing evidence for syntheses like Williams, et al., 2017). Further along the spectrum are projects that may need only to provide qualitative evidence of outcomes to laypersons' audiences: perhaps photo-plots illustrating the growth in vegetation following an eradication; or "traffic-light" assessments of ecological integrity (e.g. Tierney, et al., 2009) of an island system following eradication.

To serve most purposes, monitoring can likely focus on taxa or habitats identified when the target outcomes of the eradication were defined. But, when potential secondary outcomes have been identified, such as increases in invasive invertebrates or prey-switching by meso-predators, taxa or habitats predicted to be affected can also be selected as monitoring targets. When outcomes are highly uncertain, we recommend wider surveillance monitoring is undertaken to detect hard-to-predict secondary outcomes. In that case, taxa can be selected for monitoring based first upon need (they are predicted to be affected, but with unknown consequences), and then opportunity, e.g. continued monitoring is worthwhile because baseline data exist and monitoring can be continued easily; there are people involved in the project with particular expertise; there are taxa present for which monitoring is likely to be particularly cost-effective.

Sampling design should ideally occur before eradication. In some instances a Before-After-Control-Impact (BACI) approach may be possible (Quinn & Keough, 2002), whereby control islands (either those with invasive species but where no eradication is carried out, or those with no comparable invasive species at all) can be compared to experimental islands (those with the eradication e.g. Samaniego-Herrera, et al., 2017).

6. Monitoring cost

A major determinant of monitoring design is the economic cost, relative to available budget. For monitoring planned shortly after an eradication, an opportunity for cost saving is to combine efforts with activities to confirm the success or failure of the operation itself.

The amount invested should increase relative to risk and leverage potential (Montambault & Groves, 2009), but there are no clear guidelines on what proportion of a budget to allocate for monitoring and evaluation. There has been no review of proportional expenditures by conservation projects on monitoring and evaluation, but within the development sector and across major foundations typical expenditure is 3–5% of programme costs (Austrian Development Agency, 2009; Twersky & Arbreton, 2014), rising to an upper ceiling of 10% (Zondag, 2009). Establishing a fixed limit for monitoring budgets helps to guide monitoring design, and may result in iterative design to keep monitoring within budget. Including monitoring costs in the overall eradication budget is perhaps the most straightforward way of funding monitoring, when it is a relatively small component of the overall fund-raising target. However, funds secured in this way are often time-bound and not goal dependent—they often expire before monitoring has been conducted for enough years to demonstrate that a target outcome has been reached. Addressing this issue by exploring financial mechanisms such as endowment funds to separate and safeguard monitoring budgets and ongoing biosecurity, or integrate ecosystem monitoring with biosecurity monitoring, could help future projects and improve upon the current approach that relies on post-eradication fund-raising specifically for monitoring.

Making the most of monitoring results

With so many eradication projects now being carried out worldwide and many of them generating data through associated monitoring, it is increasingly difficult for scientists, managers and field officers to keep up to date with new findings, and they can be hampered by language barriers. Furthermore, the data generated are not always disseminated widely. Understandably, positive changes to target beneficiaries and to flagship species, are the most widely reported. Changes in the abundance of other taxa, especially plants and invertebrates, are less often reported, or likely monitored (Jones, et al., 2016; Towns, 2018).

Understanding the outcome of previous eradication projects' pre- and post-eradication monitoring may help new projects gain support for their work, may help to identify and thus allow minimisation of negative secondary impacts, and may help to optimise the allocation of resources to conservation actions where monitoring can be reduced. It is very important, therefore, that first the results from any monitoring that has occurred are disseminated, and second that the information is curated in a readily accessible and searchable manner accessible to technicians, land managers, scientists, conservation bodies, educators and other interested parties. Ideally, they would all be available via a single repository but nothing exists currently (although there is a searchable database of island eradications; Holmes, et al., 2019). Although the outcomes of individual eradications may not be considered sufficiently novel for higher profile journals (Brooke, et al., 2018), a number of journals specifically promote the dissemination of evidence by promoting publication of the outcomes of conservation interventions (Sutherland, et al., 2017). There are opportunities for open access publishing with no limit on the number of papers publishable or the geographies covered, and a streamlined submission and review process. There is a range of ways in which results can then be disseminated more widely (Table 2).

Monitoring informs future conservation practice; it enables us to increase our likelihood of success and reduces uncertainty. We believe that there is a need to broaden information availability and shared resources through diverse platforms, in order to facilitate knowledge exchange. To date, the findings of post-eradication monitoring have not been consistently disseminated, so a behavioural change must be supported and requires incentivising. Including these costs in eradication budgets and encouraging donors to support the collection of evidence that confirms return on investment are first steps in tackling the problem.

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Table 2 Summary of dissemination routes for pre- and post-eradication monitoring data, particularly to the invasive species-specific technical community.

Media	Dissemination route	Target audience
Raw data	Inter-agency scientific collaborations	Invasive species practitioners and scientists
	Small organisations collaborating with larger ones who can support with data analysis and interpretation	Any
Analysed results	Journals reporting outcomes of conservation intervention in a searchable database	Invasive species-specific scientific and technical community
	Aliens-l listserver	Invasive species-specific scientific and technical community
	Held in central database (e.g. Database of Island Invasive Species Eradications—subject to copyright issues)	Invasive species-specific scientific and technical community
	Eradication project/ organisation websites	Invasive species-specific technical community
Technical reports	Briefing documents, e.g. POST (Parliamentary Office of Science and Technology) notes	Local, regional and national government
	Aliens-l listserver	Invasive species-specific technical community
	Regional websites, e.g. Pacific Invasives Learning Network, PestSmart Connect	
	Annual compendium	Invasive species-specific technical community
	Island Invasives Conference proceedings	Scientific and academic community, invasive species-specific scientific and technical community
	Community forums (newsletters, magazines, websites)	Community in which eradication project was conducted, communities in which similar projects are planned
Layperson reports	Through Aliens-l listserver	Invasive species-specific technical community
	Held in central database?	Invasive species-specific technical community
	Schools	Primary and secondary school children, and teachers
Educational materials	Universities (use examples in lectures on island restoration and species recovery)	Students
	Talks/presentations	Community in which eradication project was conducted, communities in which similar projects are planned, special interest groups (e.g. local bird or mammal groups)
	Web sites	
Social media	Projects own Facebook pages, and links to reports via twitter and instagram	Scientific and academic community, invasive species-specific scientific and technical community. Community in which eradication project was conducted, communities in which similar projects are planned, special interest groups (e.g. local bird or mammal groups)

REFERENCES

ATTC (2001). *Restoration Plan & Environmental Assessment for Seabirds Injured by the American Trader Oil Spill*. [Online.] Report of the American Trader Natural Resource Trustee Council, U.S. Fish and Wildlife Service, California Department of Fish and Game, and National Oceanic and Atmospheric Administration. <https://casedocuments.darrp.noaa.gov/southwest/amtrader/pdf/amtradefinal_restoreplanon.pdf>. Accessed 21 October 2017.

Austrian Development Agency (2009). *Guidelines for Project and Programme Evaluations*. Vienna, Austria: Austrian Development Agency.

Baker, C.M., Gordon, A. and Bode, M. (2017). 'Ensemble ecosystem modeling for predicting ecosystem response to predator reintroduction'. *Conservation Biology* 31: 376–384.

Brooke, M. de L., Bonnaud, E. Dilley, B.J., Flint, E.N., Holmes, N.D., Jones, H.P., Provost, P., Rocamora, G., Ryan, P.G. Surman, C. and others (2017). 'Seabird population changes following mammal eradications on islands'. *Animal Conservation* 21: 3–12.

Brooke, M. de L., Bonnaud, E. Dilley, B.J., Flint, E.N., Holmes, N.D., Jones, H.P., Provost, P., Rocamora, G., Ryan, P.G. and Surman, C. (2018). 'Enhancing the value of future island eradications needs improved understanding of past outcomes'. *Animal Conservation* 21: 19–20.

Dawson, J., Oppel, S. Cuthbert, R.J., Holmes, N., Bird, J.P., Butchart, S.H., Spatz, D.R. and Tershy, B. (2015). 'Prioritizing islands for the eradication of invasive vertebrates in the United Kingdom overseas territories'. *Conservation Biology* 29: 143–153.

Donlan, C.J. and Wilcox, C. (2007). 'Complexities of costing eradications'. *Animal Conservation* 10: 154–156.

Donlan, C.J., Luque, G.M. and Wilcox, C. (2015). 'Maximizing return on investment for island restoration and species conservation'. *Conservation Letters* 8: 171–179.

Doran, G.T. (1981). 'There's a SMART way to write management's goals and objectives'. *Management Review* 70: 35–36.

Galapagos Conservancy (2017). 'Project Isabela'. <https://www.galapagos.org/conservation/conservation/project-areas/ecosystem-restoration/project-isabela/>. Accessed 21 October 2017.

Griffiths, R., Bell, E., Campbell, J., Cassey, P., Ewen, J.G., Green, C., Joyce, L., Rayner, M. Toy, R. Towns, D. Wade, L., Walle, R. and Veitch, C.R. (2019). 'Costs and benefits for biodiversity following rat and cat eradication on Te Hauturu-o-Toi/Little Barrier Island'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 558–567. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Hamann, O. (1979). 'Regeneration of vegetation on Santa Fé and Pinta Islands, Galápagos, after the eradication of goats'. *Biological Conservation* 15: 215–235.

Hamann, O. (1993). 'On vegetation recovery, goats and giant tortoises on Pinta Island, Galápagos, Ecuador'. *Biodiversity and Conservation* 2: 138–151.

Helmstedt, K.J., Shaw, J.D., Bode, M., Terauds, A., Springer, K., Robinson, S.A. and Possingham, H.P. (2016). 'Prioritizing eradication actions on islands: It's not all or nothing'. *Journal of Applied Ecology* 53: 733–741.

Holmes, N.D., Keitt, B.S., Spatz, D.R., Will, D.J., Hein, S., Russell, J.C., Genovesi, P., Cowan, P.E. and Tershy, B.R. (2019). 'Tracking invasive species eradications on islands at a global scale'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 628–632. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Howald, G., Donlan, C.J., Faulkner, K.R., Ortega, S., Gellerman, H., Croll, D.A. and Tershy, B.R. (2010). 'Eradication of black rats *Rattus rattus* from Anacapa Island'. *Oryx* 44: 30–40.

Jones, H.P., Williamhenry, R., Howald, G.R., Tershy, B.R., and Croll, D.A. (2005). 'Predation of artificial Xantus's murrelet *Synthliboramphus hypoleucus scrippsi* nests before and after black rat *Rattus rattus* eradication'. *Environmental Conservation* 32: 320–325.

Jones, H.P., Holmes, N.D., Butchart, S.H.M., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P.E., Cuthbert, R.J., Ebbert, S., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R., and Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences* 113 :4033–4038.

Kapos, V., Balmford, A., Aveling, R., Bubb, P., Carey, P., Entwistle, A., Hopkins, J., Mulliken, T., Safford, R., Stattersfield, A., Walpole, M. and Manica, A. (2008). 'Calibrating conservation: New tools for measuring success'. *Conservation Letters* 1: 155–164.

Lindenmayer, D.B. and Likens, G.E. (2010). 'The science and application of ecological monitoring'. *Biological conservation* 143: 1317–1328.

McCarthy, D.P., Donald, P.F., Scharlemann, J.P., Buchanan, G.M., Balmford, A., Green, J.M., Bennun, L.A., Burgess, N.D., Fishpool, L.D., Garnett, S.T. and others (2012). 'Financial costs of meeting global biodiversity conservation targets: Current spending and unmet needs'. *Science* 338: 946–949.

Montambault, J.R. and Groves, C.R. (2009). *Improving Conservation Practice by Investing in Monitoring Strategy Effectiveness*. Conservation Measures Working Paper 2. Arlington, Virginia: The Nature Conservancy.

Nam, S., Ampofo, F.O., Shields, L., Alatamirano, E.S., and Toure, M.S. (2013). *Enhancing Learning in Monitoring and Evaluation: Six Cases from Philanthropic Organizations Working in the Health Sector*. Seattle, Washington: Bill and Melinda Gates Foundation.

Newton, K.M., McKown, M., Wolf, C., Gellerman, H., Coonan, T., Richards, D., Harvey, A.L., Holmes, N., Howald, G., and Faulkner, K. (2016). 'Response of native species 10 years after rat eradication on Anacapa Island, California'. *Journal of Fish and Wildlife Management* 7: 72–85.

NPS (2000). *Anacapa Island Restoration Project: Final Environmental Impact Statement.* Ventura County, California: National Park Service. <http://www.nps.gov/chis/learn/nature/upload/ AIRP%20Final%20EIS%20Full.pdf>. Accessed 21 October 2017.

Possingham, H.P., Wintle, B.A., Fuller, R.A., and Joseph, L.N. (2012). 'The Conservation Return on Investment from Ecological Monitoring'. In: D. Lindenmayer and P. Gibbons (eds.) *Biodiversity Monitoring in Australia*, pp. 49–58. Clayton, Australia: CSIRO Publishing.

Prior, K.M., Adams, D.C., Klepzig, K.D., and Hulcr, J. (2018). 'When does invasive species removal lead to ecological recovery? Implications for management success'. *Biological Invasions* 20: 267–283.

Quinn, G.P. and Keough, M.J. (2002). *Experimental Design and Data Analysis for Biologists*. Cambridge: Cambridge University Press.

Russell, J.C., Cole, N.C., Zuël, N., and Rocamora, G. (2016). 'Introduced mammals on Western Indian Ocean islands'. *Global Ecology and Conservation* 6: 132–144

Samaniego-Herrera, A., Clout, M.N., Aguirre-Muñoz, A., and Russell, J.C. (2017). 'Rodent eradications as ecosystem experiments: A case study from the Mexican tropics'. *Biological Invasions* 19: 1761–1779.

Schweizer, D., Jones, H.P., and Holmes, N.D. (2016). 'Literature review and meta-analysis of vegetation responses to goat and European rabbit eradications on islands'. *Pacific Science* 70: 55–71.

Springer, K. (2016). 'Methodology and challenges of a complex multi-species eradication in the sub-Antarctic and immediate effects of invasive species removal'. *New Zealand Journal of Ecology* 40: 273–278.

Sutherland, W.J., Pullin, A.S., Dolman, P.M., and Knight, T.M. (2004). 'The need for evidence-based conservation'. *Trends in Ecology and Evolution* 19: 305–308.

Sutherland, W.J., Dicks, L.V., Ockendon, N., and Smith, R.K. (2017). *What Works in Conservation*. Cambridge, UK: Open Book Publishers.

Thomas, S., Brown, A., Bullock, D., Lock, L., Luxmoore, R., Roy, S., Stanbury, A. and Varnham, K. (2017) 'Island Restoration in the UK – past, present and future'. *British Wildlife* 28(4): 231–242.

Tierney, G.L., Faber-Langendoen, D., Mitchell, B.R., Shriver, W.G., and Gibbs, J.P. (2009). 'Monitoring and evaluating the ecological integrity of forest ecosystems'. *Frontiers in Ecology and Environment* 7: 308–316.

Towns, D., Borrelle, S., Thoresen, J., Buxton, R., and Evans, A. (2016). 'Mercury Islands and their role in understanding seabird island restoration'. *New Zealand Journal of Ecology* 40: 235–249.

Towns, D.R. (2018). 'Understanding seabird responses to invasive mammal eradications from islands needs systematic monitoring'. *Animal Conservation* 21: 15–16.

Twersky, F. and Arbreton, A. (2014). *Benchmarks for Spending on Evaluation*. Menlo Park, California: William and Flora Hewlett Foundation.

Tye, A. (2006). 'Restoration of the vegetation of the Dry Zone in Galapagos'. *Lyonia* 9: 29–50.

Underwood, E.C., Shaw, M.R., Wilson, K.A., Kareiva, P., Klausmeyer, K.R., McBride, M.F., Bode, M., Morrison, S.A., Hoekstra, J.M., and Possingham, H.P. (2008). 'Protecting biodiversity when money matters: Maximizing return on investment. *PLOS One* 3: e1515.

Veitch, C.R., Clout, M.N., and Towns, D.R. (eds.) (2011). *Island invasives: eradication and management.* Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Whitworth, D.L., Carter, H.R., and Gress, F. (2013). 'Recovery of a threatened seabird after eradication of an introduced predator: Eight years of progress for Scripps's murrelet at Anacapa Island, California'. *Biological Conservation* 162: 52–59.

Williams, D.R., Child, M.F., Dicks, L.V., Ockendon, N., Pople, R.G., Showler, D., Walsh, J.C., zu Ermgassen, E.K.H.J., and Sutherland, W.J. (2017). 'Bird Conservation'. In: W.J. Sutherland, L.V. Dicks, N. Ockendon, and R.K. Smith (eds.) *What Works in Conservation*, pp. 95–244. Cambridge, UK: Open Book Publishers.

Wilson, K.A., McBride, M.F., Bode, M., and Possingham, H.P. (2006). 'Prioritizing global conservation efforts'. *Nature* 440: 337–340.

Zavaleta, E.S., Hobbs, R.J., and Mooney, H.A. (2001). 'Viewing invasive species removal in a whole-ecosystem context'. *Trends in Ecology and Evolution* 16: 454–459.

Zondag, M.-J. (2009). 'Practical Handbook for Ongoing Evaluation'. INTERACT Program Secretariat. <http://wiki.interact-eu.net/pages/viewpage.action?pageId=23756932>. Accessed 21 October 2017.

Community-based conservation and recovery of native species on Monuriki Island, Fiji

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Abstract The small uninhabited island of Monuriki (40.4 ha) in western Fiji is of national and international conservation concern for its several protected species. Exotic invasive species and a Category 5 cyclone have exacerbated conservation challenges. The cooperation of local, national, and international stakeholders continues to be crucial in restoration of the island's native flora and fauna. This summary presents a timeline of restoration efforts and current status of the recovery programme for Monuriki. The critically endangered dry forest habitat of Fiji is only found in a few isolated patches on disparate islands. The Fijian crested iguana (*Brachylophus* cf. *vitiensis*) is a critically endangered species restricted to a few small islands in this dry forest zone of western Fiji. The population of crested iguanas on Monuriki Island is the third largest remaining population. Even before iguanas were first documented on the island in the early 1980s, individuals had been removed by local resorts for display purposes, a practice that was previously unregulated. In the late 1990s, the first efforts to conserve and restore Monuriki Island were discussed, but conservation initiatives were not implemented until the development of the Crested Iguana Recovery Plan in 2008. In 2011, domestic goats (*Capra hircus*) and non-native rats (*Rattus exulans*) were removed from the island, and 10 pairs of iguanas were brought into captive breeding facilities within Fiji. In 2015, the first 32 captive-bred crested iguanas were released back on Monuriki Island. More than half of these iguanas (N=26) were radio-tracked for 56 days post-release in order to assess survivorship and help provide insights into their short-term movement patterns. Of the 26 iguanas that were tracked, nearly 70% (N=18) were found after 56 days indicating excellent short-term survival. In February 2016, Tropical Cyclone Winston, a Category 5 storm passed through Fiji and devastated some of the tropical dry forest habitat on Monuriki. With sustained winds of up to 230 km/hr nearly all of the canopy leaves from trees on Monuriki Island were removed and large amounts of debris covered the forest floor. Following the cyclone, a brief wildlife survey revealed Monuriki's iguana and bird populations were still present. In 2017, the crested iguana captive breeding programme was brought to an end when 16 of the original 20 iguana founders, and an additional 32 captive bred offspring, were reintroduced onto Monuriki. This was accomplished, in part, due to successful breeding and reestablishment of the remaining wild iguanas on the island. Despite a major storm event, reestablishment likely resulted from reduced egg and hatchling predation by the rats, and excellent habitat recovery after goat removal. Overall these invasive species eradications have proven highly successful for the recovery of the iguanas, wedge-tailed shearwaters (*Puffinus pacificus*), and several other non-target species including the banded rail (*Gallirallus philippensis*) and endangered Fijian peregrine falcon (*Falco peregrinus*). Furthermore, eradication of non-native species has also helped the recovery of the highly threatened tropical dry forest ecosystem in which these species exist.

Keywords: *Brachylophus* cf. *vitiensis*, eradication, Fijian crested iguanas, goats, island restoration, Pacific rats, *Puffinus pacificus*, wedge-tailed shearwater

INTRODUCTION

Tropical dry forest habitats are globally rare and often contain highly endemic faunas. These forests are typically impacted by anthropogenic fires to convert them into lands for agriculture on mainland regions and degraded by multiple invasive alien species such as grazing and predatory mammals, and various invertebrates on islands. In Fiji, most of the dry forest on the two large islands of Viti Levu and Vanua Levu has been transitioned into sugar cane, cattle grazing, or invasive grasslands (Olson, et al., 2010). Dry forest persists only on some of the smaller islands, or in very limited patches on larger islands. Of the smaller islands Monuriki and Monu have been identified as Key Biodiversity Areas (KBA's) by Conservation International because of their significance as critical refugia for the Fijian crested iguana (*Brachylophus* cf. *vitiensis*) and tropical dry forest (Conservation International, 2005; Olson, et al., 2010). These islands, and particularly Monuriki, also support the largest wedge-tailed shearwater (*Puffinus pacificus*) population in Fiji. This paper outlines progress made with the restoration of Monuriki Island by working with the traditional land owners through an innovative and inclusive conservation partnership.

Location

The uninhabited Monuriki Island (12.610° S, 177.034° E) lies within the Mamanuca group in the province of Nadroga, western Fiji (Fig. 1). This 40.4 ha volcanic island reaches its peak at 177 m above sea level (Fig. 2). Monuriki is owned by the Mataqali Vunaivi, the traditional Fijian clan living on the nearby island of Yanuya. Monuriki is listed under the National Biodiversity Strategic Action Plan as a site of national significance due to its tropical dry forest and two particular species of international or national conservation concern, the Fijian crested iguana (*Brachylophus* cf. *vitiensis*) and the wedge-tailed shearwater (*Puffinus pacificus*) (Coulston, et al., 2010; Olson, et al., 2010). Monuriki is the location of the third largest population of the endemic Fijian crested iguana (IUCN, 2014). This iguana is listed on CITES Appendix I, as Critically Endangered by the IUCN Red List (IUCN, 2014), and Endangered by the US Fish and Wildlife Service; it is protected in Fiji under the Endangered and Protected Species Act (2002). Monuriki Island crested iguanas are genetically distinct from all other crested iguana populations (Keogh, et al., 2008), and the 2008 Iguana Species Recovery Plan (Harlow, et al., 2008) prioritised Monuriki as the single most important site

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for immediate conservation action for this taxon. These iguanas were discovered only in 1980. At that time there was "a high density of iguanas" on Monuriki (Gibbons, 1984); however, less than 20 years later a survey revealed fewer than 100 iguanas remained (Harlow & Biciloa, 2001). A more recent survey indicated the population had dropped precipitously further with more extensive surveys reporting only eight individuals found in 2003 (Harlow, et al., 2007). The island also hosts several nesting colonies of the wedge-tailed shearwater (*Puffinus pacificus*), a species known from seven islands in Fiji and among which Monuriki supports the most significant population. These sea birds excavate burrows, often in the fragile coastal strand substrates, to rear their chicks. It was estimated that more than a thousand pairs of wedge-tailed shearwaters annually breed on this island (Rasalato, et al., 2012). Exotic faunal and floral species have invaded many of the islands in Fiji and pose a serious environmental threat to Monuriki's native biodiversity.

Threats to native species

Fire is an anthropogenic threat to Monuriki, due to the island's small size and lack of natural ignition sources. Exacerbating this is the threat of exotic non-native species. The history of exotics on the island of Monuriki may date back more than 3000 YBP. Pacific rats (*Rattus exulans*) were the first exotic species introduced to this island, most likely as stowaways with the early human arrivals (Roberts, 1991). This adaptable species has been able to sustain itself on most islands left unchecked until eradication is implemented. Rats are known to prey on eggs and chicks of nesting birds, as well as lizards, juvenile tuatara (*Sphenodon punctatus*), and seeds (Towns, et al., 2006). Domesticated goats (*Capra hircus*) were established on Monuriki during the 1970s. Originally brought as livestock, they provided an income for the Yanuya owners of the island. As voracious grazers, goats denuded the island of its undergrowth and ate the seedlings of forest trees, and leaf litter causing serious habitat degradation and severe erosion. The dry forest habitat may recover from infrequent dry season burning in the next rainy season provided seedlings are left intact. However, when goats are present the seedlings are grazed, preventing the regeneration of these native plants and trees. This causes the endangered dry forest habitat to convert to a mostly non-native composition while any remaining mature native trees senesce and eventually die. Following this cycle with fire and goat grazing on Monuriki, the lack of native food plants posed a threat to the diminished population of iguanas. Most of the surviving vegetation was unpalatable to both goats and iguanas. The open ground left by the continual grazing created space for opportunistic invasive exotic and unpalatable plants to take hold, including the native but invasive vaō (*Neisoperma oppositifolium*). Normally found in low abundance, vaō became overabundant with disturbance and an increase in light through the canopy increasing its representation in the forest composition. Goats also threatened the ground-nesting shearwaters by trampling nests, and causing the collapse of fragile burrows containing eggs, chicks, and nesting adults. The loss of insulating vegetation (leaf litter and understorey structure), moderating water runoff and erosion during heavy rain events, also potentially reduced shearwater breeding success due to inundation of burrows and nests. Most documented extinctions and current causes of declining numbers of Pacific island birds result from the effects of invasive alien species, and particularly vertebrates such as rats and goats (McCreless, et al., 2016).

Tourism and poaching are additional disturbances to this island which greatly impact iguanas, and increased foot traffic during the breeding season might impact wedge-tailed shearwaters, and other natives such as the banded rail (*Gallirallus phillippensis*), and peregrine falcon (*Falco peregrinus*). Monuriki is the site where the award-winning 1999 Dreamworks movie "Castaway" was filmed, and subsequently the island has become popular as a tourist stop with the remaining movie set maintained as a primary attraction. Ecotourism is the major contributor to Fiji's economy in this region. An estimated 70–100 tourists visit the island daily although the number could be much larger some days. Local resorts are still removing iguanas for display on other nearby islands as an "ecotourism" prop to draw in customers, and for tourist activities such as staged photos.

Restoration plans and community conservation

In the late 1990s, the first efforts to conserve and restore Monuriki were discussed. In 1998, 2000, and 2003, surveys of Monuriki detected a rapid decline in the iguana population as a result of continued major habitat degradation by goats, with only adult iguanas being detected and no evidence of recent recruitment (Harlow, et al., 2007). The landowners were approached on at least three occasions to remove goats from Monuriki, but declined to participate each time. In 2004 the IUCN Iguana Specialist Group (ISG) met in Fiji to discuss the current impacts on Monuriki, as part of identifying potential conservation actions possible for the species, including captive breeding. Through the development of the IUCN Crested Iguana Recovery Plan in 2008, conservation action steps for Monuriki were fully-developed and finally implemented. In 2009, BirdLife International undertook surveys that documented rats and goats to be major threats to nesting seabirds on Monuriki and nearby Monu and Kodomo islands (NTF, 2012). It was concluded that, if left unchecked, the persistence of rats and goats would lead to the loss of the dry forest and nesting seabirds on Monuriki. With the endorsement of the Yanuya

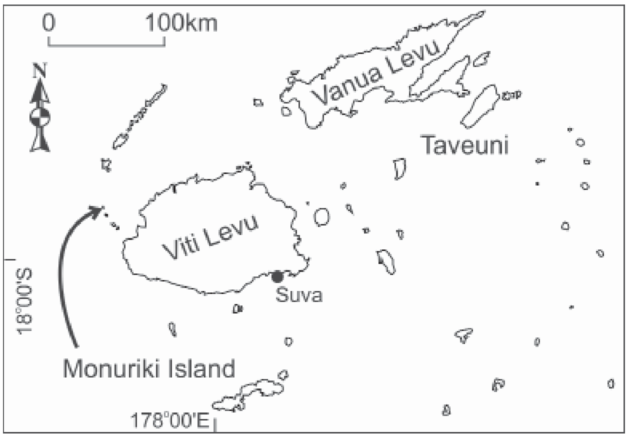

Fig. 1 Map of Fiji with arrow showing the location of Monuriki Island.

Fig. 2 View of Monuriki Island from 2017, looking to the south-west.

Village Chief and chief landowner, Sitiveni Drigi, an eradication programme was therefore implemented during 2010–2011, specifically targeting invasive rats and feral goats remaining on Monuriki (NTF, 2012). The goal of this programme was to restore an ecosystem suitable for the wedge-tailed shearwater, birds native to Fiji's tropical dry forest, and the critically endangered crested iguana. Regular post-eradication monitoring, along with environmental education campaigns specifically targeting biosecurity among island users including tourists and fishermen, were critical components to sustaining the eradication results and restoration outcomes (Donlan & Keitt, 1999). The eradications were designed in collaboration with eradication experts (i.e., Pacific Invasives Initiative), local land owners, the Provincial and National government, and consultants. Two years of monitoring on Monuriki (and Kodomo) did not detect rats and goats and by the end of 2013 their eradication was confirmed.

METHODS

Consultation

Many stakeholders were included in discussions regarding the invasive species eradication plan. The plan was launched as a community-based effort to successfully restore and protect the natural state of Monuriki. The involvement of the landowning unit in the village of Yanuya (the Mataqali Vunaivi) and their chief, the late Taukei Yanuya (Ratu Sitiveni Drigi) played a pivotal role in ensuring the restoration and protection of Monuriki was supported by the Yanuya community.

Other stakeholders since 2009 include (but are not limited to):

- BirdLife International
- Commissioner Western's office of the Fiji Government
- Community members and village groups
- Critical Ecosystem Partnership Fund
- David & Lucile Packard Foundation
- Department of Heritage and Arts
- Disney Conservation Fund
- School of Geography at the University of the South Pacific
- Iguana Specialist Group (IUCN)
- International Iguana Foundation
- Kula Wild Adventure Park
- Mamanuca Environment Society
- Ministry of Agriculture (Fiji)
- Ministry of Local Government, Housing & Environment (Fiji)
- Nadroga/Navosa Provincial Office
- National Trust of Fiji Islands
- NatureFiji-MareqetiViti
- New Zealand Department of Conservation
- Pacific Invasives Initiative
- San Diego Zoo Global
- South Sea Cruises
- Survivor Entertainment Group
- Taronga Conservation Society Australia
- US Embassy Suva, Fiji, Regional Environmental Affairs Office
- US Geological Survey, Western Ecological Research Center
- Yanuya Rugby Team.

Consultations with landowners were (and will continue to be) conducted before, during and after each activity or site visit. Transparency of all information and intentions of any actions are disclosed to stakeholders (BirdLife, 2011b).

From 2009–2010, discussions that addressed goat grazing on the island were held with landowners. Although the BirdLife surveys in 2009 confirmed goats to be a significant factor in the decline of the native species, goats were also a contributor to the village of Yanuya's financial income through market sale or occasional harvest for meat. Therefore, compensation for the village of Yanuya to halt goat grazing was agreed upon by the community and supported by various stakeholders. In 2010 a Memorandum of Understanding (MOU) was signed with the Mataqali Vunaivi, National Trust, and Kula Eco Park (now Kula Wild Adventure Park) for rat and goat removal, and iguana harvest for the captive breeding programme. The National Trust of Fiji and BirdLife International jointly carried out operations to eradicate rats in August of 2011 and goats between June 2010 and November 2011. At the same time, it was decided the best way to conserve the crested Fijian iguana was to harvest 10 sexually mature adult pairs (N = 20) from the remaining iguanas on the island for captive breeding and subsequent reintroduction. From April 2010 to February 2012 Monuriki iguanas were collected and brought to captive breeding facilities located at Kula Eco Park as part of the MOU.

Goat removal and eradication

The local Yanuya Rugby Team was employed to muster and catch goats on the island utilising mustering routes and techniques established by the local communities from decades of catching goats on Monuriki. From June to November 2010, 151 goats were mustered from the island over 12 days, and as of January 2011 an estimated 20 goats remained (BirdLife, 2011a). Captured goats were taken to the Viti Levu mainland for sale. Two professional hunters from New Zealand using trained dogs eliminated more than 50 additional goats over an 11-day period in September 2011. A final four day follow up hunting effort in November 2011 detected no additional goats (BirdLife, 2011a). To compensate for expected revenue loss of these animals, the owners received FJ$100 per goat. Post-eradication monitoring of the forest vegetation using fixed photo points and of the wedge-tailed shearwater population was conducted to assess the response to the goat eradication (Rasalato, et al., 2012).

Rat eradication

Eradication of Pacific rats (*Rattus exulans*) was carried out by delivering specially formulated rodenticide (brodifacoum at 20 ppm) baits (PestOff 20R) from a helicopter using standard procedures and equipment including a specifically designed spreader bucket calibrated to the required application rate (20 kg/ha) and GPS (Seniloli, et al., 2011). To determine the success of the rat eradication, a series of transects with rat-trap (Victor Professional) and rat-detection stations were created in two to three main locations across the island for each monitoring event in 2012, 2013, 2015, 2016, and 2017. The transects were set up in areas of the eastern, northern, and south-western beaches with between 10 and 20 stations for each transect (Rasalato, et al., 2012; Fig. 3). Each station comprised at least one snap trap, but the first assessments also included a peanut butter wax tag, a tracking tunnel and a second snap trap. The peanut butter wax tags were nailed to trees at random heights so as to reduce hermit crab access. Ink pads were placed in tracking tunnels with roasted coconut placed on the pads to act as baits. Snap traps were also baited with roasted coconuts and positioned to minimise non-target interference (e.g. hermit crabs).

These monitoring stations were set-up for three trap nights and were maintained and checked daily for any signs of rat activity (Rasalato, et al., 2012).

Biosecurity control

There is on-going training and outreach to the local tour companies about the conservation activities on Monuriki Island. Furthermore, biosecurity protocols were established to help reduce the potential negative impacts of the tours and other visitations to the island. A biosecurity plan prepared in 2013 (Thaman & Niukula, 2013) is reviewed and updated every two years (Seniloli, et al., 2015). The plan includes three main biosecurity procedures: preventing the entry of invasive alien species, systematic checking for such species, and rapid response procedures if any are found. Measures include the establishment of a community-based ranger programme to train local rangers on invasive species surveys, response methods and the prevention of wildlife poaching.

Kula Eco Park captive breeding programme

Concurrently with the rat and goat eradication efforts (2010–2011), 20 Monuriki iguanas were harvested and brought to Kula Eco Park, on the main Fijian island of Viti Levu, to develop a captive breeding colony. Pairs were successfully bred in managed care with the intention of re-introducing the offspring to their home island once the forest vegetation had recovered from grazing.

Species recovery

Monitoring for native species recovery has taken place so comparisons to pre-eradication surveys can be conducted. This includes using standard protocols for the iguanas and shearwaters, and recording other incidental species recoveries (Harlow, et al., 2007; Rasalato, et al., 2012). Vegetation surveys were conducted in fixed plots prior to the mammal eradications, and these plots were resurveyed in 2016, after the eradications, following the same survey protocols (Harlow, et al., 2007).

RESULTS

Goat removal and eradication

Since the goat removal and eradication was undertaken, there have been no detections of goats on the island during the last five years (through to 2017), confirming this action to be a success.

Rat eradication

In March 2012, five months after the helicopter spread of rodenticide, no rats were trapped nor were there any obvious signs of rat presence such as droppings, gnawed fruits or sightings (Rasalato, et al., 2012). Similarly, no indications of rat presence were found during subsequent assessments to date (2013, 2015, 2016, 2017), confirming eradication of Pacific rats from Monuriki Island and an ongoing rat free status generally.

Fig. 3 Monuriki Island showing the transects with the rat sampling stations on the west and east beaches.

Biosecurity control

Ships/vessels, commercial and private, are required to abide by the biosecurity measures detailed in the most recent version of the biosecurity plan (Seniloli, et al., 2015). These measures include setting up and maintaining bait and trap stations near wharfs and landing sites, regular checks of vessels for alien species stowaways, proper storage of food and regular decontamination of equipment including footwear; detection of any alien species on the island requires immediate reporting to a designated regional support centre. Although outlined, implementation of all of these actions has been slow, due to lack of resources. Camera stations to detect and assess risk of reinvasion, especially by rodents, were set up to monitor tourist visitations at designated areas of Monuriki. Some cameras are obvious and others are hidden; this programme helped identify what items are being brought ashore. Community awareness and involvement in implementing and enforcing biosecurity measures has been important in preventing additional invasive species from establishing on the island. This includes the cooperation of tour boats and yachts that must now follow biosecurity guidelines by having to report to local landowners or designated personnel and crew for a biosecurity briefing before anchoring near the island, although this is still to be fully enforced (NTF, 2012).

Kula Eco Park captive breeding programme

After four wet seasons following goat removal, the vegetation of Monuriki showed significant recovery, so we initiated the reintroduction of iguanas from Kula Eco Park. In mid-May 2015, 32 captive-bred crested iguanas, all implanted with unique PIT tags, were released into four different areas on Monuriki Island (Chand, et al., 2016) after a major community ceremony highlighting this milestone in the programme. Community members participated in the release of the iguanas and the event has been recorded in a video documenting the story (<https://vimeo.com/163325268>). In February 2017, 32 additional captive bred juveniles along with 16 of the original wild caught adults were released; 10 of each group were tracked for five months to measure post-release survivorship. Because this event signalled the end of the captive breeding programme and to thank the community for their permission and participation, a second major community event involving many levels of the Government was planned around the release. Only a few young iguanas remain in captivity for release in 2018 or 2019.

Released iguanas were monitored in 2017 and any wild captured individuals or recaptures from the 2015 release were measured and weighed to document post-release growth and general health. Transmitters used to help track released iguanas were removed before final re-release after the five-month period. Currently, the crested iguana population on Monuriki is recovering with the release of captive breeding programme animals, and naturally with existing wild animals (see below). Due to this success the captive breeding programme was ended on 24 February 2017, after the final release of the remaining 16 wild founder iguanas. Overall, between 2015 and 2017, a total of 80 iguanas, including the founders, were released into the wild.

Species recovery

Between February and June 2017, 35 wild iguanas (not passing through captivity) were caught and marked. Many of these were young animals that would have hatched after the rat eradication was completed. This sample of iguanas revealed that, within five years, the small remaining population of iguanas is reproducing and recruiting back into the recovering habitat.

A monitoring survey was conducted for wedge-tailed shearwater nests post-eradication. Out of the 159 burrows searched, 110 (69%) were occupied by chicks and one had an egg. Pre-eradication wedge-tailed shearwater nest site occupancy was 41% whereas post-eradication this had risen to 69% occupancy (Rasalato, et al., 2012). The size of the wedge-tailed shearwater population on the south-western beach colony during the 2011–2012 seasons was estimated to be 1383 breeding pairs (Rasalato, et al., 2012). As an additional positive outcome, banded rails (*Gallirallus philippensis*) and peregrine falcons (*Falco peregrinus*; critically endangered in Fiji) are more frequently encountered on Monuriki following rat and goat eradications.

Habitat monitoring to document terrestrial dry forest habitat recovery following goat removal was conducted in 10 previously established 100 m^2 plots in the lowlands behind Rogua Beach. Vaō trees dominated the forests on Monuriki before the eradications. This species is unpalatable to both goats and iguanas. However due to goats' removal of the edible understory and lower branches of the taller edible trees, the goats were now starting to chew on the stems of the vaō. During the presence of goats and rats, vaō comprised 91.3% of the seedlings per plot, but only 58.7% of the seedlings per plot after goats and rats were removed. Individual vaō trees (> 2 m in height) had decreased from 155 to 78 individuals across the plots after mammal removal. Although this plant is native, it spreads like an invasive with disturbance. There is no plan to control it, but the recovery of the other native forest diversity will reduce its cover over time. For example, only three other tree species seedlings were documented when goats were present; now there are about 11 species of seedlings present per plot including *Hibiscus*, *Diospyros*, and *Pongomia*. Ground cover of vine (some invasive but edible such as *Passiflora foetida*) increased from 0% with goats to 30.2% after goats were removed. Habitat recovery was determined to be successful through these habitat surveys, although no similar repeated surveys have been done on the higher slopes where a greater diversity of dry forest trees are present.

DISCUSSION

Overall, the Monuriki Island restoration programme has been a great success and a model for Fiji and other nations in the region. Table 1 is a timeline that reviews the overall impacts to the island and the major milestones relevant to the plan. Monitoring will continue over the next decade as the tree canopy expands and invasive plants continue to be removed from the island. On the two main beaches, invasive plants are being removed manually and dry-forest trees are being planted within the coconut groves that persist. Over time these areas will recover to dry forest also. Most of the obvious habitat damage from cyclone Winston was on these coconut groves, breaking them in half, and the native trees on the ridges that were damaged seem to be recovering well from the event.

Table 1 Timeline of impacts and recovery actions on Monuriki Island.

Rattus exulans introduced to Fiji	ca. 3,000 years ago
Goats introduced to Monuriki	1970s
Iguanas discovered on Monuriki	1980
Iguanas captured for resort displays	1980s–2000s
Castaway was filmed	1999
IUCN Iguana Specialist Group Suva, Fiji	2004
Crested Iguana Species Recovery Plan	2008
Discussions with landowners over goats	2009–2010
Goat eradication operation	2010–2011
20 Monuriki iguanas harvested for captive colony	2010–2012
Rat eradication operation	2011
Goat and rat eradication confirmed	2013
32 captive-bred iguanas released	2015
Cyclone Winston (Category 5)	2016
Reality TV Show now being filmed on Monuriki	2016–2017
32 captive-bred iguanas released	2017
16 remaining original founder iguanas released	2017

Community engagement

This programme has continued with renewed investment in the local community through the development of a regional Ranger Programme for Fiji iguana conservation. Various stakeholders have supported the development and training of local residents to act as regional iguana experts and habitat managers. This programme includes capacity building by training local rangers in field survey techniques, habitat management methods, reforestation efforts, guest experience training for tourist interactions, and anti-poaching. Support for the Rangers and other local level science educators to attend conferences and workshops (such as the IUCN Iguana Specialists Group) geared toward engagement in conservation initiatives for threatened species has continued to provide valuable training and resources for long-term capacity building within Fiji. Education and outreach materials have been developed with the goal of reaching the regional children through programmes for the classrooms. Visits to the local communities to conduct outreach programmes have provided additional opportunities to reach the local communities and encourage their continued support in the conservation of the native threatened species.

To protect the regenerating forest, the community Rangers also established tourist hiking paths with the intent of educating visitors about tropical dry forest habitat while keeping their impacts on the island to a minimum. Interpretive kiosks are being developed and will be installed at the tourist beach.

Additional threats

In February of 2016, Tropical Cyclone Winston passed through Fiji with a peak intensity of ten-minute sustained winds of 230 km/hr that removed a significant proportion of the canopy leaves from trees on Monuriki Island (Fig. 4). Terrestrial surveys conducted in March 2016, indicated that the iguanas and birds were still present and increasing, but that long-term studies after this storm event would be critical for helping to understand species resiliency and recovery in the wake of massive tropical storms. BirdLife continues to monitor the wedge-tailed shearwater population and, while no assessment was made following cyclone Winston (which occurred during the chick feeding phase), population measures assessed will inform recruitment trends. The final 16 original founding iguanas, along with their 32 offspring produced as part of the captive breeding programme, were released after this extreme weather event (Table 1). Robust population monitoring for the iguanas will take place during 2018 to assess the longer-term survivorship of the released iguanas and track recovery of the remaining wild individuals.

2016–2017 Reality TV show, Survivor, now being filmed on Monuriki

Regional conservation efforts for Fijian iguanas have continued and grown to include additional partners. One such collaborator, Survivor Entertainment Group, has been strongly supportive of the programme since their arrival to the region. Although this partner may not be conventional when considering species conservation and habitat restoration, they have embraced and supported our efforts to save the Fijian iguanas from extinction and have assisted in efforts to continue local level engagement. There might be impacts of the filming activities on the shearwaters, but these are hard to measure, and are being minimised by marking active nest sites and putting in avoidance trails to move film crews and contestants around these sensitive sites. By providing significant local employment opportunities and incorporating and investing in biosecurity training methods and native species conservation as part of their local strategic plan they have become an advocate for these restoration and recovery efforts in the region.

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Fig. 4 Rogua Beach on Monuriki in March of 2016 after Cyclone Winston. Broken and denuded trees are seen along the beach and up the hillslope.

REFERENCES

BirdLife International. (2011a). *Monuriki and Kadomo Goat Eradication Operation Report*. Unpublished report. Suva, Fiji: BirdLife International, Pacific Secretariat.

BirdLife International, National Trust of Fiji, Pacific Invasives Initiative. (2011b). *Operational Plan for the Eradication of Feral Goats in the Mamanuca Islands, Republic of Fiji.* Unpublished report. Suva, Fiji: BirdLife International, Pacific Secretariat.

Chand, R., Niukula, J., Vadada, J., Fisher, R., Lovich, K., Pasachnik, S., Rasalato, S., Thaman, B., Seniloli, E., Tuamoto, T., Yanuya, T. and Harlow, P. (2016). 'Captive Breeding and Re-introduction of the Monoriki Island Crested Iguana in Fiji'. In: P.S. Soorae (ed.) *Global Re-introduction Perspectives: 2016. Case-studies from Around the Globe.* Gland, Switzerland: IUCN/SSC Reintroduction Specialist Group and Abu Dhabi, UAE: Environment Agency-Abu Dhabi.

Conservation International. (2005). *Conservation Outcomes: Polynesia and Micronesia, Fiji Inset, Draft Map, 1:3,200,000*. Washington, DC, USA: Conservation International and The Critical Ecosystem Partnership Fund.

Coulston, G., Ratu, M. and Nagle, B. (2010). *Monuriki Island Goat Removal; Report of a Feasibility Study Visit*. Unpublished report. Auckland, New Zealand: Pacific Invasives Initiative.

Donlan, C.J. and Keitt, B.S. (1999). 'Using research and education to prevent extinction'. *California Coast and Ocean* 15: 20–23.

Gibbons, J. (1984). 'Iguanas of the South Pacific'. *Oryx* 18 (2): 82–91.

Harlow, P.S. and Biciloa, P.N. (2001). 'Abundance of the Fijian crested iguana (*Brachylophus vitiensis*) on two islands'. *Biological Conservation* 98: 223–231.

Harlow, P.S., Fisher, M., Tuiwawa, M., Biciloa, P.N., Palmeirim, J.M., Mersai, C., Naidu, S., Naikatini, A., Thaman, B., Niukula, J. and Strand, E. (2007). 'The decline of the endemic Fijian crested iguana *Brachylophus vitiensis* in the Yasawa and Mamanuca archipelagos, western Fiji'. *Oryx* 41: 44–50.

Harlow, P.S., Hudson, R. and Alberts, A. (2008). 'Fijian Crested Iguana *Brachylophus vitiensis* Species Recovery Plan 2008–2012'. IUCN Species Survival Commission, Iguana Specialist Group. < http://www.iucn-isg.org/publications/actionsplans/>.

Keogh, J.S., Edwards, D.L., Fisher, R.N. and Harlow, P.S. (2008). 'Molecular and morphological analysis of the critically endangered Fijian iguanas reveals cryptic diversity and a complex biogeographic history'. *Philosophical Transactions of the Royal Society*, Series B. Vol. 363 (1508): 3413–3426.

McCreless, E.E., Huff, D.D., Croll, D.A., Tershy, B.R., Spatz, D.R., Holmes, N.D., Butchart, S.H.M. and Wilcox, C. (2016.) 'Past and estimated future impact of invasive alien mammals on insular threatened vertebrate populations'. *Nature Communications* 7: 12488.

NTF. (2012). *Restoring the Native Vegetation of Monuriki Island, Fiji. Final Report Submitted to Critical Ecosystems Partnership Fund*. Suva, Fiji: National Trust of Fiji Islands.

Olson, D., Farley, L., Patrick, A., Watling, D., Tuiwawa, M., Masibalavu, V., Lenoa, L., Bogiva, A., Qauqau, I., Atherton, J., Caginitoba, A., Tokota'a, M., Prasad, S., Naisilisili, W., Raikabula, A., Mailautoka, K., Morley, C. and Allnutt, T. (2010). 'Priority forests for conservation in Fiji: landscapes, hotspots and ecological processes'. *Oryx* 44: 57–70.

Rasalato, S., Niukula, J., Thaman, B., Mateiwai, M. and Vadada, J. (2012). *Post-eradication Monitoring Survey on Monuriki Island, Mamanuca group*. Unpublished field report. Suva, Fiji: BirdLife International, Pacific Secretariat.

Roberts, M. (1991). 'Origin, dispersal routes, and geographic distribution of *Rattus exulans*, with special reference to New Zealand'. *Pacific Science* 45: 123–130.

Seniloli, E. Rasalato, S. Tuamoto, T. and Cranwell, S. (2011). *Feasibility of Rat and Goat Eradication in the Mamanuca group Fiji*. Unpublished report. Suva, Fiji: BirdLife International, Pacific Secretariat.

Seniloli, E., Rasalato, S. and Tuamoto T. (2015). *Draft Biosecurity Action Plan for Monuriki and Kadomo Islands, Mamanuca Group, Fiji.* Suva, Fiji: Nadroga/Navosa Provincial Council, BirdLife International, National Trust of Fiji Islands, NatureFiji-MarqetiViti.

Thaman, B. and Niukula J. (2013). *Draft Biosecurity Action Plan for Monoriki and Kadomo Islands, Mamanuca Group, Fiji.* Suva, Fiji: Nadroga/Navosa Provincial Council, BirdLife International, National Trust of Fiji Islands, NatureFiji-MarqetiViti,

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?' *Biological Invasions* 8: 863–891.

Costs and benefits for biodiversity following rat and cat eradication on Te Hauturu-o-Toi/Little Barrier Island

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Abstract Considerable benefits can be achieved for indigenous biodiversity when invasive vertebrates are removed from islands. In New Zealand, two logistically challenging eradications were undertaken, one to remove cats (*Felis catus*) and the other Pacific rats (*Rattus exulans*) from Te Hauturu-o-Toi/Little Barrier Island (Hauturu). Here we document the short- and long-term impacts of these interventions on the biodiversity of Hauturu. We also assess the extent to which predicted outcomes were reflected in the measured responses for a wide range of species. Short-term impacts of the eradication program encompassed individual mortality for some native species but no measurable impact to populations. In contrast, at least 11 native vertebrates and one native invertebrate species increased in abundance after rat and cat removal. Fifteen of 34 plant species monitored had significantly more seedlings on Hauturu after rat eradication compared with control islands, indicating future changes in forest composition. Several native species previously not recorded on the island were discovered, including the New Zealand storm petrel (*Fregetta maoriana*) (formerly considered extinct), the forest ringlet butterfly (*Dodonidia helmsi*) and eight species of aquatic invertebrate. The chevron skink (*Oligosoma homalonotum*) has been found in increasing numbers and tuatara (*Sphenodon punctatus*), raised in captivity on the island, are now re-established and breeding in the wild. These results illustrate an island gradually recovering after a long period of modification. We conclude that more success stories such as Hauturu must be told if we are to allay the public's concerns about such eradication campaigns. And more public support is required if the conservation community is to tackle invasive species at a scale commensurate with the threats they pose.

Keywords: conservation management, ecosystem, restoration, species recovery

INTRODUCTION

Worldwide, more than 1,000 invasive vertebrate eradications have been successfully completed to prevent biodiversity loss (DIISE, 2017) and many benefits to species and ecosystems have been documented (Jones, et al., 2016). However, eradication projects continue to attract controversy (e.g. Howald, et al., 2010; Griffiths, et al., 2012; Capizzi, et al., 2019) suggesting that, despite transparent consultation processes, sectors of the public remain unconvinced of the relative cost benefits of this conservation strategy.

To illustrate the value of invasive vertebrate eradication, we present the short- and long-term impacts on biodiversity following the removal of cats (*Felis catus*) and Pacific rats (*Rattus exulans*) from Te Hauturu-o-Toi/Little Barrier Island (hereafter referred to as Hauturu). Specifically, we ask whether the claimed benefits of cat and Pacific rat eradication were met.

The eradication of cats from Hauturu raised little public concern and, under New Zealand environmental law, did not require consent. In contrast, the proposed rat eradication raised cultural and environmental concerns and, because rodent bait was broadcast by helicopter, required local government consent (Resource Management Act 1991). Some members of the public were opposed to the aerial application of rodent bait and some Māori iwi (tribes) contested the removal of rats because of their cultural significance. Consequently, public hearings were held and an Assessment of Environmental Effects (AEE) (Griffiths, 2002) was presented to a panel of independent commissioners. The AEE identified the legal mandate for the removal of rats and the risk to native species if rats remained. The application was approved as the potential benefits to native biodiversity were judged to outweigh the short-term environmental costs.

Cats were removed from Hauturu in an operation that spanned four years from 1977 to 1980. To support this work, a 67 km long track network was established across the island and three huts built at strategic locations (Veitch, 2001). Leg-hold traps and baits containing the toxin 1080 were the principal methods employed to remove cats, although cage traps, the introduction of pathogens and dogs were also used (Veitch, 2001). Mitigation of potential impacts on non-target species was undertaken through careful placement of traps.

Rats were eradicated in 2004 by the New Zealand Department of Conservation (DOC) in an operation utilising the aerial application of Pestoff 20R™ rodent bait containing brodifacoum at 20 ppm (Griffiths, 2004). Rodent bait was applied twice by three helicopters in two successive operations during winter, the first on 8 and 9 June and the second on 12 July. At the same time, baits were placed in bait stations within all buildings and huts on the island. The operation used a total of 55 tonnes of rodent bait with rates for the first and second bait applications averaging 11.7 kg/ha (ca. 1 bait per 1.7 m^2) and 6.16 kg/ha (ca. 1 bait per 3.2 m^2), respectively, across the island. The success of the eradication operation was confirmed in January 2006 after extensive monitoring both on and off the track network across the island with tracking tunnels, spotlight searches and indicator dogs (Griffiths, 2006).

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Our findings are based on research and monitoring completed on the island from 1962 to the present. We review published and unpublished studies, but also present new data that illustrate changes following rat and cat removal.

STUDY AREA

Hauturu (3,083 ha), 36°11′56.76″S, 175°4′53.04″E is almost midway between Great Barrier Island/Aotea and the mainland (Fig. 1). Rugged and steeply dissected, the island arose from the partly eroded core of a composite volcanic cone that formed 1.5–3 Ma (Lindsay & Moore, 1995)(Fig. 2). Hauturu was first settled around the 14th C by the descendants of the Maori ancestor and voyager Toi te Huatahi and was occupied continuously until the arrival of Europeans in the 1800s. Over this and the ensuing period of European settlement, approximately one third of the island (the south-west) was cleared, burnt and subjected to grazing by sheep (*Ovis aries*) and cattle (*Bos taurus*) (most of these areas have since reverted to native vegetation and are now secondary successional forest or older). Because of its unique forest and threats to the diverse birdlife, the island was gazetted as New Zealand's first Nature Reserve in 1896 (Young, 2004).

Except for feral cats and rats, the island escaped many of the invasive vertebrate introductions to the main islands of New Zealand and, consequently, its fauna is still largely representative of northern New Zealand prior to European colonisation. By area, the island supports a greater diversity of native birds and reptiles than any other part of of New Zealand. Nonetheless, the introduction of rats and cats to Hauturu had a huge impact. Pacific rats, considered to have arrived early during the period of Maori settlement (Campbell, 2011), likely extirpated most of the small seabird species still seen on nearby rat-free islands. Rat predation may also explain the absence of milk tree (*Streblus banksii*), coastal maire (*Nestegis apetala*) (Campbell, 2011), large land snails and slugs on the island (Campbell, 2011). The introduction of cats sometime around 1867 resulted in the extinction of the last population of North Island snipe (*Coenocorypha barrierensis*) and extirpation of the tieke (*Philesturnus rufusater*), grey-faced petrel (*Pterodroma macroptera*), and probably other seabirds. It is likely that other species such as the black petrel (*Procellaria parkinsoni*) would have suffered a similar fate had cats not been removed.

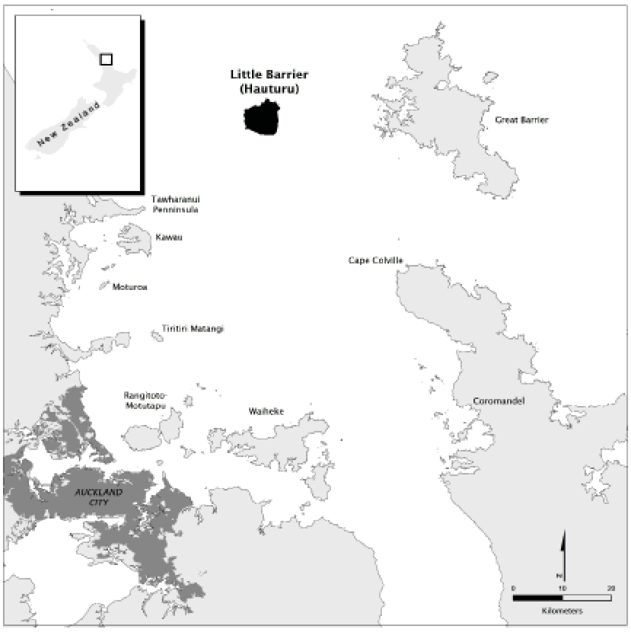

Fig. 1 Map showing the location of Hauturu.

METHODS

A literature search of published and unpublished monitoring, undertaken to measure the environmental impacts of cat and rat eradication, was conducted but also included general research and monitoring completed on Hauturu for other reasons. We assessed impacts as measured costs, measured benefits, and unknown costs or benefits to biodiversity. Unknown costs and benefits were largely a function of an absence of monitoring prior to and following eradication, and/or due to environmental changes unrelated to the eradications. For example, Hauturu supports breeding populations of the New Zealand lesser short-tailed bat (*Mystacina tuberculata*) and long-tailed bat (*Chalinolobus tuberculatus*), both endemic to New Zealand. Bat populations were not monitored before, during or after cat and rat eradication.

We first summarise the predicted outcomes (Tables 1 and 2), then collate previously published monitoring results obtained from a Web of Science and Google Scholar search completed on 24 January 2017 using the key words 'Little Barrier Island' and 'Hauturu'. We then summarise unpublished research and monitoring reports and other unpublished data including the methods used (Table 3) and analyse data on terrestrial birds (see mist-netting below). Changes in island species composition after rat and cat eradication were determined by comparing recent literature with historical reports. For simplicity, results are grouped by taxa: marine birds, terrestrial birds, reptiles, freshwater fish, terrestrial invertebrates, aquatic invertebrates and terrestrial plants.

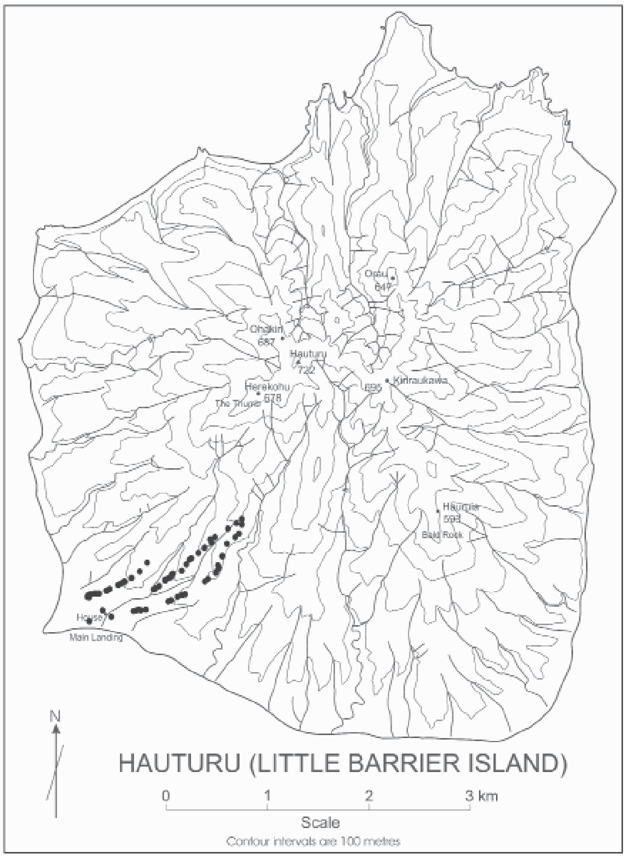

Fig. 2 The rugged nature of Hauturu and locations of mist nets (black dots) used to sample forest bird abundance before and after rat eradication.

Table 1 Predicted benefits of cat eradication from Hauturu with species marked* identified in Recovery Plans for threatened species.

Native biodiversity	Evidence	References
Black petrel	Direct evidence of cat predation and declining breeding distribution	(Girardet, et al., 2001)
Cook's petrel/titi	Direct evidence of cat predation and declining breeding distribution	(Imber, et al., 2003a)
Tieke	Potential for reintroduction	(Girardet, et al., 2001; Hoosen & Jamieson, 2003)
Kakapo*	Potential for ex-situ management of Stewart Island population heavily impacted by feral cats	(Lloyd & Powlesland, 1994; Anon., 1996; Elliott, et al., 2001)
Kokako*	Potential for reintroduction	(Innes & Flux, 1999)
Grey-faced petrel	Recolonisation expected	(Girardet, et al., 2001)

Table 2 Predicted responses to rat removal from Hauturu as identified in the assessment of environmental effects (Griffiths, 2002), with conventions as in Table 1.

Native biodiversity	Evidence	References
Species that faced local extinction in the absence of intervention		
Cook's petrel	Fledgling recruitment on Whenua Hou (Codfish Island) increased to 90% following rat eradication	(Imber, et al., 2003a)
Tuatara*	Documented impacts on juvenile recruitment. Local extinction predicted by population models completed for Marotere and Taranga Islands	(Gaze, 2001; Towns, et al., 2007; Cree, 2014)
Wetapunga*	No off-site data but examples of localised extinctions in other flightless crickets e.g. tusked weta (*Motuweta isolata*; Mercury Islands)	(Towns, et al., 1990; Sherley, 1998; Towns, et al., 2006)
*Dactylanthus taylorii**	No off-site data on impacts by Pacific rats on this species but video evidence of inflorescence destruction by rats on Hauturu	(Eckroyd, 1995)
Giant-flowered broom	Examples of localised extinctions from other archipelagos (e.g. Marotere Islands)	(Towns, et al., 2003)
Species predicted to benefit from intervention		
Grey-faced petrel	Increases in abundance documented elsewhere after Pacific rat removal (e.g. Korapuki Island, Stanley Island)	(Towns & Atkinson, 2004; G Taylor pers. comm.)
Diving petrel	Localised extinctions reported in other archipelagos and increased abundance following Pacific rat removal (e.g. Mercury Islands)	(Towns & Atkinson, 2004; G Taylor, pers. comm.)
Tieke	Evidence of increased abundance after Pacific rat removal (e.g. Red Mercury Island)	(Robertson, et al., 1993)
Towns' skink	Examples of localised extinctions in other archipelagos (e.g. Mokohinau, Marotere Islands) or confinement to refugia (Hauturu)	(Towns, et al., 2003)
Duvaucel's gecko	Confined to refugia in presence of Pacific rat (Hauturu). More terrestrial activity and increased abundance after Pacific rat removed (e.g. Mercury, Marotere and Ohinau Islands)	(Towns, 1996; Hoare, et al., 2007)
Pisonia brunoniana and 10 other plant species	Examples of recovery after Pacific rat removal (e.g. Mercury Islands)	(Campbell & Atkinson, 2002)
Species that will possibly benefit from intervention		
Short-tailed bats	No off-site models for this species. Potential release from competition for invertebrates and *Dactylanthus taylorii* inflorescences	
Long-tailed bats	No off-site models for this species. Potential release from competition for invertebrates	
Smaller native passerine birds	No off-site models for these species. Potential release from competition for invertebrates and nectar sources	
Day-active skinks	Increased abundance of selected species when Pacific rats removed (e.g. Mercury Islands) but no models for chevron and striped skinks present on Hauturu	(Towns, 1991)

Table 3 Methods used to evaluate short- and long-term changes to biodiversity subsequent to cat and rat eradication on Hauturu.

Species Group	Indicator	Methods	Sources
Bats	Species composition	Literature review	(Daniel & Williams, 1984)
	Species abundance	Anecdotal reports	
	Short term mortality over the course of eradication operations	Cat trapping data and carcass searches before and after rat eradication.	(Veitch, 2001; Griffiths, 2004)
Marine birds	Species composition	Literature review	(Hutton, 1868; Turbott, 1947; Girardet, et al., 2001; Stephenson, et al., 2008; Rayner, et al., 2009; Gaskin & Rayner, 2013; Rayner, et al., 2015)
	Cooks petrel breeding success and distribution	Monitoring of marked burrows 1971 to present	(Imber, et al., 2003a; Rayner, et al., 2007b)
	Black petrel breeding success	Monitoring of marked burrows 1971 to present	(E. Bell, unpubl. data; Imber, 1987; Imber, et al., 2003b)
	New Zealand storm petrel	Monitoring of marked burrows	(M.J. Rayner, unpubl. data; Ismar, et al., 2015)
	Short term mortality over the course of eradication operations	Cat trapping data and carcass searches before and after rat eradication	(Veitch, 2001; Griffiths, 2004)
Terrestrial birds	Species composition	Literature review and bird counts	(Hutton, 1868; Turbott, 1947; Girardet, et al., 2001; Veitch, et al., 2019)
	General species abundance	Mist-netting capture rates and bird counts	(Girardet, et al., 2001; Veitch, et al., 2019). Data analysis of unpublished mist-netting data described below.
	Tieke, hihi and tui abundance.	Distance sampling and bird counts	(Toy, et al., in press; Veitch, et al., 2019)
	Kiwi	Call counts	(Wade 2009; Wade 2014a)
	Short term mortality over the course of eradication operations	Cat trapping data and carcass searches pre and post rat eradication	(Veitch, 2001; Griffiths, 2004)
Reptiles	Species abundance and distribution.	Pitfall trapping (10 L plastic buckets baited 24 h with tinned pear) and search effort	(Brown, 2013)
	Short term mortality over the course of eradication operations	Cat trapping data and carcass searches before and after rat eradication	(Veitch, 2001; Griffiths, 2004)
Freshwater fish	Species composition	Trapping and spotlight surveys	(Winterbourn, 1964; Wade, 2014b)
Terrestrial invertebrates	Species composition	Anecdotal observations	(S.Wheatley, pers. comm.; R. Walle, pers. obs.)
	Wetapunga abundance	Detections per unit of search effort	(Green, et al., 2011)
Aquatic invertebrates	Species composition	Benthic sampling and light trapping	(Winterbourn, 1964; Wade, 2014b)
Threatened native and invasive alien plants	See production, seedling recruitment; abundance and distribution	Monitoring of seed set for *D. taylorii*; search effort for other threatened species and priority weeds	(D. Havell, unpubl. data; Campbell, 2011)
Canopy trees, palms and lianes	Juvenile recruitment on rat-inhabited versus rat-free islands, post-eradication response, seedling response in exclosures	Seedling numbers of 34 species counted on marked linear plots, twice before rat eradication and two years after on Hauturu and on two control islands with rats	(Campbell, 2011)

Mist-netting

Data collected from mist-netting completed before and after rat eradication were used to assess the impact of the application of rodent bait on the abundance and composition of forest birds. As these data are not published we summarise it here. Four trips were completed, one prior (January 2004) and three subsequent (August/September 2004, February 2005, August 2005) to the eradication comprising 413 mist-netting events at 69 sites across five valleys in an area of approximately 350 ha on the south-west side of the island (Fig. 2). Each trip lasted for between five and seven whole days of mist-netting. Each mist-netting event had an average duration (±sd) of 399 minutes (6 hours 39 minutes) ± 170 minutes and ranged from 06:11

(opening time) to 19:47 (closing time). On average, the median time a net remained open was 11:44 hrs.

In the analyses we assessed whether temporal factors (e.g. year, season, day, time of day) were directly associated with variability in the number of individuals and species caught in mist-nets. All analyses were compiled in SAS V.9.0. Generalised linear mixed models (with Poisson distributed errors) were used to assess the variability between both the total number of individuals (bird abundance) and the number of species (species richness) caught per mist-netting event with respect to temporal factors.

With the data assumed to follow a Poisson distribution due to its non-negative, count nature, we used Basic Generalised Linear Models followed by a more complicated Generalised Linear Mixed Effects Model to tease out changes between years for individual species. Richness and Shannon diversity were the variables being predicted, with Year, Season, Total Number of Birds, and Corrected Net Length as explanatory variables. Site and Net were included in the model as random effects, separately and together. The 'best' model was then used to fit the six species most commonly caught as the predictor variable.

RESULTS

Potential and actual costs to native biodiversity

Seabirds

Thirteen Cook's petrel (*Pterodroma cookii*) were trapped in cat leg-hold traps and euthanised (Veitch, 2001). The breeding success of Cook's petrel decreased following the removal of cats then increased after rat eradication. This was hypothesised to be a function of mesopredator release resulting in higher numbers of rats at higher elevations after cat eradication leading to greater impacts on Cook's petrel breeding success (Rayner, et al., 2007b). No other short term negative impacts on seabirds as a result of cat and rat eradication were observed.

Terrestrial birds

Thirty-two kiwi (*Apteryx mantelli*) were caught in traps during the cat eradication. Two were euthanised, the rest released unharmed (Veitch, 2001). Three kiwi were found dead after application of rodent bait (Fisher, et al., 2011) and are presumed, based on the necropsy of one individual, to have died from secondary poisoning. Despite the loss of these individuals, no change in calling frequency was observed in kiwi call count surveys completed after the rat eradication (Wade, 2009).

Individual mortality following bait application to target rats was documented for eight other terrestrial bird species including blackbird (*Turdus merula*), robin (*Petroica australis*), pukeko (*Porphyrio porphyrio*), kakariki (*Cynoramphus novaezelandiae* and *C. auriceps*), harrier (*Circus approximans*), kaka (*Nestor meridionalis*) and morepork (*Ninox novaeseelandiae*) (Veitch, 2001; Fisher, et al., 2011). Numbers of each species found after the rat eradication are presented in Fisher et al. (2011). However, no significant short-term population impacts were detected in an analysis of bird count data collected over the course of the cat eradication (Girardet, et al., 2001). Bird counts from 2012 to 2017, after rat eradication, using the same methods as Girardet, et al. (2001), showed no significant change in overall abundance but significant changes in the abundance of some species (C.R. Veitch unpubl. data).

Data collected from mist-netting conducted before and after rat eradication showed no significant change (either increase or decline) in any of three components of catchability (bird abundance, species richness, species composition) analysed for forest bird species. The only change measured was a significant increase in the number of both bellbird *Anthornis melanura* and parakeet (*Cyanoramphus novaezelandiae* and/or *C. auriceps*) captured. In total, 1,570 birds (twenty-three species) were caught in mist-nets. The total number of birds (bird abundance) caught varied between seasons ($F_{1,374} = 5.53$, $P = 0.02$) but did not differ between periods of mist-netting completed before and after the bait applications targeting rats ($F_{1,374} = 0.34$, $P = 0.56$). Number of species (species richness) ($F_{1,374} = 0.01$, $P = 0.93$) and relative similarity in the composition of forest birds caught in mist nets ($F_{1,4} = 0.53$, $P = 0.51$) did not differ significantly over the course of the rat eradication. These data correspond with the findings of a non-toxic bait trial completed ahead of the rat eradication, that determined the risk to terrestrial bird populations to be low (Greene & Dilks, 2004).

Reptiles

No individual mortality as a result of the cat and rat eradications was documented. However, anecdotal evidence and pitfall trapping (Fig. 3) suggest an unexplained decline in skink numbers following cat eradication, prior to rats being removed.

Freshwater fish

A freshwater fish survey conducted in 2000 detected redfin bullies (*Gobiomorphus huttoni*), banded kokopu (*Galaxias fasciatus*) and longfin eel (*Anguilla dieffenbachii*) (McGlynn, et al., 2000). No mortality following bait application for rats was observed, but a less extensive survey completed in 2009 detected only banded kokopu and longfin eels (Wade, 2014b).

Invertebrates

As predicted, no negative impacts on invertebrates were observed.

Plants

Aside from small-scale clearance of vegetation to form the trail network to complete cat eradication, no negative impacts on plants were observed but a greater impact on some plant species may have resulted from the release of rats from cat predation.

Potential and measured benefits to native biodiversity

Seabirds

Cooks petrel breeding success in high altitude habitats (with 90% of the population), averaged 5% prior to rat eradication but increased to approximately

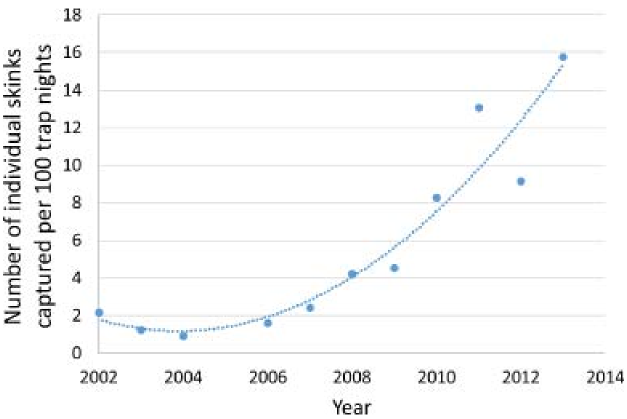

Fig. 3 Catch per unit effort for all skink species combined on Hauturu between 2002 and 2013 (Source: Department of Conservation, Warkworth, New Zealand).

60% the following breeding season as a result of reduced predation pressure (Rayner, et al., 2007a; Rayner, et al., 2007b). Improved Cook's petrel breeding success was also circumstantially reflected in a tenfold increase in the number of recently fledged chicks presented at bird rescue centres on the adjoining mainland the season following eradication and ongoing (M.J. Rayner, unpubl. data). An obvious massive increase in the extent and volume of nocturnal vocalisation by Cook's petrels has been observed over the last 12 years suggesting ongoing population recovery facilitated by increased breeding success and recruitment (M.J. Rayner, pers. obs.).

Up to 600 pairs of black petrel are now thought to breed on Hauturu (Bell, et al., 2016) up from the 50–100 pairs estimated by Imber (1987) prior to cat eradication. Similarly, breeding success increased from 1977 (50%), 1978 (60%) and 1996 (71.8%) (Imber, unpubl. data) to 2015/16 (85%). Before 1980 up to 67% of fledglings emerging from burrows were killed by cats and fewer than 5% of chicks were expected to have fledged (Imber, 1987). Between 1 and 28% of adult black petrels were also killed by cats at the colony between 1972 and 1976 (Imber, 1987). Comparisons of breeding activity within the same burrows during 1996/97 and 2015/16 showed a stable occupation rate of ca. 57% over the 19-year period and ca. 3% decline in breeding activity (Bell, et al., 2016).

New Zealand storm petrels (*Fregetta maoriana*), thought to be extinct for 110 years, were rediscovered at sea in 2003 and, after much effort, were located breeding on Hauturu in 2013 (Stephenson, et al., 2008; Rayner, et al., 2015). Mark recapture data collected between 2015 and 2017 suggest a minimum population size of 1,000 individuals (M.J. Rayner unpubl. data) and, based on at sea sightings, the population is steadily increasing.

Grey-faced petrels were discovered breeding after an apparent 60-year absence in 2009 and anecdotal observations of old colony sites suggest a gradual increase in these populations (M.J. Rayner, unpubl. data; Rayner, et al., 2009). The calls of other seabird species, such as common diving petrels (*Pelecanoides urinatrix*) and fluttering shearwaters (*Puffinus gavia*), have been also documented subsequent to rat eradication (M.J. Rayner unpubl. data) and may reflect recolonisation of the island's coastline by these predator-sensitive species.

Terrestrial birds

Three bird species have been introduced or reintroduced to Hauturu since the removal of cats: kakapo (*Strigops habroptilus*) during 1982, kokako (*Callaeus wilsoni*) during 1980–1988 and tieke during 1984–1988. Following their reintroduction, both tieke and kokako populations expanded rapidly and are now abundant across the island (K. Parker & I. Flux, pers. comm.). Kakapo were removed from the island in 1998 due to ongoing nest predation by rats but were re-established in 2012. Breeding by some individuals has subsequently been documented but whether the population will ever become self-supporting is at present unknown (L. Joyce, unpubl. data).

Annual distance sampling completed between 2005 and 2013 in the south-west of the island initially charted a decrease in numbers of hihi (*Notiomystis cincta*) and tui (*Prosthemadera novaeseelandiae*) (Toy, et al., 2018). Hihi numbers appeared to stabilise from 2009 onwards but the density of tui continued to vary. The recorded density of tieke changed little over the survey period. Forest bird counts undertaken between 2013 and 2017 within the same area recorded significantly higher numbers of bellbird, tomtit (*Petroica macrocephala*), parakeets, robin, kokako and tieke and a decline in numbers of whitehead (*Mohoua albicilla*), tui, hihi, rifleman (*Acanthisitta chloris*), grey warbler (*Gerygone igata*), blackbird and silvereye (*Zosterops lateralis*) when compared to counts undertaken before and during the cat eradication and prior to the rat eradication (C.R. Veitch, unpubl. data). No significant change was detected in the overall number of forest birds (C.R. Veitch, unpubl. data).

No significant change in calling frequency was detected in kiwi call count surveys over the period 1993 to 2014 although frequencies recorded were consistently higher than sites monitored on the North Island of New Zealand (Wade, 2014a). Despite the return of the brown teal (*Anas chlorotis*) that were removed during the rat eradication and the introduction of additional individuals, the brown teal population has not expanded. The brown teal population is considered permanent, but numbers present may be more a reflection of the species' breeding success on nearby Great Barrier Island (Aotea).

Banded rail (*Gallirallus philippensis*), last seen on the island in 1946 (Sibson, 1947), have returned to the island and reared young (C.R. Veitch, unpubl. data) and spotless crake (*Porzana tabuensis*), never previously recorded on the island, are now present and breeding (C.R. Veitch, unpubl. data). Another short-term impact worthy of note is the appearance and establishment of bellbirds at Tawharanui Regional Park subsequent to rat eradication (Brunton, et al., 2008). Invasive vertebrates were removed from Tawharanui at the same time as the rat eradication on Hauturu and this coupled with an increase in the number of bellbirds (as indicated by mist-netting data) may have created conditions suitable for dispersal and subsequent population establishment.

Reptiles

Following rat removal, numbers of reptiles caught in pitfall traps steadily increased (Fig. 3). Towns (*Oligosoma townsi*), moko (*O. moko*) and shore skink (*O. smithi*) showed the biggest increase (see Fig. 4), contributing to an 18-fold increase in the total number of skinks caught per 100 trap nights since the rat eradication (Brown, 2013). Although numbers are too low to quantify changes to the island's chevron skink (*Oligosoma homalonotum*) population, the number of additional skinks found after rat eradication may indicate population recovery. Prior to the rat eradication only one chevron skink had ever been found on the island. Four have been found since rats were removed.

Limited monitoring of the island's gecko populations was undertaken, but spotlight surveys completed in 2009 and 2013 suggest populations are recovering

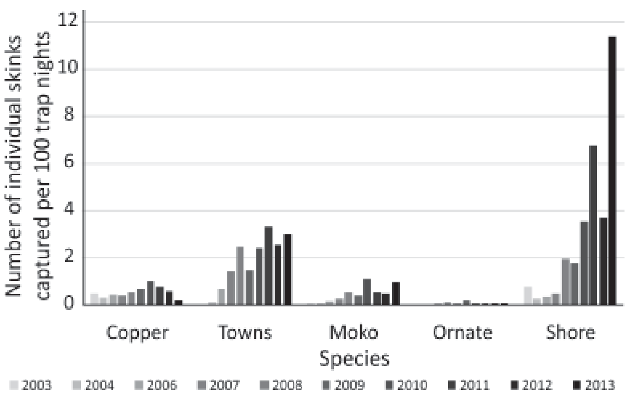

Fig. 4 Catch per unit effort for five species of skinks on Hauturu between 2002 and 2013 (Source: Department of Conservation, Warkworth, New Zealand).

from pre-eradication declines. Sighting rates of Pacific (*Dactylocnemis pacificus*), forest (*Mokopirirakau granulatus*) and common (*Woodworthia maculatus*) gecko in 2013 more than doubled relative to 2009 (Brown, 2013) and Duvaucel's gecko (*Hoplodactylus duvaucelli)* have been sighted more frequently (Hoare, 2009).

Tuatara (*Sphenodon punctatus*) were thought to be extinct on Hauturu until the species was rediscovered in 1991–1992 (Whitaker & Daugherty, 1991). Nine adults were taken into captivity on the island to ensure the relict population did not go extinct before rats were removed (Moore, et al., 2008). Since the rat eradication was confirmed successful in 2006 more than 196 young tuatara, raised in captivity, have been released at three sites and all captive adult tuatara have been returned to the wild. Additional adult survivors have been detected, breeding in the wild population has been noted and the population appears to be expanding (S. Keall, pers. comm.).

Aquatic invertebrates

A survey of aquatic invertebrates completed in 1963 was repeated in 2014 to identify changes in faunal composition. In total, 33 macroinvertebrate taxa from 12 orders were recorded from benthic samples. Six species of mayfly (*Mauiulus luma, Isothraulus abditus, Zephlebia spectabilis, Arachnocolus phillipsi, Ichthybotus hudsoni* and *Neozephlebia scita*), and two species of caddisfly (Trichoptera) (*Oxyethira albiceps* and a Chathamiidae sp.) not recorded in the 1963 survey, were found in 2014 (Wade, 2014b).

Terrestrial invertebrates

An annual monitoring programme to assess the recovery of wetapunga (*Deinacrida heteracantha*), New Zealand's largest giant weta, was instigated in 2005, a year after rats were removed. Numbers of wetapunga found in each survey had more than doubled by 2009 (Green, et al., 2011). Results indicate that the numbers increased by 50% every second year. Subsequent captive breeding, for translocation to other islands, showed that wetapunga have a two to three-year life cycle (P. Barrett, pers. comm.), potentially explaining the stepped rate of increase on Hauturu. During the monitoring programme, occupancy of daytime refuge sites remained low, suggesting the population may increase further over time. A repeat of the programme would be required to verify the level of increase and thus give a longer-term measure of the benefit of the rat eradication.

In 2017, surveys throughout New Zealand for the endemic forest ringlet butterfly (*Dodonidia helmsii*) revealed the species' presence on Hauturu. This species was widespread throughout much of the country but is now rare or absent from many areas of its previous distribution (S. Wheatley, pers. comm.). Despite the presence of suitable habitat, the forest ringlet had not previously been recorded on the island. Multiple individuals were found, indicating a resident population (L. Wade, unpubl. data; J. Knight, pers. comm.). Gibbs (1980) highlights the potential for introduced social wasps, the German wasp (*Vespula germanica*) and European common wasp (*V. vulgaris*), as a cause for the decline in forest ringlet populations. Interestingly, the European common wasp was noted by previous island rangers as a significant nuisance on Hauturu (C. Smuts-Kennedy, pers. comm.) but subsequent to the rat eradication social wasps have not been reported. The relationship between rat eradication and social wasp populations is currently the subject of a PhD study at the University of Auckland (J. Schmack, pers. comm.).

Plants

Nineteen of 34 plant species monitored on fixed plots had more than 20 seedlings and were analysed further (Campbell, 2011). Significantly more seedlings were found for 14 species following rat eradication, *Pisonia brunoniana*, *Coprosma macrocarpa*, *Ixerba brexioides*, *Knightia excelsa*, *Rhopalostylis sapida*, *Phyllocladus trichomanoides*, *Nestegis lanceolata*, *Dacrycarpus dacrydioides*, *Ripogonum scandens*, *Hedycarya arborea*, *Dysoxylum spectabile*, *Pittosporum umbellatum*, *Macropiper excelsum* and *Corynocarpus laevigatus* (Fig. 5). Seedlings of 11 others were searched for in 2008 and 2009. In 2009 *Coprosma arborea* seedlings were very abundant. Fewer seedlings were counted of *Agathis australis*, *Beilschmiedia tarairi*, *B. tawa*, *Prumnopitys ferruginea* and *Vitex lucens*. A few species that Pacific rats severely affect (e.g. *Coprosma repens*, *Elaeocarpus dentatus*, *Melicytus novae-zelandiae*, *Pouteria costata*), showed little early response because of their initial rarity (Campbell & Atkinson 2002; Campbell, 2011).

Prior to rat eradication, seedlings of *N. lanceolata*, *R. sapida* and *R. scandens* were rare, but in 2008, *N. lanceolata* was found on most plots, and *R. sapida* and *R. scandens* seedlings were common in moister sites. The number of seedlings of other tree species had also significantly increased. Seedlings of *B. tarairi*, *C. laevigatus* and *P. trichomanoides* were twice as numerous in *Kunzea ericoides* stands after rat eradication, *D. spectabile* was five times more common and *R. scandens* 41 times.

Threatened plants also showed a positive response to the removal of rats. Improved seed set by the endangered *Carmichaelia williamsii* was noted and the endangered *Euphorbia glauca* colonised new areas. Seed production

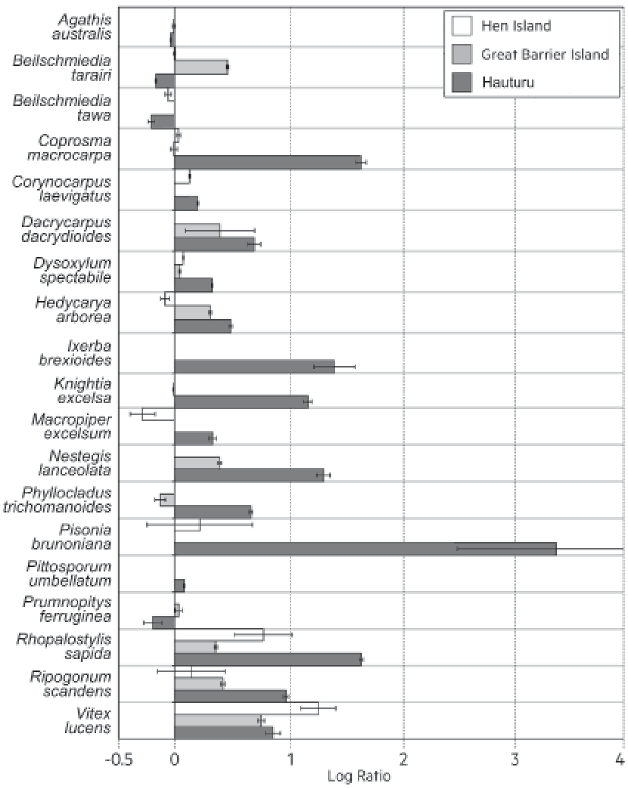

Fig. 5 Log ratios of seedling numbers on Hauturu (after eradication) and control islands with rats, with 95% confidence intervals. Log ratios of < 0, 0, > 0 indicate seedling counts that decreased, remained the same or increased, respectively.

in the endangered *Dactylanthus taylorii* increased and individuals have been discovered at new locations on the island including a site where seeds were hand sown (D. Havell, pers. comm.). No increase in seedling recruitment was noted for the invasive plant species *Asparagus scandens, Cortaderia jubata*, *C. selloana* and *Araujia hortorum* (managed as part of an ongoing eradication/ control programme) following rat and cat eradication.

DISCUSSION

The legislation that defines the management of Nature Reserves such as Hauturu, the Reserves Act 1977, mandates the removal of exotic species to protect native ecosystems. Cat and rat eradication on Hauturu became a matter of urgency because of their impacts on individual species. For cats, these impacts included extirpation of tieke and grey faced petrels as well as threats to other seabird species. The effects of rats were much wider and included impacts on seabirds, tuatara, lizards, invertebrates and an array of plants (Griffiths, 2002). Cats, rats or the two in combination were probably also responsible for the extinction of the last population of North Island snipe (Tennyson & Martinson, 2006).

Short term negative impacts from the cat eradication operation on Hauturu were minor and limited to the mortality of some non-target bird species caught in traps, apparent declines in skink populations and the removal of relatively small amounts of vegetation as a consequence of track and hut construction. An unanticipated outcome of the cat eradication, was a reduction in breeding success of Cook's petrel nesting at higher elevations. This was attributed to mesopredator release leading to increased predation pressure by a rat population no longer suppressed by cats (Rayner, et al., 2007b). This mechanism may also explain why pitfall trapping charted a decline in skink capture rates between cat and rat removal. Increased pressure on other rat foods (invertebrates, seeds and seedlings) may also have been sustained, but was not monitored.

As predicted in the AEE for rat eradication (Griffiths, 2002), negative impacts of the application of rodent bait were short-lived and minor and included no more than the loss of some individuals of at least eight bird species. Monitoring could not detect changes in abundance for these species indicating that they were not affected at the population level. The only native species not detected subsequent to the rat eradication was the red-finned bully. The absence of this species in the survey completed 10 years after the rat eradication could have been a consequence of the application of rodent bait for rats but equally the species could have been extirpated by a storm event or simply that insufficient search effort has been undertaken. This species is diadromous and likely to recolonise or could be reintroduced provided that suitable habitat on Hauturu is still available.

In contrast, the benefits of cat and rat eradication have been significant, and all species deemed vulnerable to extinction have since recovered. Tieke and kokako were successfully established following cat removal. Rat eradication resulted in recolonisation of the island by grey faced petrels and immediate recovery of the island's Cook's petrel population. As predicted by Griffiths (2002), there were increases in the abundance of skinks, geckos and invertebrates such as the wetapunga and in seedling recruitment by numerous tree species. All but one of the species identified as likely to benefit from rat eradication have shown evidence of recovery. The exception is the reintroduced tieke which could have reached carrying capacity ahead of the rat eradication.

Some species previously not recorded from Hauturu have made remarkable appearances. Examples include the New Zealand storm petrel, forest ringlet butterfly and eight new aquatic invertebrates. All such species were likely present in refugia but undetectable until rats were removed. It is unlikely that these will be the last discoveries to be made on the island. Several seabird species are expected to recolonise and highly cryptic species are still likely waiting discovery or rediscovery on Hauturu in the future. For example, only one record of striped skink has been made on the island, but this cryptic species will likely be found again in the future. As noteworthy as unexpected appearances is the disappearance of German and common wasps following rat eradication. The disappearance of these two highly invasive species may have enabled the recovery of the forest ringlet.

Other ecological changes post rat and cat eradication have been unclear. Predicted increases in the abundance of forest birds after cat and rat removal have yet to eventuate. Some species undoubtedly benefited following the removal of cats, but monitoring methods were insufficiently sensitive to detect population changes; bird counts were too variable to discern significant trends (Girardet, et al., 2001). It is also likely that some of the potential benefits of cat removal were confounded by the presence of rats. Attempts to manage kakapo on the island post cat eradication, for example, were thwarted by the continued presence of rats that preyed upon eggs and chicks.

After the removal of rats, initial positive trends for species such as hihi and tui were followed by declines to pre-eradication levels. Whether, this was a result of insufficient sampling effort, inadequacy of the methods used, changes in inter-specific competition among birds, or simply that the removal of cats and rats had little effect is not well understood. Forest birds were also monitored in the most accessible part of the island, the island's SW corner, which had been subject to the greatest impacts of logging and grazing. Consequently, ongoing forest successional changes may confound monitoring results. The most recent set of bird counts completed in 2017 suggest that some forest bird species have increased in abundance whereas others have declined (Veitch unpubl. data). Further monitoring is needed to confirm these trends. The effects of cat and rat removal on black petrel is also less straightforward. Although petrel numbers appear stable and breeding success has improved, the influence of other factors such as birds fledged from Hauturu being lured away to the much larger and noisier colony on Great Barrier Island (E. Bell, unpubl. data) may be affecting population recruitment.

As evidenced by altered patterns of seedling recruitment following rat eradication, changes in forest composition will occur. Future changes within the island's plant, invertebrate and reptile communities are likely to be strongly influenced by the recovery of Cook's petrel and the return of other seabirds. The enormous influence of seabirds on ecological communities has been well described (e.g. Jones, 2010; Smith, et al., 2011). However, given the large size of Hauturu and the extent of its forest communities, the full impact of these 'ecosystem engineers' is at present unknown.

On a global scale, there are no comparative invasive mammal eradications that have been completed in such a complex environment. The value of these eradications thus derives not only from the responses of resident species and recolonisation of those lost, but also in increasing our understanding of the ways invasive species influence island community structure. The changes reported here have been documented over only 13 years, but the removal of cats and rats has instigated a process of recovery that for many species on Hauturu will not be realised for decades. For those species reduced to relict populations and with low reproductive output, the post-eradication response will be slow. Other species such as snipe and some large species of lizards have been lost and will not contribute to Hauturu's ecosystems without intervention to re-establish them. The timescales involved for the recovery process are daunting. For example, the release of the near extinct tuatara population has begun a process of recovery for this species that may require centuries to play out.

From a social perspective, the islands of Auckland's Hauraki Gulf have been a source of inspiration for many members of the public. The removal of rats and subsequent reforestation of Tiritiri Matangi Island inspired volunteers to invest tens of thousands of hours to plant trees (Galbraith & Cooper, 2013). Hauturu has been no exception with its own Community Trust formed in 1997. The Little Barrier Island/Hauturu Supporters Trust provides on average $100,000 NZD annually to conservation programmes on the island. Visitors to Hauturu often return awe struck (R. Griffiths, pers. obs.) from their first view of 'primeval' New Zealand.

It is now important that the story of Hauturu, the impact that invasive species had on the island, and the recovery witnessed subsequent to cat and rat removal is shared on the world stage. It is only through the telling of such stories that the public's imagination will be captured along with the attention of government and private agencies. This and other projects have served to cement a sense of pride in New Zealand's biodiversity and have culminated in a pledge by New Zealand's Government for the country to be free of introduced predators by 2050 (Parkes, et al., 2017).

ACKNOWLEDGEMENTS

We would like to dedicate this paper to the many organisations and individuals that contributed to the protection of Hauturu. The legacy that stands on the horizon today serves as a fitting tribute to your efforts. Thank you also to the many individuals that contributed to the development of this paper, in particular David Wilson, David Havell and Alicia Warren.

REFERENCES

Anon. (1996). 'Kakapo recovery plan 1996–2005'. *Threatened Species Recovery Plan No. 21.* Wellington, NZ: Department of Conservation.

Bell, E.A., Mischler, C.P., MacArthur, N. and Sim, J.L. (2016). *Black Petrel (*Procellaria parkinsoni*) Population Study on Hauturu-o-Toi/ Little Barrier Island, 2015/16. Report to the Conservation Services Programme.* Wellington, NZ: Department of Conservation.

Brown, N. (2013). *Little Barrier Island Annual Reptile Monitoring Programme Jan/Feb 2013 Report.* Warkworth, NZ: Department of Conservation.

Brunton, D.H., Evans, B.A. and Ji, W. (2008). 'Assessing natural dispersal of New Zealand bellbirds using song type and song playbacks'. *New Zealand Journal of Ecology*: 147–154.

Campbell, D. and Atkinson, I. (2002). 'Depression of tree recruitment by the Pacific rat (*Rattus exulans* Peale) on New Zealand's northern offshore islands'. *Biological Conservation* 107: 19–35.

Campbell, J. (2011). 'Seedling recovery on Hauturu/Little Barrier Island, after eradication of Pacific rats *Rattus exulans*'. *DOC Research & Development Series* 325.

Capizzi, D., Sposimo, P., Sozio, G., Gotti, C., Baccetti, N. and Patrassi, F. (2019). 'Black rat eradication on Italian Islands: planning by looking backward'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 15–20. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Cree, A. (2014). *Tuatara Biology and Conservation of a Venerable Survivor*. Christchurch, NZ: Canterbury University Press.

Daniel, M. and Williams, G. (1984). 'A survey of the distribution, seasonal activity and roost sites of New Zealand bats'. *New Zealand Journal of Ecology* 7: 9–25.

DIISE (2017). 'The Database of Island Invasive Species Eradications, developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group'. University of Auckland and Landcare Research New Zealand. <http://diise.islandconservation.org>. Accessed 30 September 2017.

Eckroyd, C.E. (1995). '*Dactylanthus taylorii* recovery plan'. *Threatened Species Recovery Plan 16.* Wellington, NZ: Department of Conservation.

Elliott, G.P., Merton, D.V. and Jansen, P.W. (2001). 'Intensive management of a critically endangered species: the kakapo'. *Biological Conservation* 99(1): 121–133.

Fisher, P., Griffiths, R., Speedy, C. and Broome, K. (2011). 'Environmental monitoring for brodifacoum residues after aerial application of baits for rodent eradication. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 300–304. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Galbraith, M. and Cooper, H. (2013). 'Tiritiri Matangi – an overview of 25 years of ecological restoration'. *New Zealand Journal of Ecology* 37(3): 258−260.

Gaskin, C. and Rayner, M.J. (2013). *Seabirds of the Hauraki Gulf: Natural History, Research and Conservation*. Auckland, NZ: Auckland Council.

Gaze, P.D. (2001). 'Tuatara Recovery Plan 2001−2011'. *Threatened Species Recovery Plan 47.* Wellington, NZ: Department of Conservation.

Gibbs, G.W. (1980). *New Zealand Butterflies: Identification and Natural History*. Auckland, NZ: Collins.

Girardet, S., Veitch, C. and Craig, J. (2001). 'Bird and rat numbers on Little Barrier Island, New Zealand, over the period of cat eradication 1976–80'. *New Zealand Journal of Zoology* 28(1): 13–29.

Green, C.J., Gibbs, G.W., Barrett, P.A. and City, M. (2011). 'Wetapunga (*Deinacrida heteracantha*) population changes following Pacific rat (*Rattus exulans*) eradication on Little Barrier Island'. In: C.R. Veitch, M.N Clout and D.R. Towns, (eds.). *Island invasives: eradication and management*, pp. 305–308. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Greene, T.C. and Dilks, P.J. (2004). *Effects of a Non-toxic Bait Application on Birds: Assessing the Impacts of a Proposed Kiore Eradication Programme on Little Barrier Island*. Wellington, NZ: Department of Conservation.

Griffiths, R. (2002). *Assessment of Environmental Effects for the Eradication of Kiore (Rattus exulans) from Little Barrier Island (Hauturu).* Warkworth, NZ: Department of Conservation.

Griffiths, R. (2004). *Little Barrier Island Kiore Eradication – Post Operational Report*. Warkworth, NZ: Department of Conservation.

Griffiths, R. (2006). *Hauturu Kiore Eradication – Post Eradication Target Species Monitoring and Assessment of Operational Success*. Warkworth, NZ: Department of Conservation.

Griffiths, R., Buchanan, F., Broome, K. and Butland, B. (2012). 'Rangitoto and Motutapu – a Starting Point for Future Vertebrate Pest Eradications on Inhabited Islands'. In: Timms R.M. (ed) *Proceedings of the 25th Vertebrate Pest Conference*, pp. 22–27. Monterey, CA, USA.

Hoare, J.M. (2009). *Hauturu (Little Barrier Island) Reptile Survey – 23 February to 3 March 2009.* Warkworth, NZ: Department of Conservation.

Hoare, J.M., Pledger, S., Nelson, N.J. and Daugherty, C.H. (2007). 'Avoiding aliens: behavioural plasticity in habitat use enables large, nocturnal geckos to survive Pacific rat invasions'. *Biological Conservation* 136(4): 510–519.

Hoosen, S. and Jamieson, I.G. (2003). 'The distribution and current status of New Zealand saddleback *Philesturnus carunculatus*'. *Bird Conservation International* 13: 79–95.

Howald, G., Donlan, C.J., Faulkner, K.R., Ortega, S., Gellerman, H., Croll, D. and Tershy, B. (2010). 'Eradication of black rats *Rattus rattus* from Anacapa Island'. *Oryx* 44(1): 30–40.

Hutton, W. (1868). 'Notes on the birds of Little Barrier Island'. *Transactions of the Royal New Zealand Institute* 1: 160–162.

Imber, M.J. (1987). 'Breeding ecology and conservation of the black petrel (*Procellaria parkinsoni*)'. *Notornis* 34(1): 19–39.

Imber, M., West, J. and Cooper, W. (2003a). 'Breeding biology of Cook's petrel (*Pterodroma cookii*) on Hauturu (Little Barrier Island) and Whenua Hou (Codfish Island)'. *Notornis* 50: 221–230.

Imber, M.J., McFadden, I., Bell, E.A. and Scofield, R.P. (2003b). 'Post-fledging migration, age of first return and recruitment, and results of inter-colony translocation of black petrels (*Procellaria parkinsoni*)'. *Notornis* 50(4): 183–190.

Innes, J. and Flux, I. (1999). 'North Island kokako recovery plan 1999–2009'. *Threatened Species Recovery Plan 30.* Wellington, NZ: Department of Conservation.

Ismar, S.M.H., Gaskin, C.P., Fitzgerald, N.B., Taylor, G.A., Tennyson, A.J.D. and Rayner, M.J. (2015). 'Evaluating on-land capture methods for monitoring a recently rediscovered seabird, the New Zealand storm-petrel *Fregetta maoriana*'. *Marine Ornithology* 43: 255–258.

Jones, H.P. (2010). 'Seabird islands take mere decades to recover following rat eradication'. *Ecological Applications* 20(8): 2075–2080.

Jones, H.P., Holmes, N.D., Butchart, S.H., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E. and Burbidge, A.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences* 113(15): 4033–4038.

Lindsay, J. and Moore, P. (1995). 'Geological features of Little Barrier Island, Hauraki Gulf'. *Tane* 35: 25–38.

Lloyd, B.D. and Powlesland, R.G. (1994). 'The decline of kakapo *Strigops habroptilus* and attempts at conservation by translocation'. *Biological Conservation* 69: 75–85.

McGlynn, M., McDowall, B., Roberts, C., Petrov, I. and Marsh, S. (2000). *Vertebrate Stream Survey of Hauturu*. Unpublished data. Department of Conservation & National Institute for Water and Atmospheric Research.

Moore, J.A., Nelson, N.J., Keall, S.N. and Daugherty, C.H. (2008). 'Implications of social dominance and multiple paternity for the genetic diversity of a captive-bred reptile population (tuatara)'. *Conservation Genetics* 9(5): 1243–1251.

Parkes, J.P., Byrom, A.E. and Edge, K.-A. (2017). 'Eradicating mammals on New Zealand island reserves: what is left to do?'. *New Zealand Journal of Ecology* 41(2): 263–270.

Rayner, M.J., Clout, M.N., Stamp, R.K., Imber, M.J., Brunton, D.H. and Hauber, M.E. (2007a). 'Predictive habitat modelling improves the population census accuracy of a burrowing seabird: a study of the endangered Cook's petrel'. *Biological Conservation* 138: 235–247.

Rayner, M.J., Hauber, M.E., Imber, M.J., Stamp, R.K. and Clout, M.N. (2007b). 'Spatial heterogeneity of mesopredator release within an oceanic island system'. *Proceedings of the National Academy of Sciences of the United States of America* 104(52): 20862–20865.

Rayner, M.J., Dunphy, B.J. and Landers, T.J. (2009). 'Grey-faced petrel (*Pterodroma macroptera gouldi*) breeding on Little Barrier Island, New Zealand'. *Notornis* 56: 222–223.

Rayner, M.J., Gaskin, C.P., Fitzgerald, N.B., Baird, K.A., Berg, M.M., Boyle, D., Joyce, L., Landers, T.J., Loh, G.G. and Maturin, S. (2015). 'Using miniaturized radiotelemetry to discover the breeding grounds of the endangered New Zealand storm petrel *Fregetta maoriana*'. *Ibis* 157(4): 754–766.

Robertson, H.A., Colbourne, R.M. and Nieuland, F. (1993). 'Survival of little spotted kiwi and other forest birds exposed to brodifacoum rat poison on Red Mercury Island'. *Notornis* (40): 253−262.

Sherley, G. (1998). 'Threatened weta recovery plan'. *Threatened Species Recovery Plan No. 25.* Wellington, NZ: Department of Conservation.

Sibson, R. (1947). 'A visit to Little Barrier Island'. *New Zealand Birds Notes* 2: 134–144.

Smith, J., Mulder, C. and Ellis, J. (2011). 'Seabirds as Ecosystem Engineers: Nutrient Inputs and Physical Disturbance'. In: C.P.H. Mulder, W.B. Anderson, D.R. Towns, and P.J. Bellingham (eds.) *Seabird Islands. Ecology, Invasion and Restoration*, pp. 27–55. Oxford. New York: Oxford University Press.

Stephenson, B.M., Gaskin, C.P., Griffiths, R., Jamieson, H., Baird, K.A., Palma, R.L. and Imber, M.J. (2008). 'The New Zealand storm-petrel (*Pealeornis maoriana* Mathews, 1932): First live capture and species assessment of an enigmatic seabird'. *Notornis* 55(4): 191–206.

Tennyson, A.J.D. and Martinson, P. (2006). *Extinct Birds of New Zealand*. Wellington, NZ: Te Papa Press.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (1990). 'The Potential for Ecological Restoration in the Mercury Islands'. In*:* D.R. Towns, C.H. Daugherty and I.A.E. Atkinson (eds.) *Ecological restoration of New Zealand Islands. Conservation Sciences Publication No. 2:* 91–108.

Towns, D.R. (1991). 'Response of lizard assemblages in the Mercury Islands, New Zealand, to removal of an introduced rodent: the kiore (*Rattus exulans*)'. *Journal of the Royal Society of New Zealand* 21: 119–136.

Towns, D.R. (1996). 'Changes in habitat use by lizards on a New Zealand island following removal of the introduced Pacific rat *Rattus exulans*'. *Pacific Conservation Biology* (2): 286–292.

Towns, D.R., Parrish, R. and Board, N.W.T. (2003). *Restoration of the Principal Marotere Islands*. Wellington, NZ: Department of Conservation.

Towns, D.R. and Atkinson, I.A.E. (2004). *Restoration Plan for Korapuki Island (Mercury Islands) 2004–2024*. Wellington, NZ: Department of Conservation.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?'. *Biological Invasions* 8: 863–891.

Towns, D., Parrish, G., Tyrell, C., Ussher, G., Cree, A., Newman, D., Whitaker, A. and Westbrooke, I. (2007). 'Responses of tuatara (*Sphenodon punctatus*) to removal of introduced Pacific rats from islands'. *Conservation Biology* 21(4): 1021–1031.

Toy, R., Greene, T.C., Greene, B.S., Warren, A. and Griffiths, R. (2018). 'Changes in density of hihi (*Notiomystis cincta*), tieke (*Philesturnus carunculatus*) and tui (*Prosthemadera novaeseelandiae*) on Little Barrier Island (Te Hauturu-o-Toi), Hauraki Gulf, Auckland'. *New Zealand Journal of Ecology* 42(2): 149-157

Turbott, E. (1947). 'Birds of Little Barrier Island'. *New Zealand Bird Notes* 2(92): 93–108.

Veitch, C.R. (2001). 'The eradication of feral cats (*Felis catus*) from Little Barrier Island, New Zealand'. *New Zealand Journal of Zoology* 28(1): 1–12.

Wade, L. (2009). *Hauturu (Little Barrier Island) Kiwi Survey July 2009*. Warkworth, NZ: Department of Conservation.

Wade, L. (2014a). *Hauturu (Little Barrier Island) Comparative Kiwi Survey May and July 2014*. Warkworth, NZ: Department of Conservation.

Wade, L. (2014b). *A Survey of Aquatic Invertebrates and Fish in a Selection of Intermittent Streams on Hauturu/Little Barrier Island*. Unpublished report. Whangarei, NZ: NorthTec, Applied and Environmental Sciences.

Whitaker, A.H. and Daugherty, C.H. (1991). *Research on the Tuatara (*Sphenodon punctatus*) of Little Barrier Island, 5–12 February 1991*. Unpublished report. Auckland, NZ: Auckland Conservancy, New Zealand Department of Conservation.

Winterbourn, M.J. (1964). 'A survey of the stream fauna of Little Barrier Island'. *Tane* 10: 58–69.

Young, D. (2004). *Our Islands, Ourselves*. Dunedin, NZ: University of Otago Press.

Restoring plant-pollinator communities: using a network approach to monitor pollination function

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Abstract Ecological restoration is a common tool to mitigate the loss of species and habitats, ultimately aiming to restore ecosystem functioning. Large-scale experimental evidence is lacking, however, on whether standard management techniques, e.g. the removal of invasive alien plants, indeed restore ecosystem functions at the community level. One key ecosystem function is animal mediated pollination. Based on findings from an experimental network study on rocky outcrops (inselbergs) on the island of Mahé in the Seychelles, I present recommendations for conservation practitioners about how to incorporate a network approach into an evaluation of management effectiveness. Responses to restoration actions by plant-pollinator communities and pollination functions lead to several conclusions regarding the resilience of native fauna and flora and ecosystem functioning. Pollination network structure appears to be directly related to the quality and resilience of pollination services, which suggests that network analysis can be used to monitor management efficacy. I provide recommendations and advice to encourage the uptake of a network approach by conservation practitioners seeking to restore ecosystem functions.

Keywords: biomonitoring, management effectiveness, pollination networks, Seychelles, vegetation restoration

INTRODUCTION

Despite recent efforts to slow biodiversity decline worldwide, habitat degradation continues to degrade and simplify ecosystems, especially in the species-rich tropics (Butchart, et al., 2010). To mitigate the effects of habitat modification on ecosystems and assist species and ecosystem functions to adapt to changing environmental conditions, conservation practitioners employ a diverse set of management tools, including ecological restoration (Sodhi & Ehrlich, 2010). Such management tools often rely on a few well-studied target species to assess their outcomes, primarily because of limited time and resources. However, too little is known about the efficacy of restoration for achieving self-sustaining species communities and functioning ecosystems. Habitat restoration usually modifies ecosystems with the purpose of providing suitable habitat for target native species (Miller & Hobbs, 2007). Non-target species, however, can serve essential functional roles in the restored habitat and failure to recognise these species and the ecosystem-level interactions and processes that they are involved in may compromise restoration efforts and assessment (Ehrenfeld, 2000). Pollination is one such key ecosystem function; most tropical plants and crops heavily rely on pollination services for reproduction (Klein, et al., 2007; Ollerton, et al., 2011). Pollinators are rarely targets of habitat restoration (Williams, 2011), although this is slowly changing in agricultural areas where the benefits of wild bees in crop pollination have been considered (Kremen & M'Gonigle, 2015). Given that ecosystems are characterised by networks of interactions between organisms (McCann, 2007), the effect of habitat restoration on pollination interactions is often best studied with a network approach (Jordano, 1987; Proulx, et al., 2005). Thus, to assess the impact of habitat restoration on integrity of pollination services, an understanding of the implications of structural changes of pollination networks on functional performance is critical. Recent work proposed close links between network structure and ecosystem functioning (Coux, et al., 2016; Gómez, et al., 2011; Schleuning, et al., 2015), but field experiments at the community level are required to shed light on the relationships between habitat restoration, pollination network structure, the resilience of plant-pollinator communities, and the quality of pollination services.

Restoration practitioners worldwide place vegetation rehabilitation at the centre of habitat restoration, which often involves removal of exotic plants and assisted recovery of native plant communities (Clewell & Aronson, 2013). Assistance takes the form of fencing off native habitat against large herbivores or exotic seed predators (see e.g. Florens & Baider, 2013), or the reintroduction of large herbivores to replace now extinct seed dispersers (Hansen, 2015). These interventions enable native vegetation and their mutualists to establish and adapt to subtle changes in native and novel processes, which increase resilience against future disturbance. One important prerequisite for a self-sustaining restored plant community is a large and diverse native fruit crop, which is dependent to some degree on the quality and quantity of pollination services. To provide optimal functional performance, plant-pollinator communities mutually rely on diverse and reliable resources (pollen and nectar) and services (pollination). Weighted network metrics, which take into account the quantitative importance of species for their mutualistic partners, have been developed to assess the consequences of vegetation rehabilitation on pollination services by teasing apart changes in abundance, species diversity and the topology of species interactions, e.g. species generalisations (Banašek-Richter, et al., 2004; Blüthgen, et al., 2006; Tylianakis, et al., 2007).

RESTORING PLANT-POLLINATOR COMMUNITIES

In a recent study, Kaiser-Bunbury, et al. (2017) showed for the first time, with a large-scale field experiment, that not only were species communities fundamentally changed by restoration (the removal of invasive alien shrubs), but also plant-pollinator interactions became more resilient as a result of restoration. Restoration altered pollinator behaviour and increased pollinator species richness (Kaiser-Bunbury, et al., 2017). In this instance, the removal of invasive plants modified pollinator foraging patterns, which increased pollinator efficiency (i.e. more pollen delivered per visit) and frequency (i.e. higher visitation rate per flower) of native plants in the restored community (see Fig. 3 in Kaiser-Bunbury, et al., 2017). Simultaneously,

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pollinator species became more generalised in restored communities, creating greater functional redundancy and lower mutual dependencies. These results appeared at first contradictory, as specialised pollinators tend to be more effective pollinators than generalists, due to lower interspecific pollen transfer (see Morales & Traveset, 2008 and references within). However, the data also suggested that while pollinator species became more generalised as a result of restoration, individual pollinators had increased floral constancy, providing high quality pollination services even at relatively low visitation frequencies (Kaiser-Bunbury, et al., 2017). Several plant species at the restored sites (nine species at restored vs. two species at unrestored sites) further benefitted from attracting more pollinator species – on average an increase in pollinator species richness by approximately 114% compared to the same plant species at the unrestored sites, thereby lowering their dependency on a few pollinator species for reproduction.

The effects of restoration on the plant-pollinator community and pollination services were reflected by changes in pollination network structure (Kaiser-Bunbury, et al., 2017). The findings on the connection between network structure and ecological processes are important for two reasons. Firstly, they corroborate previous theoretical and empirical, non-experimental work that suggested a direct relationship between network properties and ecosystem functioning (Gómez, et al., 2011; Schleuning, et al., 2015; Coux, et al., 2016). Secondly, network metrics, which are commonly used to characterise network properties, can now be employed to inform scientists and practitioners about the ecological and conservation status of communities and ecosystem functions when, for example, compared to baseline data. With future shifts in conservation approaches towards the protection of ecosystem services and functions (Harvey, et al., 2017), suitable tools and methods need to be developed that allow conservation biologists and practitioners to monitor and evaluate such processes. The Kaiser-Bunbury, et al. (2017) study provided an important cornerstone for interpreting processes in ecological communities by using a network approach.

Network ecologists have advocated for some time the potential of a network approach in applied ecology, based on advances in understanding the processes that shape community level interactions (e.g. Memmott, 2009; Kaiser-Bunbury, et al., 2010; Tylianakis, et al., 2010). More recently, a selection of network indicators, i.e. aggregate network metrics describing community properties, was proposed, which characterise the diversity and distribution of interactions at the species, guild (e.g. plants, pollinators) and network level (Kaiser-Bunbury & Blüthgen, 2015). These network indicators were selected because of ecological characteristics, sound empirical and theoretical support, conceptual similarities to well-established diversity indicators, and computational ease with which they can be generated (Kaiser-Bunbury & Blüthgen, 2015). The authors presented a conceptual framework on how to use network indicators to guide conservation decisions by evaluating management effectiveness, and proposed island ecosystems as suitable model system. Island biotas are not only in urgent need of extensive conservation action, but the simplicity of island ecosystems also facilitates comprehensive studies on interaction networks (Kaiser-Bunbury, et al., 2010). Thus, how can the insights gained from studies on network structure and ecosystem functioning (e.g., Kaiser-Bunbury, et al., 2017) be applied to biomonitoring and assessments of management effectiveness by island conservation practitioners?

IMPLICATIONS FOR ECOLOGICAL RESTORATION

Biotic interactions (here I refer to mutualistic interactions such as pollination and seed dispersal, but antagonistic i.e., trophic, interactions may equally be used) can be short-lived and highly variable across seasons, years or even longer time spans (Medan, et al., 2006; Olesen, et al., 2010; CaraDonna, et al., 2017). Network indicators that describe the ecological processes determining network structure may be most suitable to monitor ecologically meaningful changes in biotic interactions that reflect community-wide adaptations to specific restoration actions, for example, the removal of invasive species, reforestation with native plants, or landscape modifications. Methodological and ecological advances, however, are rarely used to their full potential for evaluating and monitoring conservation progress (Gardener, et al., 2010). To benefit from such advances, network indicators could be used to inform managers on whether conservation interventions actually restore or maintain ecosystem integrity (Noss, 2004). In the Seychelles, the positive effects of restoration on pollinator communities and native plant reproduction were reflected in corresponding changes in network indicators (Kaiser-Bunbury & Blüthgen, 2015). These included the total number of visits and interactions, interaction diversity and evenness, and the degree of network- (H_2') and species-level (d') specialisation (Kaiser-Bunbury, et al., 2017). Thus, recording community-wide biotic interactions and calculating network indicators for observed biotic interactions can provide restoration practitioners with a measure of effectiveness for achieving the overall goal of restoring ecosystem functioning.

A network approach may appear challenging, overly complicated and costly to most conservation practitioners. Instead of providing comprehensive instructions on how to apply a network approach in restoration, I aim to illustrate that using biotic interactions and network analyses are viable and effective tools to monitor conservation progress and adapt management approaches based on the outcome of the performance assessment. Below I outline four recommendations for consideration by practitioners who are interested in embracing a network approach in biodiversity conservation.

1) *Clearly define conservation goals that can be validated with network indicators*. Network indicators can only illustrate the properties of one specific ecosystem function at a time, for example, pollination, seed dispersal, or predation. It is therefore important to identify the ecosystem function to be targeted by the conservation intervention (Kaiser-Bunbury & Blüthgen, 2015). Decision-making tools that take into account multiple ecosystem functions may be required to prioritise conservation action (McCarthy & Possingham, 2007). Clear conservation objectives and outcomes will then provide the basis for selecting network indicators and setting threshold values of conservation targets (Kaiser-Bunbury & Blüthgen, 2015).

2) *Actively engage with applied network ecologists* who can assist with establishing data recording protocols and conducting network analysis, possibly via electronic data collection in the field and automated analysis (Kaiser-Bunbury & Blüthgen, 2015). At first, the network approach may appear dauntingly complex. However, the involvement of network ecologists in the planning phase of any conservation action will ensure that a suitable sampling protocol is developed, facilitating data analysis and interpretation to evaluate management effectiveness. Network ecologists are also more likely to follow advances in the field and can update protocols, sampling techniques and analyses based on the most up-to-date research. In return for the time invested, ecologists will have access to

empirical data for publications and contribute actively to maximising the impact of their research.

3) *Be realistic in sampling design.* Collecting data on biotic interactions involving all species in the community is often considered extremely time and labour intensive, and therefore costly. It is not necessary, however, to record 'every single interaction'. Interaction networks are inherently under-sampled (Vázquez, et al., 2009) but still provide meaningful insights into ecosystem complexity and functioning. It is more important to identify the most time and cost efficient sampling method (see e.g. Hegland, et al., 2010) and assess sampling completeness with appropriate extrapolation techniques (Colwell & Coddington, 1994). Depending on the conservation goals, sampling of subsets or at a lower frequency/density may suffice to reveal changes in network structure as a result of the restoration intervention.

4) *Select the most suitable sampling approach* for your habitat, available resources, and the accessibility of the management site. For example, pollination interactions can be observed using standardised transects, which is a time-efficient sampling method most suited to meadows, heathlands and other low-growing plant communities. Alternatively, by observing target plants for a set amount of time, pollination interactions can be recorded in a forest or shrubland habitat with a 3-dimensional structure and a patchy distribution of flowers (for a comparison of the methods see Gibson, et al., 2011).

Why should conservation practitioners and ecologists invest extra time and resources into monitoring processes? In short, moving conservation actions towards an ecosystem functions oriented approach (sensu Harvey, et al., 2017) will require tools that can monitor and evaluate the multi-facetted dimensions of biodiversity. The network approach can generate detailed insights into the functioning of ecological communities, is developing rapidly, and presents a promising and exciting method for improving biodiversity conservation in the 21st century.

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REFERENCES

Banašek-Richter, C., Cattin, M.-F. and Bersier, L.-F. (2004). 'Sampling effects and the robustness of quantitative and qualitative food-web descriptors'. *Journal of Theoretical Biology* 226(1): 23–32.

Blüthgen, N., Menzel, F. and Blüthgen, N. (2006). 'Measuring specialization in species interaction networks'. *BMC Ecology* 6: 9.

Butchart, S.H.M., Walpole, M., Collen, B., van Strien, A., Scharlemann, J.P.W., Almond, R.E.A., Baillie, J.E.M., Bomhard, B., Brown, C., Bruno, J., Carpenter, K.E., Carr, G.M., Chanson, J., Chenery, A.M., Csirke, J., Davidson, N.C., Dentener, F., Foster, M., Galli, A., Galloway, J.N., Genovesi, P., Gregory, R.D., Hockings, M., Kapos, V., Lamarque, J.F., Leverington, F., Loh, J., McGeoch, M.A., McRae, L., Minasyan, A., Morcillo, M.H., Oldfield, T.E.E., Pauly, D., Quader, S., Revenga, C., Sauer, J.R., Skolnik, B., Spear, D., Stanwell-Smith, D., Stuart, S.N., Symes, A., Tierney, M., Tyrrell, T.D., Vie, J.C. and Watson, R. (2010). 'Global biodiversity: Indicators of recent declines'. *Science* 328(5982): 1164–1168.

CaraDonna, P.J., Petry, W.K., Brennan, R.M., Cunningham, J.L., Bronstein, J.L., Waser, N.M. and Sanders, N.J. (2017). 'Interaction rewiring and the rapid turnover of plant–pollinator networks'. *Ecology Letters* 20(3): 385–394.

Clewell, A.F. and Aronson, J. (2013). *Ecological Restoration: Principles, Values, and Structure of an Emerging Profession.* Washington, D.C., USA: Island Press.

Colwell, R.K. and Coddington, J.A. (1994). 'Estimating terrestrial biodiversity through extrapolation'. *Philosophical Transactions of the Royal Society of London Series B-Biological Sciences* 345(1311): 101–118.

Coux, C., Rader, R., Bartomeus, I., Tylianakis, J.M. and Mouillot, D. (2016). 'Linking species functional roles to their network roles'. *Ecology Letters* 19(7): 762–770.

Ehrenfeld, J.G. (2000). 'Defining the limits of restoration: The need for realistic goals'. *Restoration Ecology* 8(1): 2–9.

Florens, F.B.V. and Baider, C. (2013). 'Ecological restoration in a developing island nation: How useful is the science?'. *Restoration Ecology* 21(1): 1–5.

Gardener, M.R., Atkinson, R. and Rentería, J.L. (2010). 'Eradications and people: Lessons from the plant eradication program in Galapagos'. *Restoration Ecology* 18(1): 20–29.

Gibson, R.H., Knott, B., Eberlein, T. and Memmott, J. (2011). 'Sampling method influences the structure of plant–pollinator networks'. *Oikos* 120(6): 822–831.

Gómez, J.M., Perfectti, F. and Jordano, P. (2011). 'The functional consequences of mutualistic network architecture'. *PLOS ONE* 6(1): e16143.

Hansen, D.M. (2015). 'Non-native megaherbivores: The case for novel function to manage plant invasions on islands'. *AoB Plants* 7: plv085.

Harvey, E., Gounand, I., Ward, C.L. and Altermatt, F. (2017). 'Bridging ecology and conservation: From ecological networks to ecosystem function'. *Journal of Applied Ecology* 54(2): 371–379.

Hegland, S.J., Dunne, J., Nielsen, A. and Memmott, J. (2010). 'How to monitor ecological communities cost-efficiently: The example of plant-pollinator networks'. *Biological Conservation* 143(9): 2092–2101.

Jordano, P. (1987). 'Patterns of mutualistic interactions on pollination and seed dispersal: connectance, dependence, asymmetries and coevolution'. *The American Naturalist* 129(5): 657–677.

Kaiser-Bunbury, C.N., Traveset, A. and Hansen, D.M. (2010). 'Conservation and restoration of plant-animal mutualisms on oceanic islands'. *Perspectives in Plant Ecology Evolution and Systematics* 12(2): 131–143.

Kaiser-Bunbury, C.N. and Blüthgen, N. (2015). 'Integrating network ecology with applied conservation: A synthesis and guide to implementation'. *AoB Plants* 7: plv076.

Kaiser-Bunbury, C.N., Mougal, J., Whittington, A.E., Valentin, T., Gabriel, R., Olesen, J.M. and Blüthgen, N. (2017). 'Ecosystem restoration strengthens pollination network resilience and function'. *Nature* 542(7640): 223–227.

Klein, A.-M., Vaissière, B.E., Cane, J.H., Steffan-Dewenter, I., Cunningham, S.A., Kremen, C. and Tscharntke, T. (2007). 'Importance of pollinators in changing landscapes for world crops'. *Proceedings of the Royal Society B: Biological Sciences* 274(1608): 303–313.

Kremen, C. and M'Gonigle, L.K. (2015). 'Small-scale restoration in intensive agricultural landscapes supports more specialized and less mobile pollinator species'. *Journal of Applied Ecology* 52(3): 602–610.

McCann, K. (2007). 'Protecting biostructure'. *Nature* 446(7131): 29.

McCarthy, M.A. and Possingham, H.P. (2007). 'Active adaptive management for conservation'. *Conservation Biology* 21(4): 956–963.

Medan, D., Basilio, A.M., Devoto, M., Bartoloni, N.J., Torretta, J.P. and Petanidou, T. (2006). 'Measuring Generalization and Connectance in Temperate, Year-long Active Systems'. In: N.M. Waser and J. Ollerton (eds.) *Plant-pollinator Interactions: From Specialization to Generalization*, pp. 245–259. Chicago: The University of Chicago Press.

Memmott, J. (2009). 'Food webs: A ladder for picking strawberries or a practical tool for practical problems?'. *Philosophical Transactions of the Royal Society B: Biological Sciences* 364(1524): 1693–1699.

Miller, J.R. and Hobbs, R.J. (2007). 'Habitat restoration—do we know what we're doing?'. *Restoration Ecology* 15(3): 382–390.

Morales, C. and Traveset, A. (2008). 'Interspecific pollen transfer: Magnitude, prevalence and consequences for plant fitness'. *Critical Reviews in Plant Sciences* 27: 221–238.

Noss, R.F. (2004). 'Some suggestions for keeping national wildlife refuges healthy and whole'. *Natural Resources Journal* 44: 1093–1111.

Olesen, J.M., Dupont, Y.L., O'Gorman, E.J., Ings, T.C., Layer, K., Melian, C.J., Trojelsgaard, K., Pichler, D.E., Rasmussen, C. and Woodward, G. (2010). 'From Broadstone to Zackenberg: Space, time and hierarchies in ecological networks'. *Advances in Ecological Research: Ecological Networks* 42: 1–69.

Ollerton, J., Winfree, R. and Tarrant, S. (2011). 'How many flowering plants are pollinated by animals?' *Oikos* 120(3): 321–326.

Proulx, S.R., Promislow, D.E.L. and Phillips, P.C. (2005). 'Network thinking in ecology and evolution'. *Trends in Ecology and Evolution* 20(6): 345–353.

Schleuning, M., Fründ, J. and García, D. (2015). 'Predicting ecosystem functions from biodiversity and mutualistic networks: An extension of trait-based concepts to plant–animal interactions'. *Ecography* 38(4): 380–392.

Sodhi, N.S. and Ehrlich, P.R. (2010). *Conservation Biology for All.* Oxford, New York: Oxford University Press.

Tylianakis, J.M., Tscharntke, T. and Lewis, O.T. (2007). 'Habitat modification alters the structure of tropical host-parasitoid food webs'. *Nature* 445(7124): 202–205.

Tylianakis, J.M., Laliberté, E., Nielsen, A. and Bascompte, J. (2010). 'Conservation of species interaction networks'. *Biological Conservation* 143(10): 2270–2279.

Vázquez, D.P., Blüthgen, N., Cagnolo, L. and Chacoff, N.P. (2009). 'Uniting pattern and process in plant–animal mutualistic networks: A review'. *Annals of Botany* 103(9): 1445–1457.

Williams, N.M. (2011). 'Restoration of nontarget species: Bee communities and pollination function in riparian forests'. *Restoration Ecology* 19(4): 450–459.

Ten years after feral goat eradication: the active restoration of plant communities on Guadalupe Island, Mexico

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Abstract As the first step towards the ecological restoration of its islands, Mexico has completed 60 eradications of invasive mammals thanks to a strong partnership between *Grupo de Ecología y Conservación de Islas, A.C.* (GECI), the federal government, local fishing communities, academia, and private donors. The removal of invasive mammals has led to the dramatic recovery of the islands' ecosystems. On Guadalupe Island, after completing the goat eradication in 2007, the native vegetation started to recover. Plants considered extinct or extirpated have been rediscovered, and plant species new to the island have been recorded. However, in order to achieve the island's full recovery, the active restoration of degraded soils and vegetation are needed. To date, GECI, in collaboration with the National Forestry Commission (CONAFOR) and the National Commission for Natural Protected Areas (CONANP), is implementing a 700 ha project to accelerate the restoration of the native vegetation communities. The project involves reforestation, erosion control, and fire prevention actions on different plant communities: forests and sage scrub. An on-site nursery has been established, seedlings—mostly from endemic trees—are being grown, and on-site reforestation planting has started. Up to June 2018, we have planted almost 40,000 trees, and will produce 160,000 seedlings during this year. Mechanical methods to control and prevent erosion have been used as we have installed more than 2,400 m of contour barriers, 57 m^3 of dams, and rehabilitated firebreaks. The actions will continue: the long-term goal being the comprehensive restoration of the vegetation communities devastated by feral goats. The Guadalupe Island experience will be useful to inform the restoration of other Mexican islands.

Keywords: erosion control, Guadalupe cypress, Guadalupe pine, reforestation, vegetation recovery

INTRODUCTION

Islands support a disproportionate amount of biodiversity in relation to their area (Myers, et al., 2000), but also are vulnerable ecosystems, highly susceptible to any alteration to their fragile equilibria (Holdgate, 1967; Simberloff, 1995; Whittaker & Fernández-Palacios, 2007). The trophic webs of oceanic islands have been described as "very simplified, with little ecological or taxonomical redundancy" (Courchamp, et al., 2003). This is one of the reasons why invasive alien species are considered one of the biggest threats to insular ecosystems and the main cause of insular biodiversity loss, as well as the alteration of ecosystem functions (Reaser, et al., 2007; Veitch, et al., 2011). Native insular species normally lack evolutionary defences, having traits that evolved in the absence of regular immigrants, and in consequence they fail to adapt to new threats posed by invasive species (Brook, et al., 2008; Berglund, et al., 2009). In contrast, invasive species have attributes that facilitate their establishment in novel environments due to their broad ecological niche (generalists) and high degree of behavioural flexibility. Consequently, they normally thrive when introduced to new environments (Mack, et al., 2000; Courchamp, et al., 2003; Sol, 2007). Among invasive mammals, feral goats (*Capra hircus*) are one of the most destructive species. Their effects include overgrazing, soil compaction, and tree and shrub damage through browsing (Coblentz, 1978; Parkes, et al., 1996; Campbell & Donlan, 2005; Chynoweth, et al., 2013).

Guadalupe Island is a priority site in terms of biodiversity conservation. It is a Biosphere Reserve, as well as an Important Bird Area (IBA; Vidal, et al., 2009) and an Alliance for Zero Extinction site (AZE, 2010). In addition, it is categorised as a Marine Priority Conservation Area by the Commission for Environmental Cooperation of North America (Morgan, et al., 2005), and it is included in the Southern Californian Pacific Marine Ecoregion (Wilkinson, et al., 2009). Unfortunately, invasive mammals – including feral goats – were introduced in the 19th century with devastating consequences for the island's flora and fauna. Goats depleted entire vegetation communities. Moran (1996) stated that "…it is most important before plants are lost, to remove all goats from the island, reversing the process of degradation and encouraging in every way the renewal of the natural vegetation. Even at best, some rare plants may die out unless propagated and replanted...". Other authors agreed, claiming that conservation actions for the island must begin by removing the feral goats and be followed with a plan of active restoration (León de la Luz, et al., 2003, Aguirre-Muñoz, et al., 2005). Also, for the Mexican Government there was an understanding that urgent restoration actions were needed. As a result, in terms of ecological restoration, much has been done during the past decade to tackle the threats posed to Guadalupe's biodiversity, particularly those from introduced species (Aguirre-Muñoz, et al., 2011; Luna-Mendoza, et al., 2007). Therefore, a long-term restoration and conservation programme has been developed for the island, aimed at removing invasive mammals to protect Guadalupe's native flora and fauna – especially seabirds – and preventing more extinctions. The successful eradication of goats, in a collaboration between GECI, federal government agencies and private donors, was the beginning of the island's recovery. The next phase is to do active restoration, mostly through reforestation of several vegetation communities.

SITE DESCRIPTION

Guadalupe Island is a 242 km^2 remote oceanic island located in the Pacific Ocean, 260 km off the Baja California peninsula, Mexico (29° N, 118° 20'W). It represents Mexico's last frontier on its western and northern margins; a unique territory in many ways, particularly in terms of biodiversity, a "naturalists' paradise" in the words of Dr Edward Palmer after his 1875 visit to Guadalupe (Huey, 1925).

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Guadalupe is a 5.8 km high seamount that emerges from a depth of 4.5 km, with a maximum elevation of 1.3 km above sea level (Delgado-Argote, et al., 1993). It comprises a main island, three islets and several offshore rocks. Guadalupe was discovered in 1602 by Spanish explorer Juan Sebastían Vizcaíno (León Portilla, 1989). Yet, it remained pristine and uninhabited until the beginning of the 19th century when Russian, English and American fur hunters visited the island in search of fur seals, sea otters and elephant seals (Hanna, 1925; Huey, 1925).

Guadalupe is a protected area decreed as a Biosphere Reserve by the Mexican government in 2005. The Reserve is managed by the National Commission of Natural Protected Areas (CONANP), and is safeguarded by the Ministry of the Navy, which has been watching over this important territory since the early 1900s. Besides the Navy base on the southern tip of the island, there are two more settlements: a settlement of the *Abuloneros y Langosteros* fishing cooperative on the west coast, and a biological field station of the Mexican NGO *Grupo de Ecología y Conservación de Islas, A.C.* (GECI, for its Spanish acronym) at about 1,200 m above sea level on the north-west portion of the island. Also, CONANP personnel are present permanently on site. A total of 100 people inhabit Guadalupe (CONANP, 2013). The only economic activity on the island is commercial fishing, carried out solely by the fishing cooperative to sustainably harvest valuable marine resources such as abalone (*Haliotis* spp.) and lobster (*Panulirus interruptus*) (Searcy-Bernal, et al., 2010; Méndez-Sánchez, 2012).

Climate

Guadalupe has a Mediterranean climate, characterised by hot, dry summers and cool, wet winters (Camps & Ramos, 2012; Granda, et al., 2014). Temperature is relatively stable throughout the year, with a mean of 17.2 ± 2 ºC. Relative humidity oscillates between 69 ± 8% to 82 ± 5% without a well-defined seasonal pattern (Castro, et al., 2005) and average annual cumulative precipitation is 193 ± 119 mm (CONAGUA-SARH Delgadillo in Moran, 1996; SARH-Colegio de Postgraduados, 2010). However, given the islands' complex topography, some microclimates are also recognised. The south end of Guadalupe is drier compared to the rest of the island, and the humidity increases northwards with elevation, mostly due to the fog influence. There are some records of ice and snow in winter, restricted to the cypress forest at the higher elevations (Moran, 1996; N. Silva-Estudillo, pers. comm.). It is also likely that rainfall is relatively more abundant at higher elevations (Moran, 1996). Guadalupe Island's local climate can also be influenced by regional climatic conditions. Occasional tropical storms from the south bring heavy rainfall to the island between summer and autumn (August to October) (Moran, 1996). In addition, the normal precipitation pattern can be disrupted by irregular El Niño or La Niña events, associated with supra- and subnormal precipitation, respectively, between December and March (winter and spring) (Minnich, et al., 2009). This oceanic island is heavily influenced by the California Current, which generates a peculiar pattern of wind, fog, and rainfall (León de la Luz, et al., 2003; Garcillán, et al., 2012). Winds prevail from the north-west, while the island's climate is influenced by a near-permanent fog system which allows the presence of forests on the island despite the low precipitation.

Flora

In a classification of Mexican Biogeographic Provinces, Guadalupe Island is considered as a separate province (i.e. Guadalupe Island province) within the Baja California Province, which is part of the Nearctic Region (Morrone, et al., 2002). This classification is based on distributional patterns of plants, invertebrates and birds (Morrone, et al., 1999). Floristically, it is very similar to the Channel Islands, USA (Raven, 1965). Originally, the island was home to a rich flora that included several insular and Guadalupe endemics. In total, 225 vascular plant species have been recorded on the island, 7% insular endemic (shared with other islands of the region) and 12% endemic to Guadalupe (Junak, et al., 2005; Rebman, et al., 2005; GECI, unpublished data). In addition, 36 non-vascular plants have been recorded for the site (Crum & Miller, 1956; Crum, 1972 in Moran, 1996) as well as 104 lichen species (Weber, 1994 in Moran, 1996).

Several original vegetation communities have been described, based on historic records (Moran, 1996;León de la Luz, et al., 2005; Oberbauer, 2005). The communities include forests, woodlands, chaparral (shrubs), native grassland and communities dominated by low shrubs. Some of the representative species of these vegetation communities are the endemic Guadalupe cypress (*Cupressus guadalupensis*), Monterey pine (*Pinus radiata* var. *binata*) and Guadalupe palm (*Brahea edulis*). Several native species such as the juniper (*Juniperus californica* - now restricted to <10 individuals), and the shrubs island redberry (*Rhamnus pirifolia*), and laurel sumac (*Malosma laurina*) were also characteristic of some communities, along with endemic succulents such as cistanthe (*Cistanthe guadalupensis*) and liveforever (*Dudleya guadalupensis*), the endemic Guadalupe senecio (*Senecio pameri*); and three endemic species of shrubby tarweeds (*Deinandra* spp.). Some insular endemics were also representative of these environments, such as the island hazardia (*Hazardia cana*; only present on San Clemente Island, USA and Guadalupe) and the insular oak (*Quercus tomentella*; only present on five of the Channel Islands, USA and Guadalupe).

INTRODUCTION OF INVASIVE MAMMALS

Guadalupe Island remained pristine and uninhabited until the beginning of the 19th century when fur hunters arrived (Hanna, 1925; Huey, 1925). It is likely that house mice (*Mus musculus*) and feral cats (*Felis catus*) were introduced following these first human settlements (Hanna, 1925; Huey, 1925; Moran, 1996). Both species were introduced around 1880, rapidly establishing feral populations (Moran, 1996). In addition, whalers, in order to have a source of fresh meat during their voyages, introduced goats (Moran, 1996). Together, goats and cats have been responsible for the extinction and extirpation of many native and endemic species (e.g. Jehl Jr & Everett, 1985; León de la Luz, et al., 2003), and the impacts of house mice remain to be evaluated. In addition, feral dogs (*Canis familiaris*) also established a population on the island (Moran, 1996). Other mammals, such as cows (*Bos taurus*), were also introduced but never established feral populations (Aguirre-Muñoz, et al., 2011; J. Rico-Cerda pers. comm.).

Effects of feral goats on native flora

After goats were introduced, only one of the original vegetation communities, comprising low shrubs present on islets where goats or mice never were introduced, remained pristine. The other plant communities either disappeared, became restricted to a very patchy distribution, or were represented only by isolated individual plants. At least 26 plant taxa became extinct or were extirpated due to feral goats (Moran, 1996; León de la Luz, et al., 2003; Oberbauer, 2005; GECI, unpublished data). Not only were entire vegetation communities depleted, and many endemic species lost, but also many non-native species have been introduced. Since 1875, at least 69 plant species have been introduced to the island, mostly European grasses and forbs (Junak, et al., 2005; Rebman, et al., 2005; GECI, unpublished data). The heavy modification of the ecosystem caused by the feral goats, in combination with the arrival of invasive plants, resulted in a vast extension of bare ground and vegetation dominated by European grasses (grassland community), such as slender wild oat (*Avena barbata*) and red brome (*Bromus rubens*).

CURRENT RESTORATION AND CONSERVATION ACTIONS

Ecosystem resilience – passive restoration

On Guadalupe Island, after completing the goat eradication in 2007, native vegetation started to naturally recover. Plants considered extinct or extirpated have been rediscovered and there have been new plant records for the island, including at least one undescribed species. A survey conducted in 2001 on the endemic variety of the Monterey pine—there are five endemic varieties: three in the USA and two in Mexico, all the original seed source for plantations around the world—estimated that there were only 220 adult pines left (Rogers, et al., 2006). Since the goats were eradicated, the number of new seedlings has increased to several thousands (Fig. 1). Not only have the trees have recovered, but shrubs are also returning with full strength, competing well with invasive grasses. *Ceanothus arboreus*, a shrub able to reach 6 or 7 m and a new record for the island (Junak, et al., 2005), is now very common around the cypress and pine-oak forests. Also, the maritime desert scrub in the area most impacted by the goats' presence has changed from almost 0% native vegetation coverage (areas dominated by European grasses) to 52% (Ceceña-Sánchez, 2014).

Active restoration of plant communities

In a review of passive vs active restoration effects on forest recovery, Meli, et al. (2017) suggest observing the system for a few years after intervention to inform better decisions regarding active restoration actions. In the case of Guadalupe Island, feral goat eradication was completed in 2007. The active restoration project started in 2015. Over a period of almost 10 years we documented and measured the recovery of species. Not all recovered at the same speed. Some trees, shrubs, and forbs are recovering at a fast pace. However, there are many species that still remain very fragile, given their low numbers (*Juniperus californica* < 10 known individuals; insular oak, *Quercus tomentella* < 50 adult trees; *Cistanthe guadalupensis*, almost absent from the main island and surviving only on islets), and there are others whose distribution has decreased historically from forests to small isolated patches (e.g. *Cupressus guadalupensis*).

In order to achieve the island's full recovery, the active restoration of vegetation and eroded and degraded soils was the next conservation step. The negative effects of overgrazing and soil compaction are exacerbated on a volcanic island where soil (even some of the most productive soil (Ugolini & Dahlgren, 2002)) is limited and very susceptible to loss due to erosion. A study focused on the cypress forest on Guadalupe Island concluded than the erosion rates were exceptionally high, with a minimum recorded loss of 44 ton/ha/year and the maximum 142 ton/ha/year (Ramos Franco, 2007). Although it was estimated only for the cypress forest, the erosion problem is evident across the whole island, especially at higher elevations. To date, GECI in collaboration with the National Forestry Commission (CONAFOR) and CONANP, and other partners, such as the Mexican Navy (SEMAR) and the local fishermen's cooperative *Abuloneros y Langosteros*, is implementing a 700 ha project to accelerate the recovery of native vegetation communities.

The project involves reforestation, erosion control, and fire prevention actions for different plant communities. Reforestation is being implemented over 583 ha: 33 ha of palm forest; 120 ha of pine-oak forest; 261 ha of cypress forest, 60 ha of juniper woodland and 109 ha of maritime desert scrub. Erosion control actions (17 ha) are focused only on the cypress forest, in an area with slopes of 27%, loss of around 75% of the superficial soil layer, and deep gullies (Ramos Franco 2007), which is considered as extreme degradation (CONAFOR, 2004). On the other hand, due to a fire which occurred in 2008 in the cypress forest, the quantity of accumulated fuel was alarming, around 110 t/ha on average, with a maximum of 1,000 t/ha in certain areas (Luna-Mendoza et al. 2016). For this reason, fire management actions were focused here. The goal is to carry out fuel reduction (through manual removal of surface fuels and increasing the height to live crown) in 100 ha and to restore 10 km of firebreaks.

An on-site nursery was built as part of the project (Fig. 2). The nursery (480 m^2) is surrounded by a mouse-proof, galvanised steel fence of 50 m × 30 m, as mice are responsible for the loss of huge amounts of seed and seedlings at early stages. Around 15 species of native and endemic species are being produced: Guadalupe pine, Guadalupe cypress, Guadalupe palm, insular oak, island hazardia, Guadalupe lupin (*Lupinus niveus*), Guadalupe phacelia (*Phacelia phyllomanica*), island malva (*Malva occidentalis*), Guadalupe rock daisy (*Perityle incana*), among others. Species produced were chosen based on their rarity on the island (e.g. *Leptosyne gigantea* and *Cistanthe guadalupensis*); endangerment (e.g. juniper); propagation material available; potential as nurse plant (e.g. *Sphaeralcea* spp.); importance as food or shelter for native invertebrates and landbirds (e.g. *Senecio palmeri*) and effectiveness at retaining soil (e.g. *Calystegia macrostegia* ssp. *macrostegia*). Their allocation was based on historic information of former vegetation communities as well as observations of where there has been natural recruitment.

Fig. 1 Recovery of Guadalupe pine (*Pinus radiata* var. *binata*) from 220 individuals (adult trees) to several thousands in ten years. Photo credits: GECI Archive/J.A. Soriano.

Fig. 2 Plant nursery on Guadalupe Island with the capacity to produce over 60,000 plants per year. Photo credits: GECI Archive/J.A. Soriano.

For example, goat removal allowed the recovery of the island *Ceanothus* from seed surviving in the seed bank. Now the species grows close to the cypress and pine-oak forests, but individuals are sparse. However, for this species, as seed is not capable of long distance dispersal by itself (Minnich, 1982), it can take a long time to recover its original coverage. Another species, the Guadalupe lupin, is spreading at a fast pace, but being a legume with big seed depends on rainfall to disperse seed downslope. On Guadalupe the dispersal pathways are limited, as seed-eating birds are few and native terrestrial mammals are absent. In the case of the endemic cistanthe, although the species is very common on the islets, on the main island only three individuals have been recorded in the last 10 years. This is one of the lost species of the maritime desert scrub, and a goal of this project is to reintroduce it to this vegetation community.

Up to June 2018, 90,000 plants have been produced in the nursery. The final goal is to produce 160,000 plants, mostly trees and shrubs. We have planted almost 40,000 plants, most of them trees. To date we have nearly completed reforestation of the pine-oak and cypress areas (Fig. 3). Some challenges have arisen: logistics linked to working on an island; lack of plant propagation information for some species (especially endemics); limited amount of seed (insular-endemic or Guadalupe-endemic species and very few individuals left); diseases; and limited amount of water (relying mostly on the fog). In the last rainy season there was virtually no rainfall and fog has been very intermittent and scarce. We are therefore using resources such as the commercial hydrophilic polymer based on polyacrylamide, called Lluvia sólida®. We add 1.5 to 2 l of this hydrated polymer to each plant. So far, results are encouraging, as survivorship of planted individuals is above 85%. Regarding soil restoration, mechanical methods to control and prevent erosion, such as check dams and contour barriers, have been implemented. More than 1,500 m of contour barriers of rocks and logs have been built as well as 66 m^3 of rock and log check dams. So far, 27 hectares have been cleared of fuel and 1,500 m^3 of material has been removed.

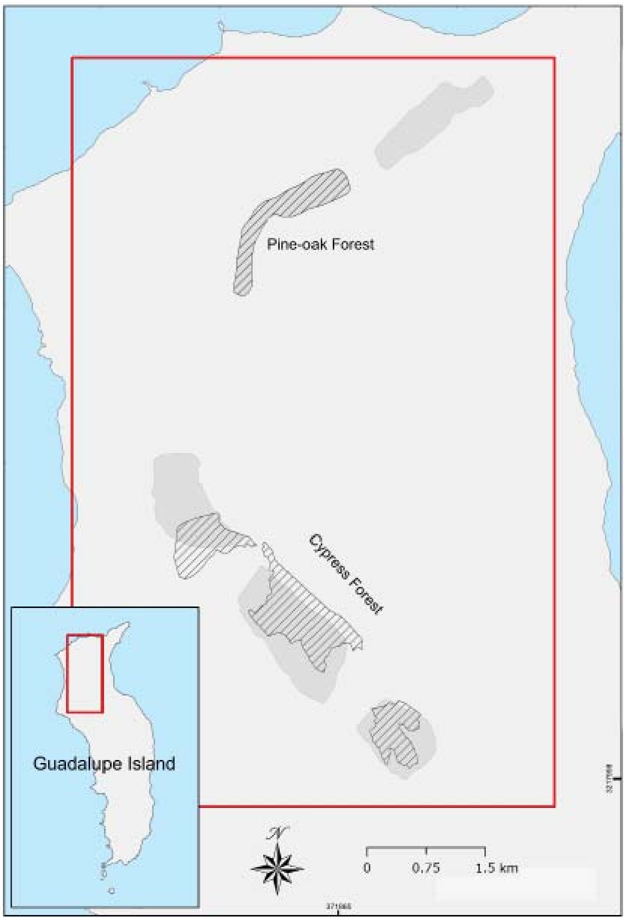

Fig. 3 Reforested areas on Guadalupe Island. The polygon in the north relates to pine-oak forest and the three polygons in the south are in the cypress forest area. Shaded (grey) areas are the original areas proposed in the project and striped areas are the ones completed (up to June 2018).

WHAT'S NEXT?

There still much to be done on Guadalupe Island. A period of 130 years of feral goats on the island caused severe ecosystem degradation. However, with collaborative projects, such as the one described here, we are heading in the right direction to restore the island's ecosystem services and biodiversity. Future projects, conducted in collaboration with CONANP, are to continue with fuel reduction actions in the cypress and pine-oak forests, to collect seed of endemic species to be stored at national seed banks and to estimate carbon sequestration in forest ecosystems. As a country, Mexico is now fully committed to the recovery of its islands, going a step further than eradication actions. On Socorro Island (Revillagigedo Archipelago; Pacific Ocean), feral sheep were removed a few years ago. This island is very similar to Guadalupe and faces the same challenges: soil degradation and a need to do some active restoration of the forest, not only for the vegetation itself but to restore the habitat for many endemic land birds which were close to extinction. In some cases we cannot wait for the islands to recover naturally, especially where other threats are still present (other invasive mammals). Currently other islands in Mexico, such as Espíritu Santo Island (Gulf of California), and María Cleofas Island (Las Marías Archipelago, Pacific Ocean), are being cleared of herbivores, and hopefully more active restoration actions could be established on these sites in the near future.

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REFERENCES

Aguirre-Muñoz, A., Samaniego-Herrera, A., García-Gutiérrez, C., Luna-Mendoza, L., Rodríguez-Malagón, M. and Casillas-Figueroa, F. (2005). 'El Control y la Erradicación de Fauna Introducida Como Instrumento de Restauración Ambiental: Historia, Retos y Avances en México'. In: O. Sánchez, E. Peters Recagno, R. Márquez-Huitzil, E. Vega, G. Portales, M. Valdés and D. Azuara (eds.) *Temas sobre Restauración Ecológica,* pp. 215–229. México, D.F.: Instituto Nacional de Ecología.

Aguirre-Muñoz, A., Samaniego-Herrera, A., Luna-Mendoza, L., Ortiz-Alcaraz, A., Rodríguez-Malagón, M., Méndez-Sánchez, F., Félix-Lizárraga, M., Hernández-Montoya, J.C., González-Gómez, R., Torres-García, F., Barredo-Barberena, J.M. and Latofski-Robles, M. (2011). 'Island restoration in Mexico: ecological outcomes after systematic eradications of invasive mammals'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 250–258. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

AZE. (2010). *2010 AZE Update*. Alliance for Zero Extinction. <www.zeroextinction.org>. Accessed 18 June 2017.

Berglund, H., Järemo, J. and Bengtsson, G. (2009). 'Endemism predicts intrinsic vulnerability to nonindigenous species on islands'. *The American Naturalist* 174**:** 94–101.

Brook, B.W., Sodhi, N.S. and Bradshaw, C.J.A. (2008). 'Synergies among extinction drivers under global change'. *Trends in Ecology and Evolution* 23**:** 453–460.

Campbell, K. and Donlan, C.J. (2005). 'Feral goat eradications on islands'. *Conservation Biology* 19**:** 1362–1374.

Camps, J. and Ramos, M. (2012). 'Grape harvest and yield responses to inter-annual changes in temperature and precipitation in an area of north-east Spain with a Mediterranean climate'. *International Journal of Biometeorology* 56**:** 853–864.

Castro, R., Mascarenhas, A., Sánchez-Barba, A., Durazo, R. and Gíl-Silva, E. (2005). 'Condiciones Meteorológicas en el sur de Isla Guadalupe'. In: E. Peters and K. Santos Del Prado (eds.) *Restauración y Conservación de la Isla Guadalupe,* pp. 27–37. México, D.F.: Instituto Nacional de Ecología.

Ceceña-Sánchez, M.L. (2014). 'Estudio Fitosociológico de las Comunidades Vegetales de los Matorrales de Isla Guadalupe, México'. MSc thesis. Ensenada: Universidad Autónoma de Baja California.

Chynoweth, M.W., Litton, C.M., Lepczyk, C.A., Hess, S.C. and Cordell, S. (2013). 'Biology and impacts of Pacific Island invasive species. 9. *Capra hircus*, the feral goat (Mammalia: Bovidae)'. *Pacific Science* 67: 141–156.

Coblentz, B.E. (1978). 'The effects of feral goats (*Capra hircus*) on island ecosystems'. *Biological Conservation* 13: 279–285.

CONAFOR. (2004). *Protección, Restauración y Conservación de Suelos Forestales. Manual de Obras Prácticas*. Jalisco, México: Comisión Nacional Forestal, Secretaría del Medio Ambiente y Recursos Naturales.

Courchamp, F., Chapuis, J.-L. and Pascal, M. (2003). 'Mammal invaders on islands: Impact, control and control impact'. *Biological Reviews* 78**:** 347–383.

Crum, H. and Miller, H.A. (1956). 'Bryophytes from Guadalupe Island, Baja California'. *The Southwestern Naturalist* 1**:** 116–120.

Delgado-Argote, L.A., García-Abdeslem, J. and Mendoza-Borunda, R. (1993). 'Correlación Geológica entre la Batimetría y los Rasgos Estructurales del Oriente de la Isla Guadalupe, México'. In: L.A. Delgado-Argote and A. Martín-Barajas (eds.) *Contribuciones a la Tectónica de México,* pp. 1–11. México: Monografía No. 1 de la Unión Geofísica Mexicana.

Garcillán, P.P., Vega, E. and Martorell, C. (2012). 'The *Brahea edulis* palm forest in Guadalupe Island: A North American fog oasis?' *Revista Chilena de Historia Natural* 85**:** 137–145.

Granda, E., Rossatto, D., Camarero, J.J., Voltas, J. and Valladares, F. (2014). 'Growth and carbon isotopes of Mediterranean trees reveal contrasting responses to increased carbon dioxide and drought'. *Oecologia* 174**:** 307–317.

Hanna, G.D. (1925). 'Expedition to Guadalupe Island, Mexico, in 1922'. *Proceedings of the California Academy of Sciences, Fourth Series* XIV**:** 217–275.

Holdgate, M.W. (1967). 'The influence of introduced species on the ecosystems of temperate oceanic islands'. In: C.E. Conrad and W.C. Oechel (eds.). *Towards a new Relationship of Man and Nature in Temperate Lands. Part III. Changes due to Introduced Species,* pp. 152–177. Lucerne: IUCN.

Huey, L.M. (1925). 'Guadalupe Island: An object lesson in man-caused devastation'. *Science* 61**:** 405–407.

Jehl Jr., J.R. and Everett, W.T. (1985). 'History and status of the avifauna of Isla Guadalupe, Mexico'. *Transactions of the San Diego Society of Natural History* 20**:** 313–336.

Junak, S.J., Keitt, B., Tershy, B., Croll, D., Luna-Mendoza, L. and Aguirre-Muñoz, A. (2005). 'Esfuerzos Recientes de Conservación y Apuntes sobre el Estado Actual de la Flora de Isla Guadalupe'. In: E. Peters and K. Santos del Prado (eds.) *Restauración y Conservación de la Isla Guadalupe,* pp. 83–93. México, D.F.: Instituto Nacional de Ecología.

León de La Luz, J.L., Rebman, J.P. and Oberbauer, T.A. (2003). 'On the urgency of conservation on Guadalupe Island, Mexico: Is it a lost paradise?' *Biodiversity and Conservation* 12**:** 1073–1082.

León de La Luz, J.L., Rebman, J.P. and Oberbauer, T.A. (2005). 'El Estado Actual de la Flora y la Vegetación de Isla Guadalupe'. In: E. Peters and K. Santos del Prado (eds.) *Restauración y Conservación de la Isla Guadalupe,* pp. 55–65. México, D.F.: Instituto Nacional de Ecología.

León Portilla, M. (1989). *Cartografía y Crónicas de la Antigua California.* México, D.F.: Universidad Nacional Autónoma de México.

Luna-Mendoza, L., Keitt, B. and Junak, S. (2007). 'The restoration of Guadalupe Island revisited'. *Fremontia* 35**:** 14–17.

Luna-Mendoza, L., Aguirre-Muñoz, A., Méndez-Sánchez, F., Vargas-Hernández, J.J., Rodríguez-Chávez, O.G., Martínez-Rodríguez, J.M., Cosio-Muriel, D., Gómez-Reyes, J., Puebla-Hernández O. y Luvianos-Colin, S. (2016). *Restauración Integral del Ecosistema del Bosque de Ciprés de la RB Isla Guadalupe. Reporte final*. Ejercicio PROCER 2016. GECI-COLPOS-CIPACTLI. Report for the Comisión Nacional de Áreas Naturales Protegidas-Reserva de la Biosfera Isla Guadalupe.

Mack, R.N., Simberloff, D., Lonsdale, W.M., Evans, H., Clout, M.N. and Bazzaz, F.A. (2000). 'Biotic invasions: Causes, epidemiology, global consequences, and control'. *Ecological Applications* 10**:** 689–710.

Meli, P., Holl, K.D., Benayas, J.M.R., Jones, H.P., Jones, P.C., Montoya, D. and Mateos, D.M. (2017). 'A global review of past land use, climate, and active vs passive restoration effects on forest recovery'. *PLOS ONE* 12: e0171368.

Méndez-Sánchez, F.A. (2012). 'Co-management and Small-scale Fisheries in Mexico: The Case of a Fishers' Cooperative in Cedros and San Benito islands'. MSc thesis. Auckland, NZ: University of Auckland.

Minnich, R. A. (1982). 'Grazing, Fire and the Management of Vegetation of Santa Catalina Island, California'. In: C.E. Conrad and W.C. Oechel (eds.). *Proceedings of the Symposium on Dynamics and Management of Mediterranean-type Ecosystems,* pp. 444–449. Berkeley, CA: USDA, Forest Service, Pacific Southwest Forest and Range Experiment Station, General Technical Report PSW-58.

Minnich, R.A., Franco Vizcaino, E. and Dezzani, R.J. (2009). 'The El Niño/Southern Oscillation and precipitation variability in Baja California, Mexico'. *Atmósfera* 13**:** 1–20.

Moran, R. (1996). *The Flora of Guadalupe Island, Mexico,* San Francisco, USA: California Academy of Sciences.

Morgan, L., Maxwell, S., Tsao, F., Wilkinson, T.A.C. and Etnoyer, P. (2005). *Marine Priority Conservation Areas: Baja California to the Bering Sea*. Montreal, Canada: Commission for Environmental Cooperation of North America and the Marine Conservation Biology Institute.

Morrone, J.J., Organista, D.E., Zúñiga, C.A. and Bousquets, J.L. (1999). 'Preliminary classification of the Mexican biogeographic provinces: A parsimony analysis of endemicity based on plant, insect, and bird taxa'. *The Southwestern Naturalist* 44**:** 507–514.

Morrone, J.J., Espinosa-Organista, D. and Llorente-Bousquets, J. (2002). 'Mexican biogeographic provinces: Preliminary scheme, general characterizations, and synonymies'. *Acta Zoológica Mexicana* 85**:** 83–108.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G.A. and Kent, J. (2000). 'Biodiversity hotspots for conservation priorities'. *Nature* 403: 853–858.

Oberbauer, T.A. (2005). 'La Vegetación de Isla Guadalupe: Entonces y Ahora'. In: E. Peters and K. Santos del Prado (eds.). *Restauración y Conservación de la Isla Guadalupe,* pp. 39–53. México, D.F.: Instituto Nacional de Ecología.

Parkes, J., Henzell, R. and Pickles, G. (1996). *Managing Vertebrate Pests: Feral Goats*. Canberra: Australian Government Publishing Service.

Ramos Franco, C.A. (2007). 'Propuesta de Manejo de la Erosión Hídrica para la Restauración de Suelo del Bosque de Ciprés de Isla Guadalupe.' MSc thesis. Ensenada: Universidad Autónoma de Baja California.

Raven, P.H. (1965). 'The Floristics of the California Islands'. In*:* R.N. Philbrick (ed.) *1st Symposium on the Biology of the California Islands*, pp. 57–71. Santa Barbara, California: Santa Barbara Botanic Garden.

Reaser, J.K., Meyerson, L.A., Cronk, Q., De Poorter, M., Eldrege, L.G., Green, E., Kairo, M., Latasi, P., Mack, R.N., Mauremootoo, J., O'Dowd, D., Orapa, W., Sastroutomo, S., Saunders, A., Shine, C., Thrainsson, S. and Vaiutu, L. (2007). 'Ecological and socioeconomic impacts of invasive alien species in island ecosystems'. *Environmental Conservation* 34**:** 98–111.

Rebman, J.P., Oberbauer, T.A. and León de La Luz, J.L. (2005). 'La Flora de Isla Guadalupe y sus Islotes Adyacentes'. In: E. Peters and K. Santos del Prado (eds.) *Restauración y Conservación de la Isla Guadalupe,* pp. 67–81. México, D.F.: Instituto Nacional de Ecología.

Rogers, D.L., Matheson, A.C., Vargas-Hernández, J.J. and Guerra-Santos, J.J. (2006). 'Genetic conservation of insular populations of Monterey pine (*Pinus radiata* D. Don)'. *Biodiversity and Conservation* 15**:** 779–798.

SARH-Colegio de Postgraduados. (2010). *Normales Climatológicas 1951–2010*. <http://www.cm.colpos.mx/meteoro/progde/norm/norm24/listd.htm>. Accessed 28 November 2012.

Searcy-Bernal, R., Ramade-Villanueva, M.R. and Altamira, B. (2010). 'Current status of abalone fisheries and culture in Mexico'. *Journal of Shellfish Research* 29**:** 573–576.

Simberloff, D. (1995). 'Why do introduced species appear to devastate islands more than mainland areas?' *Pacific Science* 49**:** 87–97.

Sol, D. (2007). 'Do Successful Invaders Exist? Pre-Adaptations to Novel Environments in Terrestrial Vertebrates'. In: W. Nentwig (ed.) *Biological Invasions,* pp. 127–141. Berlin, Heidelberg: Springer.

Ugolini, F.C and Dahlgren, R.A., (2002). 'Soil development in volcanic ash'. *Global Environmental Research–English Edition* 6: 69–82.

Veitch, C.R., Clout, M.N. and Towns, D.R. (eds.) (2011). '*Island invasives: eradication and management'.* Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Vidal, R.M., Berlanga, H. and del Coro Arizmendi, M. (2009). 'Important Bird Areas: Mexico'. In: C. Devenish, D.F. Díaz Fernández, R.P. Clay, I. Davidson and I. Yépez Zabala (eds.) *Important Bird Areas Americas - Priority Sites for Biodiversity Conservation.* Quito: BirdLife International (BirdLife Conservation Series No. 16).

Whittaker, R.J. and Fernández-Palacios, J.M. (2007). *Island Biogeography: Ecology, Evolution, and Conservation.* New York: Oxford University Press.

Wilkinson, T., Wiken, E., Bezaury-Creel, J., Hourigan, T., Agardy, T., Herrmann, H., Janishevski, L., Madden, C., Morgan, L. and Padilla, M. (2009). *Marine Ecoregions of North* America*.* Montreal: Commission for Environmental Cooperation.

Canna seabird recovery project: 10 years on

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Abstract Rats were eradicated in 2005–2006 from the islands of Canna and Sanday, Scotland (total area 1,320 ha). Poison bait was laid from December 2005 onwards and the last rat was killed in February 2006. An intensive period of monitoring over the next two years confirmed that no rats remained on the islands. Seabirds have been monitored on Canna for nearly 50 years and some species have shown good evidence of recovery since the eradication. Other species have not recovered and this may have been due to mortality caused by food shortages or storm events which have been impacting seabirds in the region. These regional changes in pressures affecting the seabird populations make the interpretation of the impacts of the rat eradication programme much more difficult. Atlantic puffins, formerly confined to offshore stacks, have recolonised sites on the mainland of Canna and a count of over 2,000 was recorded in 2016. Manx shearwaters, which had ceased nesting in the monitored colony have made a slow recovery to one or two pairs in 2016. Productivity has also increased from a low of 0.2 chicks per nest in the 1990s to 0.74 in 2017. European shags nesting in boulder colonies were most susceptible to rat predation. One such colony has recovered from 45 nests in 2005 to 75 in 2016 and productivity increased from less than 0.7 chicks per nest to an average of 1.6 following eradication. Populations of shags nesting in cliff locations have shown no recovery or have declined. Mew gulls, which nest along the shoreline, have increased from five to over 30 pairs. Other seabirds, such as common guillemots and black-legged kittiwakes, have shown no clear trends and are probably affected by other factors. Rabbit populations have increased on both islands, reaching an estimated 15,500 animals in 2013 that were causing considerable damage through grazing, erosion, and disturbance of archaeological remains. It is unclear whether the increase in rabbit numbers can be attributed to rat eradication. An intensive control programme has brought the rabbit population under control. While some seabirds have responded positively to the rat eradication, the response of some has been slow and others have not responded, probably as a result of regional pressures on their survival. It is important that monitoring of both seabirds and rabbits continues to track the success of this important seabird colony.

Keywords: contingency plans, invasive species, monitoring, quarantine, rabbits, rat eradication

INTRODUCTION

The islands of Canna and Sanday, which are connected at low tide (total area 1,320 ha) are in the Inner Hebrides, off western Scotland. They are owned and managed by the National Trust for Scotland and are designated as a Special Protection Area because of their internationally important seabird colony. Seabird populations and breeding success have been monitored on the islands since 1969 by the Highland Ringing Group, making this one of the best monitored sites in Scotland. The main species present are common guillemot (*Uria aalge*), razorbill (*Alca torda*), black-legged kittiwake (*Rissa tridactyla*), northern fulmar (*Fulmaris glacialis*), European shag (*Phalacrocorax aristotelis*) and Atlantic puffin (*Fratercula arctica*). Manx shearwater (*Puffinus puffinus*) used to be present in large numbers (1,500 apparently occupied burrows) but suffered very poor breeding success and, by 2000, had been virtually wiped out (Swann, 2002) Other seabirds were also recorded as declining. Predation by a large population of brown rats (*Rattus norvegicus*) was identified as the likely cause of this decline, from three types of evidence: 1. Direct observation of increasing numbers of rats foraging in the seabird colonies and of stashes of predated egg shells and carcases; 2. Declining numbers and decreasing breeding success of vulnerable species; and 3. Changing nesting behaviour of breeding seabirds moving to less accessible sites. After favourable feasibility studies (Bell & Bell, 2004), it was decided to eradicate the rats using poison bait, and funding was obtained from the EU LIFE fund, Scottish Natural Heritage and the National Trust for Scotland. The programme objective was to halt declines in breeding seabird populations on Canna and Sanday and to facilitate their recovery and long-term protection. It was carried out under contract by Wildlife Management International, starting in late 2005. By February 2006 the last rat sign was detected and, after a two-year period of intensive monitoring, the island was declared rat-free in 2008 (see Bell, et al., 2011).

Canna and Sanday are inhabited by a population of 15–20 people and are farmed with a mixture of sheep and cattle. They are served by a ferry service five days a week. The harbour and all houses are in the eastern portion of the islands, where there are a number of fenced pastures and some planted woodlands. There are high cliffs around much of the coast, particularly to the north and west, and the higher ground is mostly covered in wet heath. There is a population of distinctive, large (presumed introduced) field mice (*Apodemus sylvaticus*) that were not removed by the rat poisoning programme and a substantial (introduced) population of rabbits (*Oryctolagus cuniculus*). There is a small number of (introduced) hedgehogs (*Erinaceus europaeus*), and regular sightings of European otters (*Lutra lutra*), but no other ground predators. Two pairs of white-tailed eagles (*Haliaeetus albicilla*), one pair of golden eagles (*Aquila chrysaetos*), up to two pairs of peregrine falcons (*Falco peregrinus*), about 15 pairs of common buzzards (*Buteo buteo*) and ravens (*Corvus corax*) regularly breed on the island and there are small numbers of great skuas (*Catharacta skua*) and great black-backed gulls (*Larus marinus*). Predation by eagles, and possibly locally by otters, may impact populations of northern fulmar, while ravens may impact shags, particularly on the cliff-nesting colonies (Swann, 2008; Swann, et al., in press), but none of these predators exerts substantial pressure on other seabird populations.

Following the eradication programme, biosecurity measures were put in place, consisting of continuous monitoring (wax blocks and kill traps), quarantine and contingency plans. No incursions of rats have been detected.

The main post-eradication monitoring has been a continuation of the long-running seabird programme which can be used to detect any changes following the eradication of rats. A rapid expansion in the rabbit population was

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noted in 2011–2013 which caused locally severe grazing and considerable erosion through collapsed burrows. This necessitated the introduction of control measures consisting of a rapid reduction cull in January–March 2014, followed by continuous lower level culling thereafter. There have been surveys of vegetation condition (SNH, 2014), grassland fungi (Murfitt & Macdonald, 2012), invertebrates (Rotheray & Lyszkowski, 2012) and lichens (Acton, 2011). Any changes in the vegetation are thought to be due to rabbit grazing or livestock management, rather than directly related to the removal of rats.

This paper reviews the changes in seabird population size and breeding success reported in Swann, et al. (in press) and discusses fluctuations in rabbit populations and the control measures employed.

MATERIAL AND METHODS

All seabird population and productivity estimates follow the methodology of Walsh, et al. (1995) and are described in Swann, et al. (in press). Seabird population estimates for the years 1995 to 2017, derived from Swann (2008) and Swann, et al. (in press), were analysed. For northern fulmar, European shag, black-legged kittiwake, mew gull, herring gull, lesser black-backed gull and greater black-backed gull, the population figures represent the number of apparently occupied territories or apparently occupied nest sites throughout the two islands. For common guillemots and razorbills, they are the number of nest sites at a small number of accessible monitoring plots. Nest sites were recognised by the presence of an egg, chick or, in the case of razorbill, a shell or dense mass of droppings. The data were divided into the 11 years prior to the eradication (1995–2005) and the 12 years following (2006–2017). Population trends were determined by fitting an exponential line to each set of data. The exponents shown on the graphs represent r in the equation $N = e^{rt}$, where N is the population size (pairs) and t is the time in years.

Productivity estimates represent large young/chicks per occupied nest within all the monitoring plots for northern fulmar and black-legged kittiwake. For great black-backed gulls and herring gulls, they represent large chicks per apparently occupied territory. For European shags, the breeding success (chicks per occupied nest) is separated into plots within the boulder colonies and plots on cliff sites. Productivity of Manx shearwaters was calculated as the number of large chicks produced per occupied burrow located.

In June 2013 and 2014 (the latter following a rabbit reduction programme in January–March 2014) rabbit populations were estimated by walking over the entire island using the Modified McLean scale, an 8-point scale based on the observation of rabbits and pellets (NPCA, 2012). Densities were estimated within different topographic or vegetation units and the area of each unit was calculated using GIS. The approximate population of rabbits was then calculated by multiplying the area of each unit by the densities corresponding with the McLean scale. Rabbit culls were recorded by the trappers on a weekly basis in different zones.

RESULTS

Seabird population size

Seabird population trends are shown in Fig. 1. Numbers of breeding pairs of almost all species, except the black-legged kittiwake were declining prior to 2005 but after 2006, three gull species were stable or slowly increasing and populations of all other species, except the northern fulmar, showed a reduced rate of decline. It should be noted that the correlation coefficients (R^2) were low, showing significant declines for most species prior to eradication but significant increases only for mew gull and lesser black-backed gull after eradication.

In the case of Manx shearwater, the population had already fallen to very low levels prior to 1995 from a high

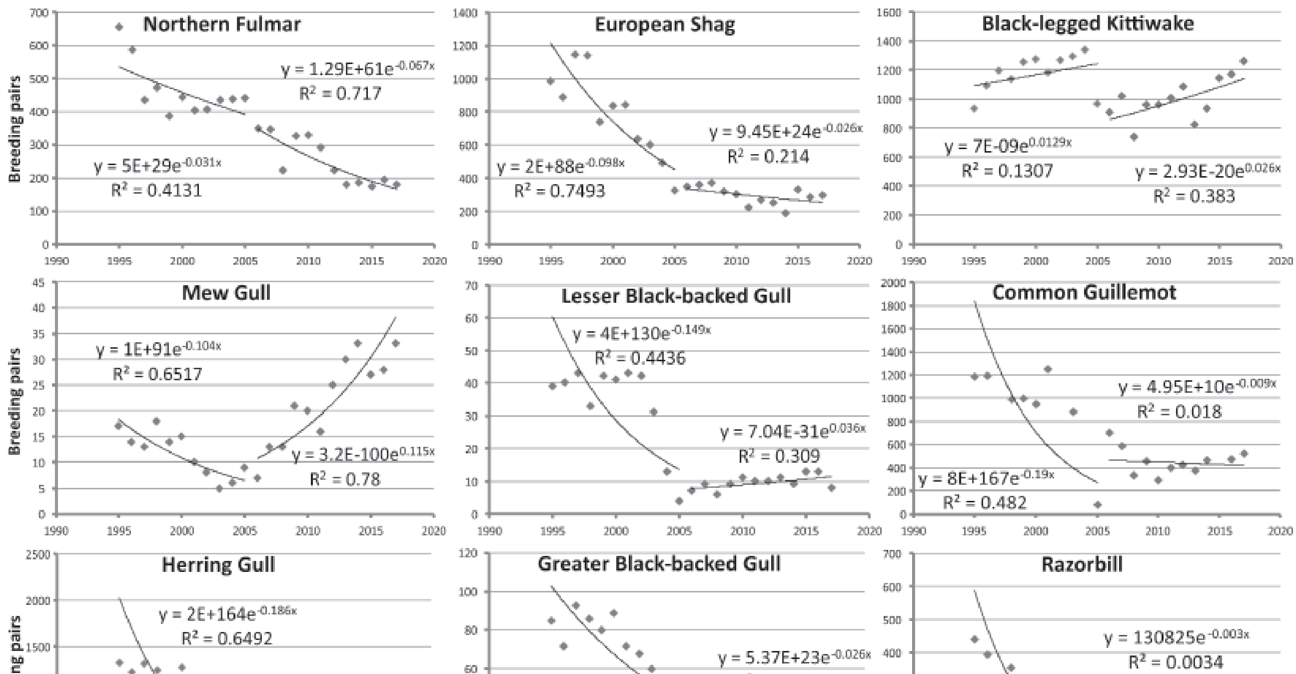

Fig. 1 Population trends in the numbers of breeding pairs of nine species of seabird on Canna during the period 1995–2017. Data were split into pre-eradication (1995–2005) and post-eradication (2006–2017) and exponential graphs fitted to each.

of 1,500 pairs in the mid-1970s. By 2000, no nests could be located in the main colony, along Tarbert Road (see Fig. 2). Following the rat eradication in 2006 the first nesting shearwater was detected again in the Tarbert Road colony, but by 2017 this had grown to only two nests. A further four nests were located at accessible locations in the west of the island and all of these nests were monitored for breeding success. Based on calling behaviour at night, it was estimated that there were more nests, possibly 10–20 pairs, in inaccessible locations (M. Carty, pers. comm.)

Mew gulls (*Larus canus*) nest along the shore on Canna. Numbers have increased from nine pairs at the time of the eradication to around 30 pairs in recent years. Other large gulls, especially herring gulls (*Larus argentatus*), declined rapidly in the 1990s and early 2000s similar to colonies elsewhere in Scotland. Since 2006, populations on Canna have remained low.

The overall number of breeding European shags had dropped to about 300 nests by 2005. Different sub-colonies have performed differently, with those nesting under boulders declining most rapidly (Swann, 2005). One boulder colony, Lamasgor, has subsequently shown an increase from 45 nests in 2005 to 75 in 2016. Other sub-colonies, particularly those on cliffs, however, have remained stable, or declined, so that overall the population has not increased. Nevertheless, the overall rate of decline of all of the nests on the island has slowed (Fig. 1).

Atlantic puffin breeding populations are difficult to monitor, especially because, prior to 2005, they were virtually confined to two inaccessible stacks. After 2006 they began to spread to sites along the north coast of Canna at Guegasgor (see Fig. 2). Where more accurate census methods are not practical, Walsh, et al. (1995) recommend counting puffins rafting on the sea near the colony. A count of rafting puffins in 1995 gave 1190 individuals while, in 2016, 2050 were counted. Though not conclusive, this is consistent with an increase in breeding numbers as well as the recorded expansion of the puffin colony to colonise previously unoccupied sites on the mainland of Canna.

Seabird breeding success

Between 1999 and 2004, breeding success of European shags in monitored nests in boulder colonies had dropped to <0.7 young per nest. Since the eradication breeding success has averaged 1.6 young per nest. In contrast, breeding success at colonies nesting on cliff ledges averaged 0.97 per nest over the years 2006–2017 (Table 1).

Manx shearwater breeding productivity has greatly improved. In the 1980s it averaged 0.6 young/nest, dropping to <0.2 young/nest in the mid-1990s. Since 2009. out of a total of 19 burrows that were known to contain an egg, 14 successfully produced a chick, an average of 0.74 young/nest.

Rabbit population

The rabbit population on Canna has routinely fluctuated in response to disease and weather conditions, but numbers had not been formally monitored. By 2013, rabbit numbers were causing serious damage to agricultural interests and

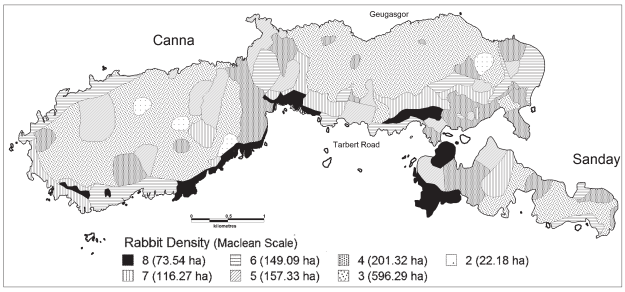

Fig. 2 Estimated rabbit density on Canna in July 2013. Densities relate to the Modified McLean Scale (NPCA, 2012).

were damaging archaeological remains. A rapid assessment of rabbit population density gave an overall population estimate of 15,500 (Fig. 2). As a consequence, a rabbit population reduction exercise was carried out in January–March 2014, with a total of 8,200 rabbits removed. This brought the population down to an estimated 7,000 by July 2014. Continuous culling of around 5,000 rabbits a year has maintained it at a level where agricultural damage is acceptable.

DISCUSSION

The response of Canna seabirds to the successful rat eradication in winter 2005/06 was species specific. For some populations, such as the Atlantic puffin, the mew gull and the boulder-nesting colonies of European shag, there have been apparent increases in numbers of breeding pairs. The colony of European shags on Canna is of international importance and formerly numbered 1,800 pairs. Many of them nest under boulders in relatively accessible locations and these birds were found to be particularly susceptible to rat predation, declining most rapidly. Simultaneously, there was a shift whereby a greater proportion of shags started nesting on more exposed cliff locations where they were less susceptible to rat predation but more exposed to avian predation (Swann, 2005). Following the eradication of rats, some boulder colonies have expanded rapidly but the cliff-nesting birds have continued to decline. The net effect has been a continuing decline in the overall shag population but at a reduced rate.

Similarly, although populations of common guillemots and razorbills have continued to decline there has been a slowing in the rate of decline. These two species of auk nest in similar locations to the European shags and were also affected by rat predation. Guillemots were badly affected by a severe period of stormy weather in western Scotland in late summer 2004 (Swann, 2004) which caused heavy mortality of both adults and chicks. Ringing studies showed that adult survival dropped from a long term average of 0.9 to 0.6 between 2004 and 2005. Breeding numbers of guillemots on Canna were very low in 2005 and breeding success remained low until 2008 probably as a result of subsequent food shortages. Breeding success improved in 2009 and the population has started to increase.

Numbers of breeding razorbills showed a sharp jump in 2006, and this was almost certainly due to a reduction

Table 1 Breeding productivity of Manx shearwaters at Tarbert Road colony and European shags in the boulder and cliff colonies on Canna, 2001–2017. "-" = Productivity not monitored.

Year	01	02	03	04	05	06	07	08	09	10	11	12	13	14	15	16	17
Manx shearwater	-	-	-	-	-	-	-	-	0	1	1	0.25	0.5	0.2	0.5	1	0.5
Shag – boulder colonies	0.1	0.26	0.16	0.01	0.7	1.2	1.4	1.5	1.8	-	1.4	-	1	1.5	1.8	2.2	1.3
Shag – cliff colonies	-	-	-	1.4	0.7	0.7	1.0	0.3	0.7	1.5	0.8	1.6	1.0	0.7	1.2	1.0	1.2

in predation by rats, eggs appearing in areas that had been clear of nesting for several years (Swann, 2008). However, the number of occupied breeding sites then remained roughly stable at this level until 2016, and the breeding success remained low, probably as a result of food shortage (Swann, et al., in press). Since occupied breeding sites in this species are identified by the presence of an egg or chick, failure to breed does not necessarily indicate the absence of adults attempting to breed – they may have laid an egg and then left again following predation of the egg. The population increase observed in 2006 may therefore indicate higher early survival of eggs rather than a greater number of adults attending the colony.

The large gulls, especially the lesser black-backed gull and the herring gull, suffered a particularly sharp decline in the period 2000–2005 and have since shown a slight increase. Foster, et al. (2017) have analysed this trend and shown that it is closely correlated with the commercial landings of fish in the nearby port of Mallaig, with the gulls feeding extensively on discards from the fishing industry. Since 2006, numbers have stabilised at a much lower level. It is unclear whether the subsequent slow increase of lesser black-backed gulls had anything to do with the reduction in rat predation.

The number of breeding pairs of other species, such as the northern fulmar, have continued to decline and there has been no reduction in the rate of decline. The fulmars nest largely in cliff locations and would have been less affected by rat predation, although it is possible that predation by eagles may be significant (Swann, 2008).

The black-legged kittiwake population has stayed roughly stable over the whole period, although it too suffered poor breeding success in the period 2005–2008, probably as a result of the food shortage experienced by other seabirds in the region. This species typically nests on near-vertical cliffs and is therefore probably the least susceptible to ground predators such as rats. The causes of any changes in population size or breeding success must therefore be sought elsewhere.

One species that was expected to benefit strongly from the removal of rats was the burrow-nesting Manx shearwater. Although there has been a tiny increase in breeding numbers and a clear improvement in breeding productivity, there are still thought to be fewer than 20 pairs nesting on the island. Shearwaters are long-lived and slow-maturing species, possibly not breeding until eight years of age and so endogenous growth of the surviving, relict population would be expected to be slow (Brooke, et al., 2018). However, Canna lies next to the larger Manx shearwater colony on Rum (estimated to be around 60,000 pairs) and is regularly visited by (presumably non-breeding) birds at night. It is apparent that these have not colonised the former colony on Canna to any great extent. The shearwater colony on Rum is subject to predation by a large population of brown rats. The impact of this predation is unclear and it is not known whether the colony is stable or declining (Lambert, et al., 2015). If predation is high, the pressure for emigration from Rum would be lower than from a colony that was limited by shortage of nest sites. An attempt was made to attract breeding birds to re-colonise Canna by playing recordings of shearwater calls at night in 2006 and 2007. This was discontinued because of a lack of obvious success.

It is possible that the growth in the rabbit population is attributable to the removal of rats as young rabbits would be likely to have suffered predation by rats. However, large fluctuations are a characteristic of rabbit populations and high numbers have been reported in previous years. Thus, while 15,500 may seem a high population for Canna, and it undoubtedly caused damage to agricultural interests, it may not be unprecedented. It has been necessary to control rabbits on Canna for many years prior to the eradication of rats although historically the rabbit populations used to cycle due to outbreaks of myxomatosis. Surprisingly, there has been no evidence of myxomatosis in recent years. Total eradication of rabbits has been deemed unfeasible (Bell, 2012) and so it is inevitable that control of rabbit populations will be necessary for the foreseeable future to prevent the build-up of excessive numbers. Although vacated rabbit burrows can provide nest sites for Manx shearwaters, at high densities there is likely to be competition for burrows and this will reduce sites available for shearwaters. The burrows have also caused severe erosion, including large landslips, in some of the former shearwater colonies (Bell, 2012).

Overall, the removal of rats from Canna has had some very beneficial impacts on some species of seabirds but this effect was masked for other species by some very difficult local conditions in the period 2004–2008, firstly by storm-related mortality and subsequently by regional food shortages. The gulls were also impacted by a lack of fisheries discards following a drop in commercial fisheries. Because these external factors occurred at approximately the same time as the rat eradication programme it may take many years for the full benefits to play out. It is clear that continued detailed monitoring of seabird populations and breeding success is vital in unravelling the complex interactions between local conditions on the breeding colony and regional changes in the marine ecosystem.

REFERENCES

Acton, A. (2011). *Canna and Sanday Lichen Survey*. Unpublished report to the National Trust for Scotland, Edinburgh, Scotland.

Bell, E.A. and Bell, B. (2004). *An Assessment of the Proposal to Eradicate Brown Rats from the Isle of Canna, Scottish Hebrides*. Unpublished report to the National Trust for Scotland, Edinburgh, Scotland.

Bell, E., Boyle, D., Floyd, K., Garner-Richards, P., Swann, B., Luxmoore, R., Patterson, A., and Thomas, R. (2011). 'The ground-based eradication of Norway rats (*Rattus norvegicus*) from the Isle of Canna, Inner Hebrides, Scotland'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 269–274. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Bell, E. (2012). *Rabbits on Canna: Assessment and Advice on the Rabbit Situation on Isle of Canna, Inner Hebrides, Scotland*. Unpublished report to the National Trust for Scotland, Edinburgh, Scotland.

Brooke, M.de L., Bonnaud, E., Dilley, B.J., Flint, E.N., Holmes, N.D., Jones, H.P., Provost, P., Rocamora, G., Ryan, P.G., Surman, C., and Buxton, R.T. (2018). 'Seabird population changes following mammal eradications on islands'. *Animal Conservation* 21(1): 3–12.

Foster, S., Swann, R.L. and Furness, R.W. (2017). 'Can changes in fishery landings explain long-term population trends in gulls?' *Bird Study* 64: 90–97.

Lambert, M., Carlisle, S. and Cain I. (2015). *The Role of Brown Rat (Rattus norvegicus) Predation in Determining Breeding Success of Manx Shearwaters (Puffinus puffinus) on Rum*. Scottish Natural Heritage Commissioned Report No. 697, Inverness, Scotland.

Murfitt, A. and Macdonald, A. (2012). *Grassland Fungi Survey; Isle of Canna*. Unpublished report to the National Trust for Scotland, Edinburgh, Scotland.

NPCA (2012). "Modified Maclean Scale". <http://www.npca.org.nz/index.php/news/84-general/226-rabbit-control-training.html>.

Rotheray, G.E. and Lyszkowski, R. (2012). *The Entomological Significance of Coastal Grassland on Canna and Sanday*. Unpublished report to the National Trust for Scotland, Edinburgh, Scotland.

SNH (2014). *Site Condition Monitoring Note, Canna and Sanday, Cycle 3*. Unpublished report. Inverness, Scotland: Scottish Natural Heritage.

Swann, R.L. (2002). *Canna Seabird Studies 2001*. Joint Nature Conservation Committee Report 327, Peterborough, UK.

Swann, B. (2004). 'Seabird Wreck, North West Scotland late summer 2004'. Seabird Group Newsletter 98: 12–14.

Swann, R.L. (2005) *Canna Seabird Studies 2005*. Joint Nature Conservation Committee Report, No. 474b, Peterborough, UK.

Swann, R.L. (2008). *Canna Seabird Studies 2008*. Joint Nature Conservation Committee Report No. 474e, Peterborough, UK.

Swann, R.L., Aiton, D.G., Call, A., Foster, S., Graham, A., Graham, K. and Young, A. (in press). *Canna Seabird Studies 2017*. Joint Nature Conservation Committee Report. Peterborough, UK.

Walsh, P.M., Halley, D.J., Harris, M.P., del Nevo, A., Sim, I.M.W. and Tasker, M.L. (1995). *Seabird Monitoring Handbook for Britain and Ireland: A Compilation of Methods for Survey and Monitoring of Breeding Seabirds*. Peterborough: JNCC, RSPB, ITE, Seabird Group.

Conservation gains and missed opportunities 15 years after rodent eradications in the Seychelles

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Abstract The Seychelles was one of the first tropical island nations to implement island restoration resulting in biodiversity gain. In the 2000s a series of rat eradication attempts was undertaken in the inner Seychelles islands which had mixed results. Three private islands with tourist resorts successfully eradicated rats: Frégate (2000), Denis Island (2003) and North Island (2005). Frégate Island was successful with the first eradication attempt whereas North and Denis Islands were initially unsuccessful, and both required second eradication operations. All three islands have developed conservation programmes including biosecurity, habitat rehabilitation, and species reintroductions, and have integrated nature into the tourism experience. Conservation actions, including rat and other invasive species eradications, on these three islands resulted in the creation of 560 ha of mammalian predator-free land, the reintroduction of seven populations of five globally threatened birds (GTB) and the safeguarding of two existing GTB populations and several reptile and invertebrate species. However, on these and many other islands in the Seychelles, the potential of this conservation "model", where island owners implement conservation programmes largely funded by the tourism businesses in collaboration with NGOs (Non Government Organisations), has not been fully realised. We review the rehabilitation on Frégate, Denis and North Islands from inception to the present, and assess factors that have facilitated the subsequent development of conservation programmes, the presence of receptive businesses and governmental/NGO/donor support and explore limitations on business-led island rehabilitation.

Keywords: eradications, invasive alien species, island conservation, rehabilitation Seychelles, tourism

INTRODUCTION

Islands harbour much of the world's endangered biodiversity (Kaiser-Bunbury, et al., 2010) and island species are very vulnerable to the impacts of Invasive Alien Species (IAS). Over the last 500 years, the majority of documented plant or vertebrate extinctions have occurred on islands (Tershy, et al., 2015). Causes include habitat modification and over exploitation; however, IAS have played a key role. In particular, invasive mammals such as rats (*Rattus* spp.) have been implicated in numerous bird extinctions, extirpations and population declines (Moors & Atkinson, 1984; Burger & Gochfeld, 1994; Hilton & Cuthbert, 2010) as well as reptile declines and impacts on other taxa (Towns, 1991; Harper & Bunbury, 2015; Thibault, et al., 2016).

The eradication of rats and other invasive mammals, often with concomitant habitat rehabilitation, was initially pioneered in New Zealand and other temperate areas but has become increasingly practiced in tropical regions, (Russell & Holmes, 2015; Russell & Broome, 2016). The understanding of the measures required to successfully execute mammal eradications on tropical islands has improved (Keitt, et al., 2015). In the late 1990s, the Seychelles was one of the first tropical island nations to implement rodent and multispecies eradications (Merton, et al., 2002)

The Seychelles archipelago in the Indian Ocean extends over an Exclusive Economic Zone of 1,374,000 km^2 (Fig. 1). The ancient "inner" islands are situated approximately 4^o S and 54^o E and are composed of continental rock, while the much more recently formed "outer" islands are formed from raised reefs and sand cays (Stoddart, 1984) scattered for approximately 1000 km to the south-west of the inner islands. The Seychelles have high endemism (Stoddart, 1984) and the inner islands are an Endemic Bird Area (EBA100); supporting 11 endemic species of bird (BirdLife International, 2017).

Since the human colonisation of the Seychelles in the late 18th century, IAS have caused range reductions, population declines, and extinctions of native species (BirdLife International, 2000). Alien predators are considered the most destructive species (Harper & Bunbury, 2015): most inner islands have had populations of black rat (*Rattus rattus*) and feral cat (*Felis catus*), and some have had brown rat (*R. norvegicus*). Only a few inner islands remained free of mammalian predators, and in the 1980s only four islands larger than 20 ha remained free of rats (Aride, Cousin, Cousine and Frégate), although feral cats were on Cousine and Frégate, and house mice (*Mus musculus*) on Aride and Frégate. Construction projects on islands resulted in the introduction of rats including black rat to Bird Island in 1968 and brown rat to Frégate in 1995.

Four Seychelles' bird species endemic to the inner islands were listed as Critically Endangered when at their lowest known population size: Seychelles magpie-robin (*Copsychus sechellarum*), Seychelles white-eye (*Zosterops modestus*), Seychelles paradise-flycatcher (*Terpsiphone corvine*), Seychelles scops-owl (*Otus insularis*). Four species were listed as Vulnerable: Seychelles warbler (*Acrocephalus sechellensis*), Seychelles fody (*Foudia sechellarum*), Seychelles kestrel (*Falco araea*) and Seychelles swiftlet (*Aerodramus elaphrus*) (BirdLife International, 2017) (Table 1).

Three species of bird that had been historically widespread (Gaymer, et al. 1969) became restricted to black rat-free islands: the Seychelles magpie-robin on Frégate (Gaymer, et al., 1969; Burt, et al., 2016), Seychelles warbler on Cousin (Komdeur, 2003), and Seychelles fody on Cousin, Cousine and Frégate (Vesey-Fitzgerald, 1940). The Seychelles white-eye had a small population in the uplands of Mahé and a larger population on Conception Island which had brown rats, but no black rats (Rocamora, 1997). Aride remained free of rats but did not retain

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Table 1 Number of island populations and threat categories of birds endemic to the Granitic Seychelles Endemic Bird Area (EBA) since 1994.

Species	Smallest documented number of populations	Current no of populations	IUCN Threat category since 1994	Year
Seychelles magpie-robin	1	5	Endangered	2005–2018
			Critical	1994–2005
Seychelles white-eye	2	4	Vulnerable	2016–2018
			Endangered	2005–2015
			Critical	1994–2005
Seychelles paradise-flycatcher	1	2	Critical	1994–2018
Seychelles scops-owl	1	1	Endangered	2004–2018
			Critical	1994–2004
Seychelles warbler	1*	5	Near threatened	2012–2018
			Vulnerable	1994–2012
Seychelles fody	3	6	Near threatened	2004–2018
			Vulnerable	1994–2004
Seychelles kestrel	1	2	Vulnerable	1994–2018
Seychelles swiftlet	3	3	Vulnerable	1994–2018
Seychelles black parrot	2	2	Vulnerable	2014–2018

* First reintroductions of Seychelles warblers undertaken prior to 1994 endangered species categories being applied.

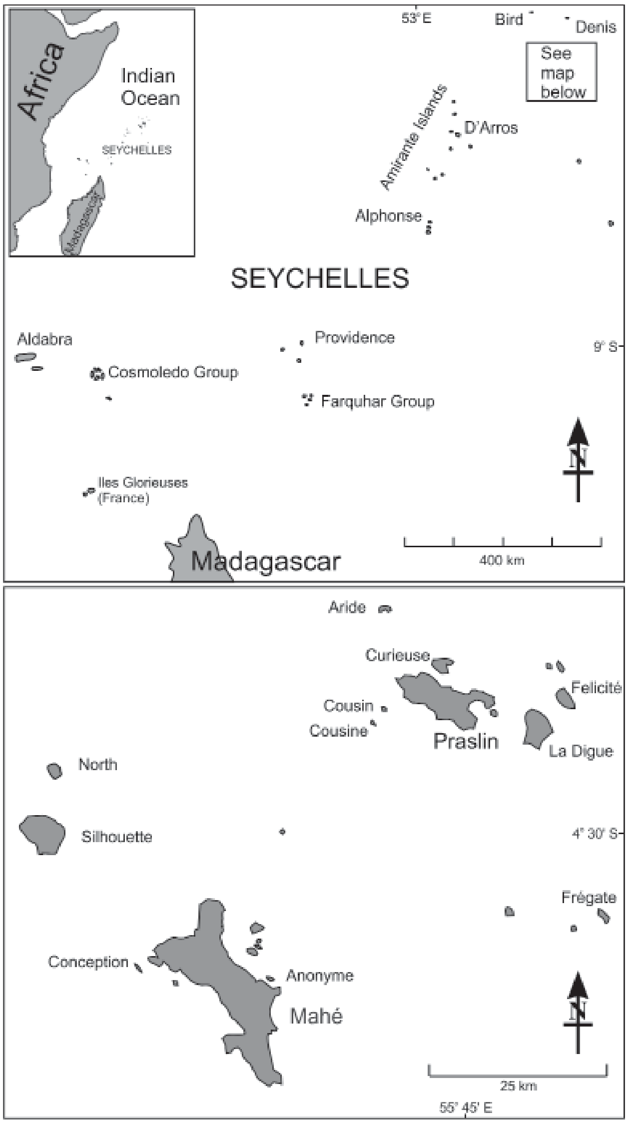

Fig. 1 The Seychelles showing islands mentioned in the text.

populations of endemic birds which may be attributed to forest loss and cat predation (eradicated in 1930s; Warman & Todd, 1984). The distributions of these endemic bird species suggest black rats were an important factor contributing to decline and extirpation of populations.

Initial conservation efforts focused on the purchase and protection of two rat-free islands (Cousin, 1968; Aride, 1973) by NGOs (Non Government Organisations). Successful attempts were made to reintroduce the Seychelles warbler to Aride and Cousine (Richardson, 2001; Komdeur, 2003). Attempts to introduce Seychelles magpie-robin to Cousin and Cousine met with success, but several introduction attempts to Aride were unsuccessful (Watson, 1978; Lucking & Ayrton, 1995).

Further progress was achieved with cat eradications on Frégate and Cousine during the 1980s (Rocamora & Henriette, 2015). Rodent eradications were not attempted in the Seychelles until 1996, when black rats were eradicated from Bird Island, and later in the early 2000s when a a series of rat and multispecies eradications were initiated that included privately owned Frégate, Denis and North Islands. Subsequent habitat rehabilitation and endemic bird reintroductions were implemented (Thorsen, et al., 2000; Merton, et al., 2002; Samways, et al., 2010).

This paper reviews the conservation programmes on three privately owned islands and the conservation outcomes.

ISLAND DESCRIPTIONS: NORTH, DENIS, AND FRÉGATE

North Island (Ile du Nord)

The native vegetation of North Island (201 ha) was replaced in the early 19th century by a coconut (*Cocos nucifera*) plantation, which was abandoned in the 1970s, and guano excavation left pits that are still present today. The island harboured black rats, cats, and feral cattle (*Bos taurus*). Hill (2002) identified the island as having a high rehabilitation potential; small enough to eradicate mammals, sufficiently isolated to manage reinvasion risk and a proportionally large coastal plateau that is likely to support rehabilitated forest suitable for Seychelles magpie-robins and Seychelles paradise-flycatchers (Currie, et al., 2003). The island was privately purchased in 1997 by an eco-tourism company, which opened an exclusive resort in 2005, with the intention to ultimately fund the rehabilitation of the island. Conservation is promoted through all aspects of the tourism operation, whereby guests and staff are educated and encouraged to participate in environmental activities such as guided hikes, presentations, and data collection.

Denis Island

Denis is a coralline island of 140 ha located 80 km north of the capital island of Mahé. Early descriptions mention abundant land tortoises and seabirds (Bradley, 1940). The original vegetation of the island, as described in 1773, was of open grassy areas and forest; probably *Pisonia grandis* (Stoddart & Fosberg, 1981). Extensive guano deposits indicated the historical presence of seabirds. The island has been altered profoundly, first through the cultivation of coconuts from around 1890 (Stoddart & Fosberg, 1981), then through guano extraction in the 1930s (Baker, 1963), followed by the replanting of coconuts in the 1940s. In 1975, a new owner built an airstrip and a small hotel and abandoned the coconut plantation. In 1998 the island changed ownership again and today it is managed as a luxury tourist resort with 30 villas.

Denis was considered a priority site for rehabilitation due to the large area of flat land conducive to rehabilitation, existing native woodland, and an owner who supported conservation, if black rats and feral cats were removed (Hill, 2002). The island appears to be in the natural range of some endemic birds as, in 2004, a Seychelles magpie-robin from Aride flew to Denis (Burt, et al., 2016) and in 2009 a Seychelles sunbird (*Cinnyris dussumieri*) flew from Bird Island to Denis (R. Bristol, pers. obs., 2009).

Frégate Island

The original vegetation of Frégate Island (219 ha) was removed to make way for spice and coconut plantations, that were abandoned in the 1980s, leaving coconut-dominated forest and several areas of the introduced tree sandragon (*Pterocarpus indicus*). The original vegetation is unknown; however, a few relict plants including *Pandanus balfourii* and *Euphorbia pyrifolia* survived on rocky glacis areas, indicating some of the vegetation that existed before plantations (J. Millett, pers. obs., 2000). The availability of canopy-forming sandragon forest and the absence of rats contributed to the survival of Seychelles fody, the last Seychelles magpie-robin population, and rich assemblages of reptiles, amphibians, and invertebrates including two single-island endemics: a beetle *Polposipus herculeanus*, and a snail *Pachnodus fregatensis* (Canning 2011b; Gerlach, 2006). Frégate was re-developed in 1995–1999 as an exclusive resort, and brown rats were accidentally introduced in 1995 at the time of hotel construction (Thorsen, et al., 2000; Merton, et al., 2002). Management of the island's biodiversity was initially not highly prioritised, but renewed interest in conservation was created through increased awareness amongst stakeholders over how nature can contribute to tourism. In 2003 the hotel appointed its own environment staff, who conducted biodiversity management and monitoring (J. Millett, pers. obs., 2003).

Vertebrate eradications

The islands of Frégate, Denis and North were amongst a series of islands that had rodents and /or cats eradicated between 1982 and 2005. In total six islands over 10 ha in size had black rats successfully removed, and three had brown rats successfully removed (Rocamora & Henriette, 2015). Some of the eradications were implemented in multi-island projects; however, the work was not planned as a phased programme. On the three islands central to this paper, the rodent eradications were, in part, motivated by business interests, and two required second attempts. The eradication of rodents has been an iterative (and at times faltering) process but was ultimately successful on most islands (Table 2).

Interest in eradicating rodents in the Seychelles was stimulated in 1995, when the introduction of brown rats to Frégate Island raised national and international concerns over the impact on Seychelles magpie robins (Merton, 1996). A proposal, led by the Ministry of Environment and

Table 2 Black rat, brown rat, mouse and cat eradications in the Seychelles.

Island	Size (ha)	Species	Invasion date	Eradication date	Method	Outcome	Prevention measures
Bird	101	Black rat	1960s	1995	Ground Application	Success	Medium/ Good
Frégate	219	Brown rat	1995	2000	Aerial Application	Success	Medium/ Good
		Mice	?	2000	Aerial Application	Success	
		Cats	?	1982	Ground Application	Success	
Curieuse	286	Black rat	?	2000	Aerial Application	Reinvaded	Poor
		Cats		2000	Ground Application	Success	
Denis	143	Black rat	?	2000	Aerial Application	Reinvaded	Poor/ None
		Cats	?	2000	Ground Application	Success	
Denis	143	Black rat	2001	2002	Ground Application	Success	Poor/Medium
		Mice	?				
North	201	Black rat	~1784	2003	Aerial Application	Failed	Poor/Medium
		Cats	?	2003	Ground Application	Success	
D'Arros	150	Brown rat	?	2003	Ground Application	Success	Good
		Cats	?	2003		Success	
		Mice	?	2003		Failed	
Anonyme	10	Black rat	?	2003	Ground Application	Success	Medium/Poor
North	201	Black rat	?	2005	Aerial Application	Success	Medium/ Good

Transport (MET), to eradicate rats and other mammals on several islands in a combined operation led to the eventual attempted eradication of black rats on Denis and Curieuse, brown rats on Frégate, and cats on Denis and Curieuse. House mice present on Frégate and Denis were not specific targets of eradication , but they were eradicated from the former during the operation.. The eradication operational costs on the private islands (Denis and Frégate) were financed by the island owners, and on Curieuse (state owned) it was funded by a grant from the Dutch Trust Fund (DTF) that also covered consultancy costs for the three islands (Merton, et al., 2002; United Nations, 2002; Rocamora & Henriette, 2015; John Nevill, pers. comm. 2018).

Rodent eradication operations commenced in June 2000 with two aerial applications of brodifacoum bait totalling 18 kg/ha applied to Denis with a nine day interval, and three aerial applications (23 kg/ha) to Frégate at five and 25 day intervals (Merton, et al., 2002). Areas that could not be covered through the aerial application, including buildings, work yards and hydroponic green houses were hand baited. The third application was in response to a lactating female rat trapped in an agricultural plot after the second application (J. Millett, pers. obs., 2000) after which no further rats were observed and the eradication on Frégate was successful. Cat eradication on Denis proceeded one week after the second rat-bait application using trapping and baiting with Compound 1080 (Merton, et al., 2002). The last cat on Denis was killed 14 months after the eradication started. On the same day the last cat was killed, black rats were confirmed as being present again and breeding on Denis (J. Millett, pers. obs., 2001). It was not possible to conclude if the population arose from survivors or reintroduction. However, given the short time duration between eradication and discovery and better understanding of factors influencing tropical island rodent eradications, eradication survival is likely (Rocamora & Henriette, 2015; Keitt, et al., 2015). Subsequently it was discovered the eradication attempt on Curieuse had also been unsuccessful (G. Climo, pers. comm., 2001), possibly due to reinvasion and/or survival (Rocamora & Henriette, 2015).

The owners of Denis Island decided to undertake a second eradication attempt to eradicate rats and mice. This proceeded with a ground-based operation in 2002, using brodifacoum poison in bait stations on a 40 m grid. Monitoring indicated that rats were killed quickly, but mice persisted for several weeks around the livestock farm where alternative food sources were available (G. Climo, pers. comm., 2002). Both species were eradicated successfully within two months.

A black rat eradication was attempted on North Island in 2003 with an aerial baiting operation using three aerial applications of brodifacoum. In March 2004 black rats were still present (G. Climo, pers. comm., 2004; Rocamora & Henriette, 2015). Cats were eradicated successfully at this time with a combination of poisoning with Compound 1080 and trapping. A second attempt to remove rats was made in 2005 with four aerial applications and a grid of bait stations on the whole plateau and in the vicinity of housing (Climo & Rocamora, 2006). In response to a rat being captured four days after the third application a fourth application was conducted (Climo & Rocamora, 2006) which ultimately resulted in the eradication of rats.

Not only mammals have proved to pose problems for endemic island species: introduced Indian myna birds (Acridotheres tristis) attack some native birds and compete for nest sites with Seychelles magpie-robin (Burt, et al., 2015; Feare, et al., 2017). An attempt to eradicate mynas on Frégate in 2000–2003 by shooting was unsuccessful (Millett, et al., 2005) but eradication succeeded using traps in 2011 (Canning, 2011a). Eradication on Denis Island using an avicide (Starlicide) and shooting commenced in 2000 but was unsuccessful. A subsequent attempt used trapping with follow-up shooting, which succeeded in 2015 (Feare, et al., 2017). On North Island, in 2006, an attempt to eradicate mynas with Starlicide was unsuccessful due to difficulties importing a rifle to start the shooting phase when poisoning had reduced the population to fewer than 100 birds; shooting was finally conducted in 2008-2009 but the population had recovered and was too numerous to be effective (Rocamora & Henriette, 2015). It was reattempted with a decoy trapping campaign from May 2016 to March 2017, followed by shooting. This reduced the population to three individuals by June 2018 with the eradication attempt ongoing (Havemann, pers. obs., 2018).

Overall experiences on Denis, North and Frégate indicate that sustained trapping programmes using small decoy traps located in areas frequented by foraging mynas, followed by shooting with an experienced hunter, is effective. Shooting as a standalone measure and using avicides appear to create aversion and have not worked well. Disruption or cessation of culling results in a population recovery (Millett, et al., 2005; Feare, et al., 2017).

BIOSECURITY

Biosecurity controls have been implemented on each of the three islands since undertaking rodent eradication. North Island has rigorous biosecurity with pre-departure inspections of all cargo on Mahé, inspections on arrival, the processing of cargo and baggage through a pest containment room, and fumigation and permanent bait stations locate close to landing areas, human habitation and beaches (North Island, 2015). Frégate Island has a rodent abatement protocol which includes cargo inspection and controls as well as permanent rat bait stations (Rocamora, 2015). There is also a rodent-proof fence around the harbour, made of steel mesh set in to the ground and topped with a smooth metal strip. However, maintenance of the structure has remained a challenge, especially where the ends of the fence meet the water and are influenced by wave action (J. Millett, pers. obs., 2018). Denis Island has a rodent prevention protocol (GIF, 2015). The protocol is focussed on rodent control with measures including baiting on boats, baiting arrival points and contingency measures to respond to an incursion: Denis Island still brings cargo to the island using a beach-landing barge, which increases reintroduction risk.

On all three islands, visiting vessels need to be in possession of a rat-free certificate which is obtained after a thorough check for rats on board prior to departure from Mahé. All of the protocols have been implemented voluntarily and devote greater effort to inspection and containment of rodents on the islands and less on loading controls at departure. Although all plans concentrate on rodent prevention, they are likely to be effective at reducing wider biosecurity risks.

FOREST REHABILITATION

Endemic birds rely on forest (Vega, 2005; Njoroge, 2002). Most other native vertebrates and invertebrates are also forest dwelling species; some have been able to adapt to gardens and plantations. Invertebrate densities and diversity on foliage tended to be higher for native trees, yielding greater food availability for species such as the Seychelles warbler and Seychelles paradise-flycatcher (Komdeur, 1991; Komdeur, 1992; Richardson, 2001; Hill, 2002). Accordingly, rehabilitation of native forest has been a prerequisite to restoring endemic bird populations. The original vegetation on Frégate, Denis and North Islands is uncertain but evidence from remnant species and vegetation of similar, less modified islands suggests that the original contained *Pisonia grandis* and other native coastal trees including *Thespesia populnea*, *Heritiera littoralis* and *Calophyllum inophyllum* (Hill, 2002). The objective of rehabilitation has been to create habitat for native species, not to recreate pre-human forest.

North Island

Vegetation rehabilitation started in 2001 with the removal of invasive plant species such as *Lantana camara*, planting of native species on the coastal plateaux (including *Terminalia catappa*, *Barringtonia asiatica*, *Heritiera littoralis*) and attempts to rehabilitate vegetation on the hills by planting *Pyrostria bibracteata, Dodonaea viscosa* and other robust native shrubs. By 2017, approximately 60 ha of the coastal plateau was a native-dominated forest with *T. catappa* and *C. inophyllum* being the most abundant species. The establishment of native species on the hills has been slower and more labour intensive with <2 ha restored. The current area of native-dominated woodland is approximately 30% of the island's total surface area.

Denis

In 2001, approximately 20 ha of coconut plantation that was naturally reverting to native forest dominated by *T. catappa*, was cleared of coconuts and planted with native tree species (Hill, 2002). In 2007–2008, 12.5 ha were rehabilitated with the aim of creating habitat for Seychelles paradise-flycatchers (Bristol, et al., 2009). The rehabilitation involved removing coconut, *Nephrolepis biserrata* fern and other introduced weeds and replanting with tree species including *Terminalia catappa*, *C. inophyllum, Thespesia populnea, Cordia subcordata, B. asiatica, Ficus lutea, Guettarda speciosa, Hernandia nymphaeifolia, H. littoralis, Ochrosia oppositifolia, Pandanus balfourii, P. grandis*, *Ficus reflexa, Hibiscus tiliaceus* and *Morinda citrifolia.* In 2013–14, a further 2.5 ha area was cleared of coconut and *Casuarina equisetifolia* and replanted with *C. inophyllum* and *Mimusops sechellarum* and ca. 18 ha of *T. catappa* woodland were weeded. The current area of native-dominated woodland is approximately 40 ha (Bristol, 2014), comprising 29% of the island's total surface area.

Frégate

A small amount of native tree planting was undertaken in the 1990s to benefit Seychelles magpie-robins and, in 1998, the hotel development used mostly native tree species for landscaping. A wilt disease caused by *Fusarium oxysporum* (Boa & Kirendall, 2004) killed all the Sandragon trees on Frégate in the early 2000s. Most of the sandragon forest was on the hills and these areas were replanted with native species, mostly *Ficus reflexa*, *F. lutea*, *Premna serratifolia* and *Tabernaemontana coffeoides*. Some further coastal areas were replanted with *T. catappa* and *Guettarda speciosa*, which has resulted in approximately 30 ha of native-dominated forest, comprising 15% of the island's total surface area. In addition, quite a lot of non-native forest was under-planted with native species but, unfortunately, the habitat rehabilitation is not well documented.

ENDANGERED SPECIES RECOVERY AND INTRODUCTIONS

The eradication of vertebrate predators on three islands with a total area of 560 ha, and associated improvement in forest habitat for native birds, has contributed to the recovery of several endangered bird species by increases in existing populations or by reintroducing populations (Table 3).

North Island

In 2007, 25 Seychelles white-eyes were introduced to North Island from Conception (Rocamora & Henriette-Payet, 2008). The population established and in 2017 was estimated at between 127 and 140 birds using direct census of groups with colour-ringed and unringed birds (Pietersen, 2017).

Denis Island

Four bird species have been introduced; 47 Seychelles fodies were translocated from Frégate in February 2004 (Bristol, 2005), and the current population is estimated at 600 individuals (van de Crommenacker, pers. obs.,

Table 3 Species conservation outcomes – an estimate of the number of endangered birds on rehabilitated private islands.

Species	Frégate Island	North Island	Denis Island	Other populations		Percentage of Seychelles population on Frégate, Denis & North Islands.
Seychelles magpie-robin	145[c]	0	76[a]	Cousin Cousine Aride	46 32 10	72
Seychelles warbler	209[e]	0	400[d]	Cousin Cousine Aride	320 210 1,850	21
Seychelles white-eye	200[c, g]	134[b]	0	Conception Mahé	3,140[g] 25-35[h]	50
Seychelles Paradise-flycatcher	0	0	*84*[a]	La Digue	ca. 400	21
Seychelles fody	1,182[f]	0	600[c]	Cousin Cousine Aride D'Arros	1,000 430 ca. 500 250	35

Data: Birdlife International, 2017 except: [a] Bristol & Gamatis, 2017; [b] Pietersen, 2017; [c] van de Crommenacker, pers. obs., 2017; [d] Lopera-Doblas, et al., 2015; [e] Gala, 2017; [f] Vega, 2005 ; [g] Rocamora & Henriette, 2015; [h] Rocamora, pers. obs., 2017.

2016) using a grid point count with 58 counting points located every 150 m. Fifty-eight Seychelles warblers were introduced from Cousin in 2004, and the most recent population estimate is 400 individuals (Lopera-Doblas, et al., 2015; van de Crommenacker, pers. obs., 2016). Twenty Seychelles magpie-robins were introduced in June 2008, 16 from Frégate and four from Cousin (Burt, et al., 2016), and the population in June 2017 was 76 individuals, estimated by monitoring colour ringed birds (Bristol & Gamatis, 2017). Twenty-three Seychelles paradise-flycatchers were introduced from La Digue in November 2008 and the population in June 2017 was 84 individuals, surveyed in a direct count Bristol & Gamatis, 2017).

Frégate Island

The population of Seychelles magpie-robin prior to conservation efforts was very small, with as few as 39 in 2000 (Burt, et al., 2016). Habitat for this territorial species was limiting (López-Sepulcre, et al., 2010). Increased habitat area and quality allowed the population to rise to 137 by 2015 (Burt, et al., 2016) and approximately 145 in 2017. This was estimated by an ongoing programme to colour-ring as many birds as possible to allow identification in the field. Then, by searching the whole island for presence of birds and their nest locations, group associations and behaviour, a territory map was constructed along with a status list with the identity of all birds within each territory (van de Crommenacker, pers. obs., 2017). The Seychelles fody population was estimated to have a population of 1,182 using mark and re-sight methods (Vega, 2005).

Frégate was considered suitable to reintroduce Seychelles white-eyes because of the abundant fruiting trees, including the non-native cinnamon (*Cinnamon vernum*). Reintroduction was undertaken between 2001 and 2003, with 37 birds from Conception (Henriette & Rocamora, 2011). The most recent estimate is at least 200 individuals, based on point counts (van de Crommenacker, pers. obs., 2017). The habitat suitability for Seychelles warbler was investigated (Hammers & Richardson, 2011) and found to be suitable. Accordingly, 59 individuals were translocated from Cousin in 2011 (Wright, et al., 2014); the population was estimated to be at least 209 individuals in 2017 (Gala, 2017).

Other species appear to have benefitted from island rehabilitation, including the endemic beetle *Polposipes herculeanus* which showed a dramatic decline between 1995 and 2000, probably due to rat predation, but appears to have subsequently recovered (Lucking & Lucking, 1997; Canning, 2011b).

FACTORS THAT INFLUENCED ISLAND REHABILITATION

The Seychelles endemic birds' crisis in the 1970s and 1980s resulted in interventions by international organisations including the Royal Society for Protection of Birds (RSPB), International Council for Bird Preservation (ICBP), BirdLife International, and the Royal Society for Nature Conservation (RSNC), initially by direct funding and deployment of staff and later through the establishment and support of the local NGOs Nature Seychelles and Island Conservation Society (ICS). The investment, at its height, contributed several hundred thousand British pounds each year, and facilitated the involvement of technical expertise from New Zealand in an advisory role. The potential benefits of rodent eradication for tourism, farming and nature inspired island owners to finance eradications. At the same time a proactive approach was taken by the Seychelles government which wanted to promote eradication programmes and donor funding was available. International funding to help finance eradications and reintroduction operations was obtained by NGOs and Government through Global Environment Facility, Dutch Trust Fund and Fonds Français pour l'Environnement Mondial, among others.

As such, an enabling policy context (where a national biodiversity plan was in place and being implemented), international support, private sector interest, motivators with a "can do" approach, and finance all came together to facilitate change. At the time, the risk of failure in tropical rodent eradications was not estimated (Keitt, et al., 2015), and was therefore not perceived as a constraint.

The results of island rehabilitation, in particular rodent eradications, have not only been sustained but enhanced by hotel businesses who value it as part of their tourism product. Each island has a conservation manager and a small team of conservation staff and volunteers who implement biodiversity monitoring, biosecurity, habitat rehabilitation and education and awareness activities, including activities for hotel guests. These businesses have been able to access funds to support conservation management including directing Corporate Social Responsibility Tax (CSRT) to conservation programmes, donations from clients and paying volunteers. Each island has used independent approaches and methods to sustain this work, and cooperation improved when North Island, Denis Island and Frégate Island began working in partnership with the Green Islands Foundation (GIF), which was established in 2005 with the objective of improving cooperation and conservation work on islands. GIF has been able to assist with coordinating conservation programmes, apply for funds and manage projects on behalf of island conservation programmes and, importantly, act as a representative and advocate at national meetings related to the environment.

Limitations to biodiversity conservation on private islands

Recent years have seen a cessation of rodent eradications and species reintroductions, with no rodent eradications undertaken since 2010 (Rocamora & Henriette, 2015). International partners progressively reduced support to the Seychelles before ceasing funding, mainly due to the reduced threat to endemic birds and Seychelles being no longer considered a low-income country. National policy still supports island rehabilitation and species conservation (Nevill, et al., 2014), but specific actions are not being proactively promoted. Major donors' priorities have shifted and the national project portfolio is dominated by climate change adaptation and energy (Programme Coordination Unit, 2017). In principle, funding is available for island rehabilitation, but it is not being requested by the government.

Very few native animal species that are not birds have been introduced to Frégate, Denis or North Islands. Moreover, predator-free North Island's potential to support populations of endangered species has not yet been realised. One of the reasons is that habitat rehabilitation requires a number of years to produce a canopy-forming forest that is suitable for endemic birds and the forest has only recently become suitable for species such as Seychelles black paradise flycatcher (Bristol, 2017). Moreover, consensus building for species reintroduction takes time and may be influenced by views that are not necessarily evidence-based or pro-conservation.

There are several inner and outer islands with conservation potential for rehabilitation that have not been subject to eradication of invasive predators. For example, the inner island Félicité has potential for rehabilitation (Hill, 2002) and is currently suitable for the reintroduction of Seychelles paradise flycatchers, a species tolerant of rats, (Bristol, 2017) and potentially more endemic species if cats and rodents were eradicated. Several outer islands already have hotels that support conservation programmes run by ICS and four islands are proposed as protected areas (UNDP, 2016). Rodent and cat eradications, if feasible, are likely to be beneficial for seabird populations and other biodiversity and well as tourism (Millett, et al., 2016).

Globally, many island-based businesses have nature orientated tourism, and many islands have undertaken rat eradications. However, comparable examples, whereby tourism businesses have undertaken, often with the support of local NGOs, invasive species eradication, habitat rehabilitation and endangered species reintroductions have not been observed in other island regions such as the Pacific or Caribbean. There may be reasons for this: for example, islands may have complex traditional ownership, species that are conservation priorities may not be suited to smaller islands or eco-tourism may be less valued by tourism sectors in other regions.

CONCLUSION AND RECOMMENDATIONS

The contribution of private islands to national conservation objectives is substantial, with 560 ha of predator-free land on three islands supporting nine populations of five species of globally threatened birds. The self-financing private sector, the enabling role of the Seychelles government and the contribution played by NGOs in facilitation, information exchange and advocacy are important. Whilst the priorities and contributions of international networks and some NGOs have focussed on other areas of work, others, notably GIF and ICS, are still attempting to increase networking and cooperation between these and other islands.

Lessons and recommendations for future work are:

1. Local NGOs should work more closely together and with business to improve knowledge exchange, build capacity, and enhance rehabilitation programmes;
2. Opportunities for the restoration of species on predator-free islands should be taken to the full, notably on North Island;
3. Develop a shared biosecurity facility on Mahé to reduce the risk of invasive species reintroduction;
4. Promote mammal eradications and habitat rehabilitation on suitable inner islands including Félicité, the proposed protected areas in the outer islands of the Seychelles;
5. Government should translate national policies including the NBSAP into implementation plans for species and sites;
6. The allocation of resources for island rehabilitation should be advocated by the government of Seychelles to international donors for large-scale national projects;
7. The approach adopted by Frégate, North and Denis Islands should be promoted as good practice internationally by organisations that facilitate collaboration and information sharing between small island states.

REFERENCES

Boa, E. and Kirendall, L. (2004). *Sandragon Wilt Disease, Seychelles.* Unpublished report for Government of Seychelles, Mahé, Seychelles: CABI.

BirdLife International. (2000). *Threatened Birds of the World*. Barcelona and Cambridge, UK: Lynx Edicions and BirdLife International.

BirdLife International. (2017). 'Country Profile: Seychelles'. <www.birdlife.org/datazone/country/seychelles>. Accessed 25 July 2017.

Baker, B.H. (1963). 'Geology and mineral resources of the Seychelles Archipelago'. *Memoir of the. Geological Survey Kenya* 3: 1–140.

Bradley, J.T. (1940). *The History of Seychelles*., Victoria, Seychelles: Clarion Press.

Bristol, R. (2005). 'Conservation introductions of Seychelles fody and warbler to Denis Island, Seychelles'. *Re-introduction News* 24: 35–36.

Bristol, R., Barra, A. and Rose, M. (2009). *Denis Island Habitat Rehabilitation Final Report*. Unpublished report for Green Islands Foundation, Mahé, Seychelles: Darwin Initiative Project 15-009.

Bristol, R. (2014). *Denis Island Habitat Rehabilitation Plan 2015–2019*. Unpublished report for Green Islands Foundation, Mahé, Seychelles.

Bristol, R. (2017). *Proposal to Translocate Seychelles Paradise Flycatchers to Félicité and Curieuse Islands*. Unpublished Report for Government of Seychelles. Mahé, Seychelles: Darwin Initiative project 23-006.

Bristol R. and Gamatis I. (2017). *Seychelles Paradise-flycatcher Population Census on Denis Island, June 2017.* Unpublished report for Green Islands Foundation, Mahé, Seychelles.

Burger, J. and Gochfeld, M. (1994). 'Predation and Effects of Humans on Island-nesting Seabirds'. In: D.N. Nettleship, J. Burger, M. and Gochfeld (eds) *Seabirds on Islands: Threats, Cases Studies and Action Plans*, pp. 39–67. Cambridge: BirdLife International.

Burt, A.J., Gane, J., Olivier, I., Calabrese, L., de Groene, A., Liebrick, T., Marx, D. and Shah, N. (2016). 'The history, status and trends of the endangered Seychelles magpie-robin *Copsychus sechellarum*'. *Bird Conservation International* 1: 1–19.

Canning, G. (2011a). 'Eradication of the invasive common myna (*Acridotheres tristis*) from Frégate Island, Seychelles'. *Phelsuma* 19: 43–53.

Canning, G. (2011b). 'Population assessment of the Frégate Island giant tenebrionid beetle (*Polposipus herculeanus*)'. *Phelsuma* 19: 69–78.

Climo G. and Rocamora G. (2006). *The Successful Eradication of Black Rats from North Island (Seychelles) in September 2005 and Proposed Strategies to Reduce the Risks of Reinvasions*. Unpublished report. *Réhabilitation des Ecosystèmes Insulaires* (*FFEM*) Project. Seychelles: Island Conservation Society and North Island.

Currie, D., Bristol, R., Millett, J., Hill, M., Bristol, U., Parr, S.J. and Shah, N.J. (2003). 'Habitat requirements of the Seychelles black paradise flycatcher *Terpsiphone corvina*: A re-evaluation of translocation priorities'. *Ibis* 145: 624–636.

Feare, C.J., van der Woude, J., Greenwell, P., Edwards, H.A., Taylor, J.A., Larose, C.S., Ahlen, P.A., West, J., Chadwick, W., Pandey, S., Raines, K., Garcia, F., Komdeur, J. and de Groene, A. (2017). Eradication of common mynas *Acridotheres tristis* from Denis Island, Seychelles. *Pest Management Science* 73(2): 295–304.

Gala, C. (2017). 'Exploring the Effect of an Invasive Lepidopteran on an Endemic Insectivorous Bird'. MSc thesis. University of East Anglia: Faculty of Biological Sciences.

Gaymer, R., Blackman, R. A. A., Dawson, P. G., Penny, M. and Penny, C. M. (1969). 'The endemic birds of the Seychelles'. *Ibis* 111: 157–176.

Gerlach, J. (2006). *Terrestrial and Freshwater Mollusca of the Seychelles Islands*. Leiden: Backhuys.

Green Island Foundation. (GIF) (2012): *Denis Island Rodent Protocol*. Seychelles: Green Islands Foundation.

Harper, G. and Bunbury N. (2015). 'Invasive rats on tropical islands: Their population biology and impacts on native species'. *Global Ecology and Conservation* 3: 607–627.

Hammers, M. and Richardson, D.S. (2011). *Assessment of the Suitability of Frégate Island for Seychelles Warblers* Acrocephalus sechellensis *2011*. Unpublished report. University of Groningen and University of East Anglia.

Henriette, E. and Rocamora, G. (2011). 'Comparative use of three methods for estimating the population size of a transferred island endemic: the endangered Seychelles white-eye *Zosterops modestus*'. *Ostrich: Journal of African Ornithology* 82(2): 87–94.

Hill, M. J. (2002). 'Biodiversity surveys and conservation potential of inner Seychelles islands'. *Atoll Research Bulletin* 495: 1–272.

Hilton, G. M. and Cuthbert, R. J. (2010). 'The catastrophic impact of invasive mammalian predators on birds of the UK Overseas Territories: a review and synthesis' *Ibis* 152: 443–458.

Kaiser-Bunbury, C.N., Traveset, A. and Hansen, D.M. (2010). 'Conservation and restoration of plant–animal mutualisms on oceanic islands'. *Perspectives in Plant Ecology, Evolution and Systematics* 12(2): 131–143.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S. and Millett J. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

Komdeur, J. (1991). 'Cooperative Breeding in the Seychelles Warbler'. PhD thesis. Cambridge: University of Cambridge.

Komdeur, J. (1992). 'Importance of habitat saturation and territory quality for evolution of cooperative breeding in the Seychelles warbler'. *Nature* 358: 493–495.

Komdeur, J. (2003). 'Adaptations and maladaptations to island living in the Seychelles warbler'. *Ornithological Science* 2(2): 79–88.

Lopera-Doblas, L., McClelland, S., van der Woude, J. and van de Crommenacker, J. (2015). *Seychelles Warbler Population Census on Denis Island August–October 2015*. Unpublished report. Seychelles: Green Island Foundation.

López-Sepulcre, A., Kokko, H. and Norris, K. (2010). 'Evolutionary conservation advice for despotic populations: Habitat heterogeneity favours conflict and reduced productivity in Seychelles magpie-robins'. *Proceedings of the Royal Society of London Series B* 277: 3477–3482.

Lucking, R.S. and Ayrton, V. (1995). *The Seychelles Magpie Robin Recovery Plan. Aride Translocation: Summary Report, November 1992–November 1994*. Unpublished report. Mahé, Seychelles: BirdLife International.

Lucking, R. and Lucking, V. (1997). 'Preliminary results of monthly surveys of *Polposipus herculeanus'*. *Phelsuma* 5: 5.

Merton, D. (1996). *Frégate Island Rat Crisis – Implications, Recommendations, Eradication Plan, Indicative Budgets 1996 and 1997*. Republic of Seychelles: Unpublished report. BirdLife International and Division of Environment, Ministry of Environment and Transport.

Merton, D., Climo, G., Laboudallon, V., Robert, S. and Mander, C. (2002). 'Alien mammal eradication and quarantine on inhabited islands in the Seychelles'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 182–198. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Millett, J., Climo, G. and Shah, N.J. (2005). 'Eradication of common mynah *Arcidotheres tristis* population in the granitic Seychelles: Successes, failures and lessons learned'. *Advances in Vertebrate Pest Management* 3: 169–183.

Millett, J, Bristol, R., Feare, C., Duhec, A., Risi, M., Nogués, P, Morgan, M. and Blumel, J. (2016). *Seabird Management Plans for the Outer Islands of the Seychelles*. Unpublished report. Mahé, Seychelles: Island Conservation Society.

Moors, P.J. and Atkinson, I.A.E. (1984). 'Predation on seabirds by introduced animals, and factors affecting its severity'. *ICBP Technical Publication* 2: 667–690.

Nevill, J., Prescott, J., Shah, N.J. and Jeremie M. (2014). *Seychelles Biodiversity and Action Plan 2015 – 2020*. Seychelles: Government of Seychelles.

Njoroge, P. (2002). 'The Seychelles Magpie-robin *Copsychus sechellarum*: Territory Quality, Demography and Conservation of a Threatened Species'. PhD thesis. UK: University of Reading.

North Island (2015). *Manual for the Conservation Work, Environmental Management and Sustainability Efforts of North Island*. Seychelles: Wilderness Collection.

Pietersen, D.W. (2017). *Assessment of the Seychelles White-eye, Zosterops modestus Newton 1867, Population on North Island, Seychelles. North Island Annual Report 2017.* Seychelles: Wilderness Collection.

Programme Coordination Unit. (PCU) (2017). 'Ongoing Projects'. <http://www.pcusey.sc/index.php/pcu-projects/ongoing>. Accessed 4 July 2017.

Richardson, D. S. (2001). S*pecies Conservation Assessment and Action Plan: Seychelles Warbler (Timerl Dezil)*. Unpublished report. Norwich: Nature Seychelles and University of East Anglia.

Rocamora, G. (1997). *Rare and Threatened Species, Sites and Habitats Monitoring Programme in Seychelles: Monitoring Methodologies and Recommended Priority Actions.* Unpublished Report. Seychelles: Government of Seychelles.

Rocamora, G. and Henriette-Payet, E. (2008). Conservation introductions of the Seychelles White-eye on predator-free rehabilitated islands of the Seychelles archipelago, Indian Ocean. In: Soorae, P. (ed.) Global Reintroduction Perspectives. Reintroduction case studies from around the globe, pp. 121–129. Gland, Switerland: IUCN.

Rocamora G. (2015). *Biosecurity Protocols for the Transportation of Vessels, Cargo and People Between Islands, with Special Reference to Protected Areas and Islands of High Biodiversity Value.* Report for project 'Mainstreaming Prevention and Control Measures for IAS into Trade, Transport and Travel across the Production Landscape'. Seychelles: Government of Seychelles-UNDP-GEF.

Rocamora, G. and Henriette, E. (2015). *Invasive Alien Species in Seychelles: Why and How to Eliminate Them? Identification and Management of Priority Species.* Seychelles: Island Biodiversity & Conservation centre, University of Seychelles; Mèze: Biotope Editions, and Paris: Muséum National d'Histoire Naturelle (Inventaires & Biodiversité series).

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. Biological Conservation 185: 1–7.

Russell, J.C. and Broome, K.G. (2016). 'Fifty years of rodent eradications in New Zealand: Another decade of advances'. *New Zealand Journal of Ecology* 40(2): 197–204.

Samways, M.J., Hitchins, P.M., Bourquin, O. and Henwood, J. (2010). *Restoration of a tropical island: Cousine Island, Seychelles.* Biodiversity and Conservation 19: 425–434.

Stoddart, D.R. (1984). *Biogeography and Ecology of the Seychelles Islands.* Monographiae Biologicae Volume 55. The Hague/Boston/ Lancaster: Dr. W. Junk Publishers.

Stoddart, D.R. and Fosberg, F.R. (1981). 'Bird and Denis Islands, Seychelles'. *Atoll Research Bulletin* 252: 1–46.

Tershy, B.R., Shen, K., Newton, K.M., Holmes, N.D. and Croll, D.A. (2015). 'The importance of islands for the protection of biological and linguistic diversity'. *Bioscience* 20: 1–6.

Thibault, M., Brescia, 1F., Jourdan, H. and Vidal. E. (2016). 'Invasive rodents, an overlooked threat for skinks in a tropical island hotspot of biodiversity'. *New Zealand Journal of Ecology* 41(1): 74–83.

Thorsen, M., Shorten, R., Lucking, R. and Lucking, V. (2000). 'Norway rats (*Rattus norvegicus*) on Frégate Island, Seychelles: The invasion; subsequent eradication attempts and implications for the island's fauna'. *Biological Conservation* 96: 133–138.

Towns, D.R. (1991). 'Response of lizard assemblages in the Mercury Islands, New Zealand, to removal of an introduced rodent: The kiore (*Rattus exulans*). *Journal of the Royal Society of New Zealand* 21: 119–136.

United Nations (2002). 'Seychelles Country Profile'. <www.un.org/esa/agenda21/natlinfo/wssd/Seychelles.pdf>. Accessed 4 July 2017.

UNDP. (2016). *Seychelles' Protected Areas Finance Project: Project Document*. GEF ID 5485 / Project ID 00095320. Seychelles: United Nations Development Programme.

Vega, L.B. (2005). 'The Ecology and Conservation of the Seychelles Fody or Toc-toc *Foudia sechellarum*'. PhD thesis. UK: University of Reading.

Vesey-Fitzgerald, D. (1940). 'The birds of the Seychelles 1. The endemic birds'. *Ibis* 4: 482–489.

Watson, J. (1978). *The Seychelles Magpie Robin. WWF Project 1590: Endangered land birds, Seychelles*. Final Report 1(a). Gland, Switzerland: World Wildlife Fund. Mimeo.

Warman, S. and Todd, D. (1984). 'A biological survey of Aride Island Nature Reserve, Seychelles'. *Biological Conservation* 28: 51–71.

Wright, D.J., Shah, N.J. and Richardson, D.S. (2014). 'Translocation of the Seychelles warbler *Acrocephalus sechellensis* to establish a new population on Frégate Island, Seychelles'. *Conservation Evidence* 11: 20–24.

Eradication of invasive animals and other island restoration practices in Seychelles: achievements, challenges and scaling up perspectives

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Abstract In recent decades Seychelles has accumulated extensive experience in the management of invasive species and other island restoration practices. Non-government organisations (NGOs), governmental, parastatal and private stakeholders have conducted successful programmes to control and eradicate invasive animals and plants, particularly on small islands of high biodiversity value. Biosecurity protocols have been implemented to prevent (re)infestations. With at least 50 vertebrate populations (33 mammal, 16 bird and one reptile) from 14 different species successfully eradicated, Seychelles is the third country in the world after Australia and the USA for invasive vertebrate eradications from tropical islands, and the seventh when considering all countries. Twenty-four islands have benefited from invasive vertebrate eradications and other ecosystem restoration processes to create refuges for native biodiversity. About 470 ha of woodland have been rehabilitated through replanting and recovery of native vegetation, and at least 36 successful island translocations of native birds and reptiles have been conducted. This includes 16 conservation introductions or reintroductions of six endemic land birds (all but one threatened), two of a terrapin species and 18 of Aldabra giant tortoises. Recovery of native species and natural recolonisations have occurred on islands where invasive predators have been removed. As a result, four globally threatened endemic land birds have been down-listed in the IUCN Red List and dozens of other native species including seabirds, land birds, reptiles, invertebrates and plants have also benefited. Future challenges include increasing the proportion of the country's land area free of rats and cats from 3.9% to potentially 15.4%, mainly in the outer islands, and 50% in the long term if Aldabra and Cosmoledo are considered. Factors limiting future eradications and translocations are discussed. Alternative conservation approaches such as 'mainland-islands' are recommended for large islands, and the development of partnerships with nature-based tourism is encouraged to help fund further restoration.

Keywords: ecosystem recovery, habitat rehabilitation, invasive birds, invasive mammals, reintroductions, species recovery, species translocations, vegetation restoration

INTRODUCTION

The Republic of Seychelles comprises 115 main islands totalling 445 km^2 of land area within a marine Exclusive Economic Zone of 1,374,000 km^2. These are classified into the 'inner islands' archipelago, of granitic substrate (ca. 45), and the remote, coralline 'outer islands' (c.70) to the south and south-east, that include the Amirantes, Providence-Farquhar and Aldabra groups (Fig 1). Aldabra atoll, a nature reserve and World Heritage Site, represents about one third of the country area.

In recent decades, the restoration of small islands has been an effective conservation tool in Seychelles to create sanctuaries for native biodiversity (Rocamora, 1997; Nevill, 2001; Shah, 2001; Merton, et al., 2002; Shah, 2006; Asconit & ICS, 2010; Rocamora, 2010a; Samways, et al., 2010b; Nevill, 2011). This has been achieved by eradicating or controlling invasive alien predators and competitors. Native habitats have been restored by eliminating invasive alien plants and replanting native vegetation. Globally threatened species of endemic birds and other native wildlife have been translocated to these rehabilitated islands, contributing to their subsequent recovery (Kömdeur & Pels, 2005; Richardson, et al., 2006; Rocamora & Henriette-Payet, 2008; Shah, 2008). This paper updates the inventory of island restoration achievements in Seychelles documented in Rocamora (2015) and discusses future perspectives and challenges. It considers only actions for nature conservation purposes and excludes eradications of invasive species for agricultural (two declared; National Biosecurity Agency, pers. comm. 2017) or public health purposes. Names of islands are the ones normally used by the islanders as listed in the Constitution of Seychelles, with the exception of 'Ile du Nord' also referred to as 'North Island'.

MATERIALS AND METHODS

Rocamora (2015) used information from publications (Beaver & Mougal, 2009; Nevill, 2009), internal reports and newsletters, and unpublished information from personal knowledge, to construct a database recording all attempts made in Seychelles to eradicate vertebrate populations. For each eradication attempt the database records: island name, area, animal species, year(s), methods used, and the final outcome of the overall eradication programme but not to the immediate result of each method employed. This information was checked in 2014 and made consistent with the Database of Island Invasive Species Eradications (DIISE) managed by Island Conservation, to allow comparison of Seychelles' performance in eradication with that of other countries (DIISE, 2017). This information base was updated in 2017. No new eradication attempts have taken place since 2014. One operation formerly classified as control was re-classified as 'eradication' as the target species, the crested-tree lizard (*Calotes versicolor*), was eradicated. The status of the five operations that were ongoing in 2015, and finalised by 2017, was updated.

Rocamora (2015) also gathered information on the area of natural habitat rehabilitated (from reports, or estimates by island owners, managers or conservation staff) as a result of removing invasive plants and propagating and planting native vegetation. He also documented translocations of native species that occurred in Seychelles, i.e. reintroductions to islands where the species was formerly present, 'conservation introductions' to islands outside a species' known historical range (IUCN, 2013), and historical introductions or reintroductions of Aldabra giant tortoises (*Aldabrachelys gigantea*). For each translocation he recorded: species, island, year, type of translocation (reintroduction, conservation introduction), and outcome. He then analysed how island restoration practice has developed in Seychelles, together with nature-

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based tourism, and how this has benefited the conservation of native biodiversity.

RESULTS

Eradication of introduced predators and competitors: the first step to ecosystem recovery

By September 2017, 68 attempts to eradicate invasive animals had been made on 24 islands. Three of these operations are still in progress (Table 1).

Most island eradications conducted in Seychelles have targeted mammals (44 attempts, 68%) and birds (22 attempts, 29%) on 22 islands of at least 10 ha, plus two mammal eradications on two islets smaller than 1 ha. In five of the 65 completed eradications, species that were not the main target (feral cats and barn owls) also disappeared following the removal of rats and on three occasions island populations of feral goats and chickens died out following control. Of the remaining 55 eradication attempts completed (excluding the two small islets), 40 (72.7%) succeeded and 15 (27.3%) had a failed outcome (i.e. survival or recolonisation before the island could be certified pest-free; see DIISE, 2017). When including the rat eradications on two islets (< 1ha), success rate is 73.7% and failure 26.3% (n=57).

By the end of 2017, 50 alien vertebrate (33 mammal, 16 bird and one reptile) populations had been eradicated from islands in Seychelles. One operation targeting common myna (*Acridotheres tristis*) on Ile du Nord is almost finished and one (the ring-necked parakeet, *Psittacula krameri*, on Mahé) is in the final phase of monitoring. Fig. 2 lists the 14 species of vertebrates eradicated from islands in Seychelles (plus one yet to be confirmed) and gives the outcomes of eradication attempts. Success rates of eradication attempts vary: 33% for house mouse, 56% for common myna, 57% for feral goat, 75% for black rat (*Rattus rattus*) and brown rat (*R. norvegicus*) (excluding the two islets), and 100% for other species.

Domestic pigs (*Sus scrofa*) and cows (*Bos taurus*) were also removed from Cousine and Ile du Nord. These were small numbers: some of the animals were not completely wild and may have still depended on supplementary food from humans, so they were easy to catch, and it is unclear if some of them were reproducing in the wild (Samways, et al., 2010a; Bruce Simpson/North Island Ltd; pers. comm.; Victorin Laboudallon, pers. comm. 2015). Feral cats (*Felis catus*) died out on Picard after the 1970s, with no control or eradication programme involved. These pig, cow and cat cases are not included in the calculations as eradications, but we did include the reported eradication of goats on Aride by shooting before 1920 (Warman & Todd, 1984), the removal of feral goats from the Aldabra atoll islands of Polymnie and Ile Esprit in the 1970s in response to localised control (Nancy Bunbury/SIF, pers. comm.), and the extinction of feral chickens on Desnoeufs in 2007 by local staff for consumption (Roland Nolin, pers. comm.).

The numbers of eradication attempts and success rates have varied over time (Table 2): 73% (over 11 attempts) before 1995, 64% (over 28 attempts) during 1995–2004, and after 2004 up to 89% (over 18 attempts finalised by 2017).

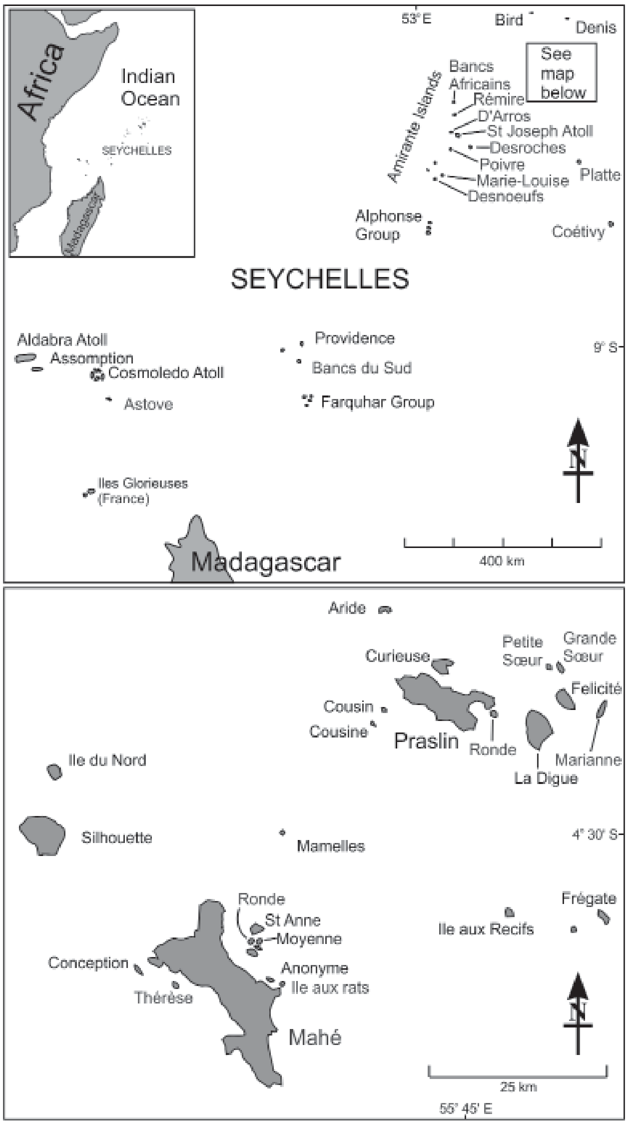

Fig. 1 Islands of the Republic of Seychelles.

Removal of invasive alien plants and replanting native vegetation

Control of invasive plants and habitat rehabilitation has been important for restoring ecosystems and protecting native biodiversity (Table 3). Significant areas (over 60 ha) have been rehabilitated since the 1990s on Praslin (National Park) and Mahé (Morne Seychellois National Park). Most invasive plant control and native species replanting activities have taken place on small and medium sized granitic islands, as part of programmes to restore

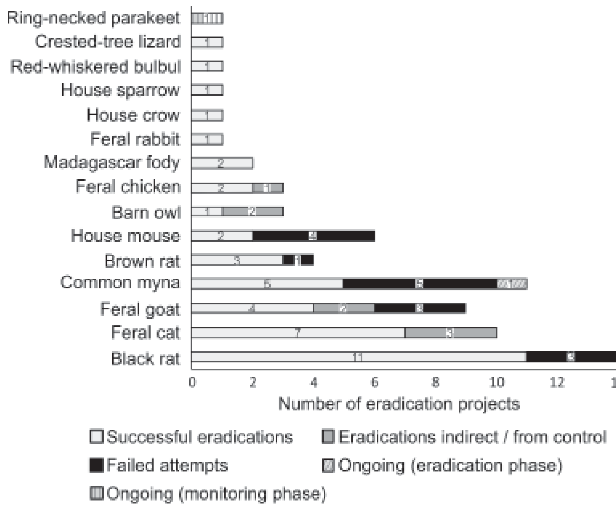

Fig. 2 Number of eradication attempts (n = 68), and success outcomes for the 15 species of invasive vertebrates targeted in Seychelles. Species are listed in increasing order of successful attempts. Success or failure refers to the final outcome of the operations (see text).

Table 1. Attempts to eradicate invasive alien animals from islands in Seychelles. Total number of attempts = 68 (65 completed; 3 in progress). Successful attempts (in **bold**) = 50 (42 direct; + 5 indirect eradications induced by rat removal and 3 from control, in brackets). Success or failure refers to the outcome of the operation. 'Failed' attempts involved animals that either survived or immediately reinvaded after a technically successful eradication phase. Only the active removal of individuals that had established a substantial wild breeding population is considered a genuine eradication attempt. Removal of small numbers of wild or semi-feral domestic animals (cattle, pigs, chickens), or populations of the same that died out naturally are not included here. * = eradications almost finished or in monitoring phase, to be confirmed in 2018. Occ. inc. = occasional incursions. Species and islands are listed in chronological order of their first eradication.

Outer Islands chronological order of eradications	Polymnie, Aldabra	Ile Esprit, Aldabra	Picard, Aldabra	Malabar, Aldabra	D'Arros	Desnoeufs	Grande Ile, Cosmoledo	Grand Polyte, Cosmoledo	Petit Polyte, Cosmoledo	Grande Terre, Aldabra	Assomption	Total attempts	Successful outcomes
Area (ha)	475	51	940	2,680	140	35	143	21	<1	11,610	1,171		
Feral goat *Capra hircus*	(died out by **1976** after control)	(died out by **1976** after control)	**1993–95**	**1993–95** 1987–88 failed	–	–	–	–	–	**2007–12** 93–97 failed 87–88 failed	–	8	3 (+ 2 control)
Feral cat *Felis catus*	–	–	(died out after1970s)	–	**2003**	–	**(2008)**	**(2007)**	–	–	–	3	1 (+ 2 indirect)
Feral chicken *Gallus gallus*	–	–	–	–	–	(died out by **2007** after harvest)	–	–	–	–	–	1	(+harvest)
Black rat *Rattus rattus*	–	–	–	–	–	–	**2007**	**2007**	**2007**	–	–	3	3
Brown rat *Rattus norvegicus*	–	–	–	–	**2003**	–	–	–	–	–	–	1	1
House mouse *Mus musculus*	–	–	–	–	2003 failed	–	–	–	–	–	–	1	0
Red-whiskered bulbul *Pycnonotus jocosus*	–	–	–	–	–	–	–	–	–	(inc. 2012–14)	**2012–14**	1	1
Madagascar fody *Foudia madagascariensis*	–	–	–	–	–	–	–	–	–	**2012–15**	**2012–15**	2	2
Total attempts	1	1	1	2	3	1	2	2	1	4	2		
Success (Outcome)	(1 from control)	(1 from control)	1	1	2	(1 harvested for food)	1 (+ 1 indirect)	1 (+ 1 indirect)	1	2	2	20	11 (+ 2 indirect & 3 after control/ harvest)

Inner Islands chronological order of eradications	Aride	Mahé	Frégate	Cousine	Bird Island	Cousin	Curieuse	Denis	Sainte Anne	Anonyme	Ile aux rats	North Island (Ile du Nord)	Conception	Grande Sœur	Petite Sœur	Total attempts	Successful outcomes
Area (ha)	**73**	**15252**	**219**	**26**	**101**	**29**	**289**	**143**	**219**	**10**	**<1**	**201**	**69**	**85**	**35**		
Feral goat *Capra hircus*	**before 1920**	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	1
Feral cat *Felis catus*	**1930s**	–	**1981–82**	**1983–85**	–	–	**2000**	**2000**	–	–	–	**2003**	–	**(2010)**	–	7	6 (+ 1 indirect)
House crow *Corvus splendens*	–	**1977–94** occ. inc.	–	–	–	–	–	–	–	–	–		–		–	1	1
Common myna *Acridotheres tristis*	**1993–94** +occ. inc.	–	**2010–11** (98–02 failed) +occ. inc.	**2001–02** +occ. inc.	–	**2000–02** +occ. inc.	–	**2010–15** 2000–01 failed	–	–	–	2006–09 failed. 2012–ongoing*	–	2011 failed	–	9 (+1 ongoing)	5
Feral chicken *Gallus gallus*	–	–	–	**1996**	–	–	–	–	–	–	–	**2003**	–		–	2	2
Feral rabbit *Oryctolagus cuniculus*	–	–	–	–	**1996–97**	–	–	–	–	–	–		–		–	1	1
Barn owl *Tyto alba*	**1996** +occ. inc.	–	–	–	–	–	–	–	–	–	–	**(2005)** occ. inc.	–	**(2010)** occ. incv.	–	3	1 (+ 2 indirect)
Black rat *Rattus rattus*	–	–	–	–	**1996–97**	–	2000 failed	**2002** 2000failed	–	**2003 & 2006**	**2006**	**2005** 2003 failed	–	**2010**	**2010**	11	8
Brown rat *Rattus norvegicus*	–	–	**2000** 1996 failed	–	–	–	–	–	–	–	–	–	**2007**	–	–	3	2
House mouse *Mus musculus*	–	–	**2000**	–	1996–97 failed	–	2000 failed	**2002** 2000 failed	–	–	–	–	–	–	–	5	2
House sparrow *Passer domesticus*	–	**2003–04** +occ. inc.	–	–	–	–	–	–	–	–	–	–	–	–	–	1	1
Crested-tree lizard *Calotes versicolor*	–	–	–	–	–	–	–	–	**2003–14**	–	–	–	–	–	–	1	1
Ring-necked parakeet *Psittacula krameri*	–	2012–ongoing*	–	–	–	–	–	–	–	–	–	–	–	–	–	(1 ongoing)	
Black-headed ant *Pheidole megacephala*	2014–ongoing*	–	–	(control 2008)	–	–	–	–	–	–	–	–	–	–	–	(1 ongoing)	
Total attempts	4 (+ 1 ongoing)	2 (+ 1 ongoing)	6	3	3	1	3	7	1	2	1	6 (+ 1 ongoing)	1	4	1	45 (+ 3 ongoing)	
Success (Outcome)	4	2	4	3	2	1	1	4	1	2	1	3 (+ 1 indirect)	1	1 (+ 2 indirect)	1		31 (+ 3 indirect)

Table 2 Temporal distribution and success outcomes of attempts to eradicate invasive animals in Seychelles (n = 68). The few eradications declared successful after 2015 had their last individuals eliminated on this year or before.

	Pre-1995	1995–2004	2005–2015	Total attempts
Direct eradications	8	18	16	42
Indirect eradications	2	-	6	8
Failed outcomes	3	10	2	15
Ongoing attempts	-	-	3	3
Total attempts	13	28	27	68

Table 3 Approximate areas rehabilitated (see explanation in text) on islands where vegetation management (replanting and spontaneous native woodland recovery after exotic species removal) has been undertaken.

	Area of Planting after exotic sp. removal (ha)	Area of Woodland / Shrubland rehabilitation (ha)	Area of Woodland recovery (Pisonia dominated) (ha)
Frégate	60	-	-
Ile du Nord	45–50	-	-
Félicité	40	-	-
Denis	35	-	4
Praslin	25 (inc. 10 bare land)	20	-
Mahé	15–20	-	-
Curieuse	18–20	-	-
Aride	7	-	62
Cousin	?	-	27
Cousine	10	-	16
Conception	2 (coconut removal)		> 1
Bird (Island aux Vaches)	< 1	-	35
St Anne	1	-	-
Silhouette	< 1	-	-
Moyenne	0.5	-	-
Anonyme	0.5	-	-
Inner Islands	ca. 228	20	ca. 145
Desroches	12	-	-
D'Arros	11	-	-
Aldabra	(sisal removal)	<5	-
Alphonse	1	-	-
Outer Islands	ca. 72	<5	-
TOTAL	ca. 300	25	ca. 145

abandoned coconut plantations and lowland coastal forests previously dominated by invasive species.

Habitat rehabilitation was initiated on Aride and Cousin in the 1970s (Warman & Todd, 1984; Kömdeur & Pels, 2005), and since the mid-1990s has been implemented on Frégate, Ile du Nord, Denis, Curieuse, Cousine and Félicité as well as to a minor extent on other granitic islands. Very little vegetation restoration has taken place in the outer islands. On D'Arros, some 11 ha of former coconut plantations have progressively been replaced by plantations of native trees since 2009 (von Brandis, 2012; von Brandis, pers. comm. 2015). On Alphonse and Desroches, since 2006 and 2009 respectively, small areas have been cleared of exotics and replanted. At Aldabra, rehabilitation activities to control sisal (*Agave sisalana*) have taken place since the 1970s on Picard, Polymnie and Ile Michel and, since 2013, to eradicate it (van Dinther, et al., 2015).

In Seychelles, control and clearing of exotic plants has mostly been done physically, using machetes and chainsaws for woody plants, pulling by hand for creepers, and sometimes using heavy machinery, as on Frégate or Ile du Nord. Chemical treatments have rarely been used to eradicate invasive plants, although some trials have been conducted on several islands (Kaiser-Bunbury, et al., 2015). Elimination of coconut trees inland of the beach crest has been done on most of the rehabilitated islands that had, in the past, been exploited as coconut plantations. These have been replaced by forests dominated by native and endemic species, through natural regrowth or replanting.

Although precise figures are not available for all islands, we estimate that at least 220 ha have been actively cleared of alien invasives, replanted with native trees and maintained. This reaches 300 ha when including areas partially restored and ca. 470 ha when including natural recovery of native woodland. Rehabilitated vegetation now covers 17% (405/2,480 ha) of middle-sized and small inner islands, but only a tiny proportion (c.1%) of the country area.

Nurseries were established by successive ministries and associated public authorities responsible for the environment on Mahé and Praslin. On private islands such as Frégate (early 1990s), Ile du Nord (early 2000s) and Félicité (early 2010s), nurseries dedicated to propagating Seychelles native plants and trees have been created. These have successfully multiplied most of the 85 endemic plants of Seychelles and have produced tens of thousands of saplings that have been used in island rehabilitation. Based on the areas rehabilitated at the average density of 1,000 plants/ha normally used in Seychelles (Kueffer & Vos, 2004), we estimate that a minimum of 220,000 native trees have been planted in Seychelles over the last 50 years.

Species translocations to rehabilitated islands

Species translocations to predator-free islands with suitable habitats also contribute to the process of island restoration. Table 4 lists 20 documented translocations of eight rare and threatened species and one common species that have taken place to date on ten rehabilitated islands. This includes six species of Seychelles endemic land birds, one species of reptile and one very rare insect. Ninety percent of these translocations were successful (including nine reintroductions and 10 conservation introductions of rare and threatened species). In addition, a common land bird was successfully transferred to Bird Island (Ile aux Vaches). The two translocations which failed were of Seychelles leaf-insects, *Phyllum bioculatum*, to Conception and of Seychelles white-eyes, *Zosterops*

Table 4 Translocations of rare and threatened species (other than Aldabra tortoises) and common species to rehabilitated islands in Seychelles. X = naturally present; Normal case: reintroduction; Underlined: conservation introduction; Italic: failed attempt; *: initial transfer trials to Aride between 1978 and 1995 were unsuccessful; ©: cats eradicated; ®: rats eradicated; species are listed per chronological order of translocation; IUCN threat status between brackets likely to change due to status or taxonomic re-assessment; § endemicity questioned by Fritz, et al. (2013).

	IUCN Threat Status	Cousin -	Aride ©	Cousine ©	Frégate © ®	Denis © ®	Ile du Nord © ®	Conception ®	D'Arros ® ©	Picard (Aldabra) ©	Bird ®	Total
Seychelles fody *Foudia sechellarum*	*NT*	X	2002	X	-	2004	-	-	1968	-		3
Seychelles warbler *Acrocephalus sechellensis*	*NT*	X	1988	1990	2011	2004						4
Seychelles magpie-robin *Copsychus sechellarum*	*EN*	1994–95	2002*	1995–96	X	2008						4
Seychelles white-eye *Zosterops modestus*	(VU)			*2007*	2001		2007	X				3
Aldabra rail *Dryolimnas (cuvieri) aldabranus*	(LC)									1999		1
Sey. black paradise flycatcher *Terpsiphone corvina*	CR					2008						1
Seychelles black-mud terrapin *Pelusios subniger parietalis§*	(CR)		2012				2008					2
Seychelles leaf insect *Phyllum bioculatum*	(LC)							*2010*				1
Number of translocations of rare and threatened species		1	4	3 (1 failed)	2	4	2	1 (failed)	1	1	0	19 (2 failed)
Seychelles sunbird *Cynniris dussumieri*	LC										2006	1
Number of translocations per island (all native species)		1	3	2	2	4	2	0	1	1	1	20 (2 failed)

modestus, to Cousine (Galman, 2011; Julie Gane / Cousine Island, pers. comm. 2015). The islands with the highest number of successful translocations are Denis (four) and Aride (three), then Cousine, Frégate and Ile du Nord (two), all other islands having benefited from only one species translocation.

Table 5 Main islands free of rats and cats in Seychelles. Islands are listed in order of decreasing size. Small islands of less than 10 ha in *italics*. Islands naturally free of rats and cats are also marked by *. *Note: Conception was found to have been recolonised by rats in late 2017, while writing up this paper.*

Islands free of rats and cats	Area (ha)	Rats eradicated	Cats eradicated
Inner Islands			
Frégate	219	X	X
North Island *(Ile du Nord)*	201	X	X
Denis	143	X	X
Bird Island *(Ile aux Vaches)*	101	X	
Grande Sœur	85	X	X
Aride	73		X
Conception	69	X	
Petite Sœur	35	X	
Cousin*	29		
Cousine	26		X
Ile aux Récifs*	20		
Anonyme	10	X	
Mamelles*	9		
Ile aux Vaches Marines*	5		
Ile aux Cocos*	2		
Ile aux Rats	1	X	
Outer Islands			
Grande Ile (Cosmoledo)	143	X	X
D'Arros	140	X	X
St Joseph atoll*	122		
Bancs du Sud (Providence)*	71		
Marie-Louise*	53		
Desnoeufs*	35		
Ile du Sud-Ouest (Cosmoledo)*	30		
Bancs Africains*	31		
Goëlettes (Farquhar)*	25		
Grand Polyte (Cosmoledo)	21	X	X
St Francois (Alphonse)*	17		
Ile du Nord (Cosmoledo)*	11		
Ile du Nord-Est (Cosmoledo)*	9		
Banc de Sable (Farquhar)*	7		
Pagode (Cosmoledo)*	6		
Goëlettes (Cosmoledo)*	5		

Translocations of Aldabra giant tortoises (IUCN Red List category ‘Vulnerable’) have been accounted for separately as many are ancient and/or poorly documented (dates uncertain; possible failures not accounted for). After the giant tortoises naturally present on most of the granitic islands had been overexploited and driven to extinction (Fauvel, 1909), 18 successful translocations of Aldabra giant tortoises have taken place. Eight granitic islands have been repopulated since 1850, including Frégate, Curieuse and Cousin where they were reintroduced before 1950, Moyenne (probably in the 1970s), and Ile du Nord, Cousine, Grande Sœur and Silhouette that were last repopulated during the period 1993–2012 (Gerlach, et al., 2013). Aldabra giant tortoises have also been introduced or reintroduced to 10 coralline islands (Bird, Denis, D’Arros, Desroches, Rémire, Alphonse, Farquhar, Providence, Assomption, Cosmoledo) during the past 25–50 years, although some of these populations are small and of uncertain long-term viability (Gerlach, et al., 2013). Four of these translocations were to rat and cat free islands (Cousin, Cousine, Frégate and Bird). The reintroduction of giant tortoises to Aride in 1933–34 is not counted, as the animals were removed in 1951 and brought to Cousin (Warman & Todd, 1984); however, some are planned to be reintroduced from Frégate Island in 2018.

Including the reintroductions of giant tortoises, the total number of successful species translocations between islands in Seychelles is 36.

DISCUSSION

With 50 island populations of invasive vertebrates (of 14 species) eradicated from islands, Seychelles stands as a world leader. In 2014, it was ranking third after Australia and the USA for tropical islands, and seventh when all islands are considered (DIISE, 2017; Rocamora, 2015). Despite more eradication attempts during the period 1995–2004, a lower success rate (64%) was recorded compared to the following decade (89%). This may be a result of improving project selection, field implementation, and post-eradication biosecurity measures to prevent reinvasions.

Global conservation impacts of Seychelles island restoration

Population translocations to islands that have benefited from predator eradications and habitat rehabilitation have improved the conservation status of endemic species threatened with global extinction in Seychelles (Henriette, 2011; Nevill, 2011; Russell, et al., 2016). Island restoration has allowed 17 successful reintroductions or conservation introductions of eight rare and threatened species and the down-listing of four globally threatened birds on the IUCN Red List: the Seychelles warbler (*Acrocephalus sechellensis*) from Critically Endangered to Near Threatened; the Seychelles magpie-robin (*Copsychus sechellarum*) and Seychelles white-eye (*Zosterops modestus*) from Critically Endangered to Endangered and Vulnerable respectively; and the Seychelles fody (*Foudia sechellarum*) from Vulnerable to Near Threatened. The Seychelles black paradise flycatcher (*Terpsiphone corvina*), which was transferred to Denis Island, is still considered Critically Endangered. The Aldabra giant tortoise (Vulnerable) has also benefited from 18 successful translocations (Gerlach, et al., 2013).

Ecosystem recovery

The recovery of native fauna and flora on rehabilitated islands where introduced predators and competitors have been eradicated has already been observed on many islands around the world (Mulder, et al., 2011; Veitch, et al., 2011; Russell & Holmes, 2015). This is also occurring in Seychelles, where monitoring of birds, reptiles, invertebrates and plants has been undertaken and casual observations collected (Rocamora & Henriette, 2015). After the eradication of introduced predators and competitors, some species that had become inconspicuous started to reappear (e.g. giant millipedes *Sechelleptus sechellarum* and endemic snails *Stylodonta unidendata* on Conception; Galman, 2011). Five species of seabirds (*Ardenna pacifica, Gygis alba, Anous tenuirostris, Phaethon lepturus* and *Sula dactylatra*) have (re)established nine new breeding populations on seven rehabilitated islands. Populations that already existed have increased, as observed on most other islands where invasive mammals have been eradicated around the world (Brooke, et al., 2017). Reptiles and land birds have typically shown increasing or stable trends and some (e.g. Seychelles blue pigeon *Alectroenas pulcherrima*, common moorhen *Gallinula chloropus*) have recolonised islands, whereas invertebrates showed mixed responses, including strong decreases for some groups. This is probably linked to the increase in native land birds, reptiles and large invertebrates that had previously been preyed upon by rats and cats (Galman, 2011). As part of a global study to demonstrate the impact of mammal eradications on native wildlife (Jones, et al., 2016), 67 populations of 26 native vertebrates (13 land birds, eight seabirds, five reptiles) were identified as having benefited from these operations in Seychelles (Rocamora & Henriette, 2015). This illustrates how important it is to undertake ecosystem monitoring before and after eradications to measure and understand the ecological changes that occur on islands under rehabilitation.

The reintroduction of giant tortoises, which dominated the terrestrial ecosystems of Seychelles for millions of years, is an essential step in the island restoration process. These animals fill an important (but still poorly known) role in the ecosystem by dispersing and promoting the germination of seeds, fertilising native plants, and influencing soil invertebrate communities through their dung. These mega-herbivores are used as ecological analogues to replace extinct tortoises and help restore island ecosystems in Mauritius and Rodrigues (Griffiths, et al., 2010; Hansen, et al., 2010; Griffiths, 2014). The (re)introduction of Seychelles white-eyes (Rocamora & Henriette-Payet, 2008), which disseminate many native berry-producing trees, also contributes to the restoration process. Future challenges include a more integrated 'ecosystem approach', aiming at rehabilitating entire habitats and communities (including invertebrates), rather than focusing only on 'flagship' threatened species (Asconit & ICS, 2010; Kaiser-Bunbury, et al., 2010; Galman, 2011).

Seabirds play a critical role in ecosystem recovery as they boost soil nutrients thereby assisting the development of the ground microfauna (Mulder, et al., 2011). Seabird recolonisation can be slow, although decoys and sound recordings to attract passing adults can speed this process (Jones & Kress, 2012). In Seychelles, this has only been done for the sooty tern (*Onychoprion fuscatus*) on Denis Island, with little success (Feare, et al., 2015). Seabird translocations may also be tried in future in Seychelles; this technique has been employed successfully in the Pacific (Kappes & Jones, 2014) and is being trialled in Mauritius (Carl Jones & Nik Cole, pers. comm., 2016).

Scaling up eradication projects and increasing the rat and cat free area of Seychelles to create more biodiversity refuges

Since the 1970s, ecosystem restoration has taken place on 25 small and medium sized islands of Seychelles (i.e. < 2,000 ha; see Fig. 2). As a result, island refuges for native biodiversity and particularly for rare species threatened with extinction have multiplied. This process was started on NGO-owned islands in the 1970s, then followed on government and privately-owned islands, the public trust Seychelles Island Foundation and more recently on government islands managed by the parastatal Island Development Company and associated partners (private hotel/villa owners and the Island Conservation Society).

With about 30 small and medium-sized islands free of rats and cats (see Table 5), Seychelles probably has proportionally more territory (3.9%) free of invasive predatory mammals than most island countries. Rats and cats have been removed from 11 islands larger than 10 ha. Between 1996 and 2011, the number of islands of ≥10 ha free of rats has increased from four (Aride, Cousin, Cousine, Ile aux Récifs) to 12, and the total rat-free area of Seychelles has more than tripled from 581 ha to 1,757 ha (Fig. 3).

Nevertheless, there is scope for more eradications to benefit both wildlife and humans. This would require scaling up the size of islands tackled for eradications. Rats

Table 6 Additional islands in Seychelles where rodents and cats could be eradicated and their reinvasion prevented with currently available techniques. X = presence. Moyenne, Longue and Ronde (Ste Anne group) would require a combined operation owing to their mutual proximity.

	Area (ha)	Black rat	House mouse	Feral cat
Inner Islands				
Curieuse	286	X	?	
Félicité	230	X		X
Marianne	100	X		X
Bird (Ile aux Vaches)	101		X	
Aride	73		X	
Ronde (Praslin)	19	X		
Thérèse	74	X		X
Longue	17	X	?	X
Moyenne	9	X		
Ronde (Mahé)	2	X		
***Outer Island*s**				
Assomption	1,171	X		X
Coétivy	931	X		X
Astove	660	X		X
Ile du Sud (Farquhar)	400	X		X
Desroches	394	X		X
Ile du Nord (Farquhar)	300	X		X
Poivre	255	X		X
Alphonse	174	X		X
Providence	157	X		X
D'Arros	140		X	
Platte	54	X		X
Rémire	27	X		X
Manahas (Farquhar)	10	X		X
Marie-Louise	53		X	
Desnoeufs	35		X	

and cats could be removed from another 22 islands with currently available techniques (Table 6). This includes five more granitic (inner) islands, plus three small islands in the Ste Anne group which would require a permanent grid of rat bait stations owing to their proximity to other infested islands.

The outer islands have greater restoration potential, with 14 islands where rats (and cats) could be eradicated. However, reinvasion may be difficult to prevent through strict biosecurity on three of the larger islands (Coëtivy, Assomption, Desroches) depending on future developments envisaged. The maximum potential area that could be cleared of rats and cats is currently ca. 7,000 ha or 15.4% of the total area of the country (see Fig. 3 for inner and outer islands totals). Clearing rats and cats from these additional 19–22 islands would open huge possibilities for ecosystem rehabilitation and population recovery of many species (land birds, seabirds, reptiles, amphibians, invertebrates and plants). The eradication of rats from Aldabra and Cosmoledo (Menai Island) atolls, which have large extensions of mangroves, is not considered currently feasible. However, this may change if new techniques become available in future, in which case up to half (49.7%) of the country area could be made rat and cat free in the long term.

Eradication operations should now be extended to invertebrates such as invasive ants, moths or snails that also have a high negative impact on native biodiversity.

However, the ability to apply permanent biosecurity protocols will be critical for these islands to retain their pest free status. In the low-lying coralline outer islands, options for translocation of native species may be limited by sea-level rise in future decades.

Developing partnerships associating nature-based tourism to fund ecosystem restoration

Seychelles provides some examples of collaborations between private island owners, parastatals, government agencies and NGOs to achieve successful control or eradication of invasive species and to develop ecosystem rehabilitation programmes (Asconit & ICS, 2010; Government of Seychelles, 2011; Kueffer, et al., 2013). By creating synergies, such partnerships can speed-up the long-term process of ecosystem rehabilitation, and can help meet the financial, technical and ecological challenges of these complex operations (Rocamora, 2010a). Many islands are engaged in tourism activities that can help

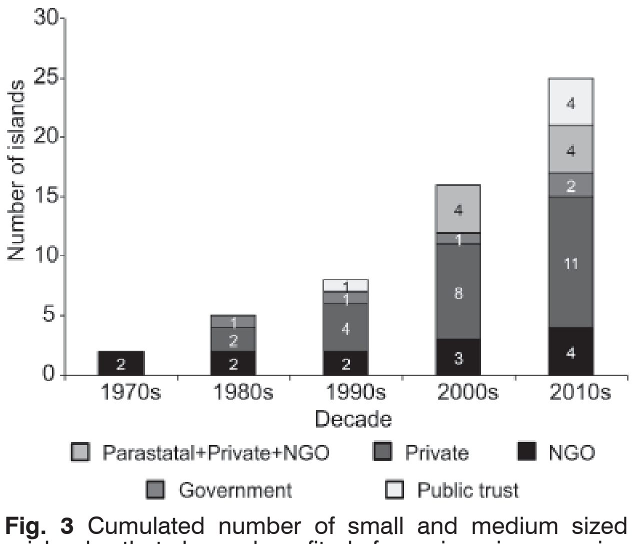

Fig. 3 Cumulated number of small and medium sized islands that have benefited from invasive species management and other forms of ecological restoration in Seychelles since the 1970s, and type of management.

conservation funding (Rocamora & Payet 2002; Nevill, 2004; Skerrett, 2010). The successful record of Seychelles in invasive species management and ecological restoration of many small islands is attributable, at least in part, to the fact that these operations can make economic sense for private owners or investors wishing to generate revenue through ecotourism operations. High densities of rats are incompatible with tourism, and eradication and biosecurity procedures are more cost-effective than long-term pest control. The progressive rehabilitation of an island ecosystem to recreate a wildlife sanctuary with its original fauna can be marketed as an attraction (Rocamora, 2010b; Samways, et al., 2010a). The development of partnerships associating sustainable nature-based tourism with funding ecosystem restoration must be strongly encouraged.

Perspectives and challenges in invasive species management and restoration

The creation of island refuges where invasive species management enables the (re)introduction of species that cannot survive in the presence of invasive predators has proven to be extremely effective in improving the conservation status of various species, including endemic birds of Seychelles that had come very close to global extinction. However, several factors limit the further development of this strategy.

Availability of additional islands suitable for restoration and preservation

Although Seychelles still has considerable potential to increase its area free of alien predatory mammals, the number of islands where such operations can be conducted is limited. In the inner islands, there are currently five to eight islands left which could be made, and kept, free of rats and cats. Most of these have actual or planned development projects or do not presently fulfil the required strict biosecurity conditions to prevent reinvasion.

Challenges to eradicate rats from larger islands in a humid, tropical climate

Techniques currently available to eradicate rats are more successful in temperate and sub-Antarctic climates than in tropical environments, particularly humid ones (Russell & Holmes, 2015). Here, rains (that can seriously affect the attractiveness and palatability of rat pellets) and abundant natural food (which reduces the likelihood that rats will eat the bait) can be present for much of the year (Varnham, 2010; Keitt, et al., 2015). Whereas rats (*Rattus rattus* and *R. norvegicus)* have been eradicated from islands of over 10,000 ha outside of the tropics, the largest tropical eradication of black rats to date is Cayo Centro, Chinchorro Bank (539 ha, Mexico) and for brown rats Frégate Island (219 ha, Seychelles) (DIISE, 2017).

Mangroves are also a limiting factor and the main obstacle to a large-scale rat eradication on Aldabra atoll (15,380 ha; c.1,300 ha of mangroves). Although small areas of mangroves can be dealt with by placing bait stations or tying rodenticide blocks to trees (Samaniego-Herrera, et al., 2015; Samaniego-Herrera, et al., 2017), using 'collars' or 'bolas' (Harper, et al., 2015; Rocamora & Henriette, 2015), efficient methods to eradicate rats from large tropical islands are not yet sufficiently well developed (Russell & Holmes, 2015).

Rat eradications may prove challenging on large islands with high densities of coconut trees, where nuts provide abundant food for rats both in the trees and on the ground, (Climo & Rocamora, 2006). This requires bait to be available at high densities and for a long-time period. Unpredictable rainfall and the year-round high primary productivity of Seychelles ecosystems add further challenges to conducting rat eradications. Abundance of bait-eating crabs can also cause problems (Griffiths, et al., 2011; Wegmann, et al., 2011; Keitt, et al., 2015) but this has not so far been a major problem in Seychelles.

Suitable habitats on restored islands non-existent or too limited for some species

Some rare and threatened species require very specific habitats that may not be found on small to medium sized islands. Examples include the Critically Endangered Seychelles sheath-tailed bat (*Coleura seychellensis*) and the Vulnerable Seychelles swiftlet (*Aerodramus elaphrus*), both of which occur only on the larger granitic islands, breeding in caves and feeding on flying insects. Such limitations also apply to endemic plants and animals (reptiles, amphibians, invertebrates) found exclusively at altitudes above 300–400 m on Mahé and Silhouette where much of the terrestrial diversity of Seychelles is concentrated (Senterre, et al., 2013). Most of these species would probably not survive on the low-lying small islands, where climatic conditions differ from the more humid and colder high altitudes. Some species may also require large expanses of specific habitats that could not be made available on small islands, such as the Vulnerable Seychelles black parrot (*Coracopsis barklii*), which requires extents of palm-dominated forests (Rocamora & Laboudallon, 2013).

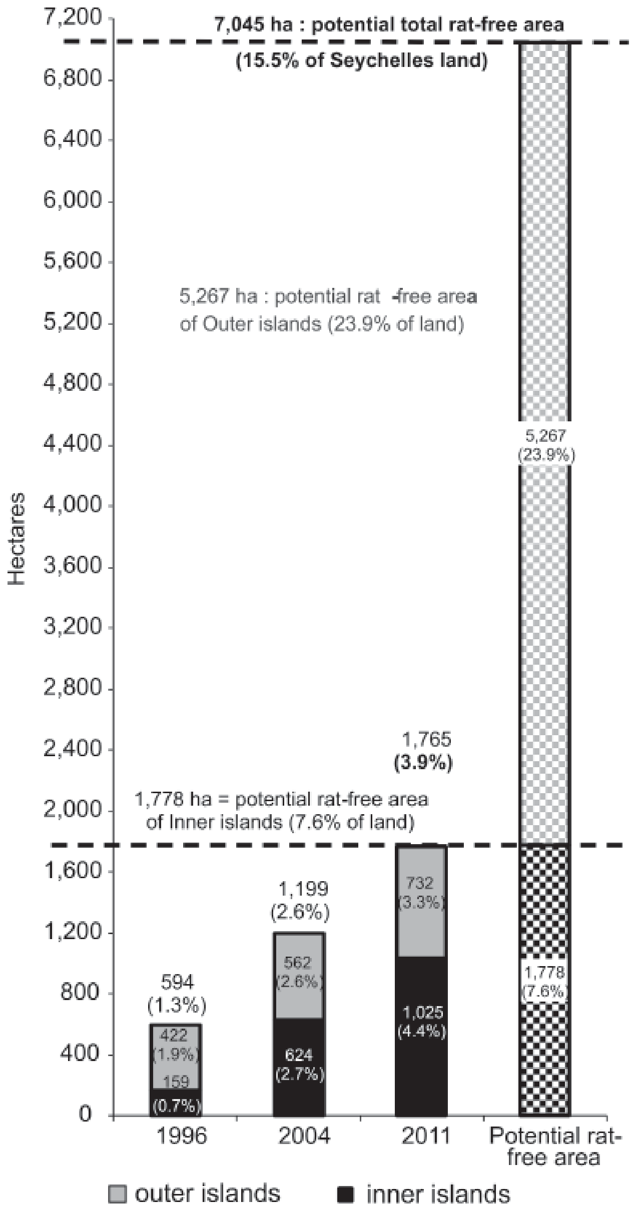

Fig. 4 Time progression of the rat and cat free area in the inner islands, the outer islands and the whole of Seychelles (reported in hectares and as % of the land surface). The total land surface that could potentially be freed of rats and cats with currently available techniques is also indicated.

Increased interspecific interactions on small islands with multiple (re)introductions

The number of species that can coexist on a given island is limited by the quality and diversity of habitats available on the island (MacArthur & Wilson,1967), which is influenced by island area and characteristics. The survival of any small newly (re)introduced population will depend on interactions with the species already there (Blondel, 1979). This factor may partly account for the failed translocation of Seychelles leaf insects to Conception Island, many of which were preyed on shortly after their release (Galman, 2011). On Cousine Island (26 ha), the 23 Seychelles white-eyes translocated in 2008 established a small breeding population, but predation of nests and fledglings and high adult mortality did not allow this population to grow despite considerable efforts. Young fledglings had to be caged and fed through the mesh by adults as they were repeatedly preyed on by another introduced species, the Seychelles magpie-robin (Rocamora, 2013). Such problems were not observed after the introduction of white-eyes to two larger islands, Ile du Nord (201 ha), and Frégate (219 ha) where a large population of Seychelles magpie-robins was present. This suggests a limit to the number of (re) introduced species that a small island can host. In other words, it will become more difficult to 'squeeze in' new species into small rehabilitated islands as their ecosystems become increasingly saturated.

The need for alternative conservation approaches on large islands

In Seychelles, the availability of many small islands suitable for rehabilitation, and the presence of private island owners willing to develop ecotourism has favoured the *in situ* approach. In other countries, such as Mauritius or New Zealand, more intensive and costly *ex situ* techniques, which require the additional step of readapting the captive reared animals into the wild, have also been used (Jones & Merton, 2012). In view of the limitations of the 'small island restoration' model, ecosystem restoration programmes may be developed on the large islands of Seychelles through the creation of "mainland islands" where invasive species are controlled or excluded to enable native species to thrive, as in New Zealand, Australia and Hawaii (Innes & Saunders, 2011). Predator control and habitat rehabilitation programmes are being developed at large scales in Mauritius and La Réunion for the conservation of threatened land birds (Vikash Tatayah & Marc Salamolard, pers. comm., 2016). Similar operations could be conducted in selected priority sites on the largest granitic islands of Seychelles. Innovative management techniques such as predator-proof fences, self-resetting traps, more effective or target-specific bait, etc. will be key to success. The only Seychelles example to date is permanent rat control using grids of bait-stations at the main breeding areas of the Endangered Seychelles white-eye on Mahé (25–40 ha) since 2006 (Rocamora & Henriette, 2015). More such projects could bring some of the rarest birds of Seychelles, now restricted to remote small island sanctuaries, back to the main islands where they once lived. By providing better access to these species and native wildlife in general, such 'mainland islands' would benefit environmental education programmes for the public and school children. This in turn would increase awareness of, and hopefully support for, pest management programmes.

CONCLUSION

Seychelles' achievements with invasive species management and other island restoration practices are remarkable. This includes a minimum of 50 island eradications of invasive vertebrate populations, the rehabilitation of ca. 470 ha of natural habitats, and at least 36 successful island translocations of native species. The rehabilitation of small and medium sized islands has made possible the down-listing of four globally threatened land birds in the IUCN Red List and the recovery of many other native animals and plants.

Scaling up the size of islands for eradications is now required in Seychelles. Factors limiting rat eradication on larger islands include high densities of coconut trees and the presence of mangroves, especially on Aldabra atoll. Invasive predators such as rat and cats could be eradicated from 19–22 more islands with existing techniques, mainly in the outer (coralline) islands. As a result, the proportion of the country's land area free of rats and cats would increase from 3.9% to 15.4%, but new techniques will be needed to remove rats from Aldabra and bring this proportion to 50%. Making half of Seychelles rat and cat free by 2030–2050 could be a commitment made by Seychelles government and the main stakeholders involved as part of the Honolulu Challenge on Invasive Alien Species, launched at the 2016 IUCN Word Conservation Congress in Hawaii.

Eradication operations need to be extended to invertebrates such as invasive ants, moths or snails that also have a high negative impact on native biodiversity. Apart from the availability of islands free of invasive predators, limiting factors to further translocations of rare and threatened species include lack of suitable habitats and increased interspecific interactions on small islands with multiple (re)introductions. Because of global warming and sea-level rise, the long-term relevance of island restoration and species translocations to outer low-lying coralline islands is questioned.

Local partnerships and financial support from nature-based tourism have been key to past successes. We recommend for these to be enhanced and alternative conservation approaches such as 'mainland-islands' be developed on large islands. Most importantly, biosecurity protocols will be critical to prevent (re)invasion of invasive species, as lack of vigilance and poor biosecurity could undo so much of what has already been achieved.

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REFERENCES

Asconit and ICS. (2010). France – Seychelles. *Rehabilitating Island Ecosystems. Eradication of invasive exotic species and Re-introduction of threatened endemic species*. Asconit Consultants, ICS and FFEM, Paris and Seychelles. <http://www.globalislands.net/userfiles/seychelles6.pdf>. Accessed 15 September 2017.

Beaver, K. and Mougal, J. (2009). *Review and Evaluation of Invasive Alien Species (IAS) Control and Eradication Activities in Seychelles. Mainstreaming Prevention and Control Measures for Invasive Alien Species into Trade, Transport and Travel across the Production Landscape*. Consultancy Report, GoS-UNDP-GEF Project. Victoria, Mahé, Seychelles: Plant Conservation Action Group (PCA). <http://www.pcaseychelles.org/uploads/1/2/3/6/12369400/seychelles_ias_control-erad_review_pca_report_only.pdf>. Accessed 15 September 2017.

Blondel, J. (1979). *Biogéographie et Écologie*. Paris: Masson.

Brooke, M. de L., Bonnaud, E., Dilley, B.J., Flint, E.N., Holmes, N.D., Jones, H.P., Provost, P., Rocamora, G., Ryan, P.G., Surman, C. and Buxton, R.T. (2017). 'Seabird population changes following mammal eradications on islands'. *Animal Conservation* 21: 3–12.

Climo, G. and Rocamora, G. (2006). *The Successful Eradication of Black Rats from North Island (Seychelles) in September 2005 and Proposed Strategies to Reduce the Risks of Reinvasions*. Réhabilitation des Ecosystèmes Insulaires (FFEM) Project. North Island: Island Conservation Society.

DIISE. (2017). *The Database of Island Invasive Species Eradications*. Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group. University of Auckland and Landcare Research New Zealand. <http://diise.islandconservation.org>. Accessed 31 December 2017.

Fauvel, A.A. (1909). *Unpublished Documents on the History of the Seychelles Islands Anterior to 1810.* Victoria: Seychelles Government.

Feare, C.J., French, G.C.A., Nevill, J.E.G., Pattison-Willits, V.S., Wheeler, V., Yates, T.L., Hoareau, C. and Prescott, C. (2015). 'Attempted re-establishment of a sooty tern *Onychoprion fuscatus* breeding colony on Denis Island, Seychelles'. *Conservation Evidence* 12: 19–24.

Fritz, U., Branch, W.R., Gehring, P-S, Harvey, J., Kindler C., Meyer, L., Du Preez, L., Široký, P., Vieites, D.R. and Vences, M. (2013). 'Weak divergence among African, Malagasy and Seychellois hinged terrapins (*Pelusios castanoides*, *P. subniger*) and evidence for human-mediated oversea dispersal'. *Organisms Diversity and Evolution* 13: 215–224.

Galman, G. (2011). *Suivi Spatio-temporel des Communautés d'Arthropodes, Effets de l'Eradication des Rats et Tentative de Réintroduction d'un Insecte Rare dans des Îles en Cours de Réhabilitation des Seychelles*. Paris: Muséum National d'Histoire Naturelle.

Gerlach, J., Rocamora, G., Gane, J., Jolliffe, K. and Vanherck, L. (2013). 'Giant tortoise distribution and abundance in the Seychelles islands: Past, present and future'. *Chelonian Conservation and Biology* 12(1): 70–83.

Government of Seychelles. (2011). *The National Invasive Alien Species (Biosecurity) Strategy for Seychelles 2011–2015*. Victoria, Seychelles: Ministry of Home Affairs, Environment and Transport & Ministry of Industry, Natural Resources and Industry.

Griffiths, C.J. (2014). 'Rewilding in the Western Indian Ocean'. In: J. Gerlach. *Tortoises of the Western Indian Ocean*, pp. 325–349. Manchester, UK: Siri Scientific Press.

Griffiths, C.J., Jones, C.G., Hansen, D.M., Puttoo, M., Tatayah, R.V., Müller, C.B. and Harris, S. (2010). 'The use of extant non-indigenous tortoises as a restoration tool to replace extinct ecosystem engineers'. *Restoration Ecology* 18: 1–7.

Griffiths, R., Miller, A. and Climo, G. (2011). 'Addressing the impact of land crabs on rodent eradications on islands'. *Pacific Conservation Biology* 17: 347–353.

Hansen, D.M., Donlan, C.J., Griffiths, C.J. and Campbell, K.J. (2010). 'Ecological history and latent conservation potential: Large and giant tortoises as a model for taxon substitutions'. *Ecography* 33: 272–284.

Harper, G.A., Van Dinther, M., Russell, J.C. and Bunbury, N. (2015). 'The response of black rats (*Rattus rattus*) to evergreen and seasonally arid habitats: Informing eradication planning on a tropical island'. *Biological Conservation* 185: 66–74.

Henriette, E. (2011). 'Conservation Introductions as a Tool for the Recovery of Endangered Island Species: Territoriality and Demography of the Seychelles White-eye *Zosterops modestus* on Frégate Island'. Docteur en Ecologie Thesis. Paris: Muséum National d'Histoire Naturelle.

Innes, J. and Saunders, A. (2011). 'Eradicating multiple pests: an overview'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 177–181. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

IUCN. (2013). *Guidelines for Reintroductions and Other Conservation Translocations*. Version 1.0. Gland, Switzerland: IUCN Species Survival Commission.

Jones, C. and Merton, D. (2012). 'A Tale of Two Islands: The Rescue and Recovery of Endemic Birds in New Zealand and Mauritius'. In: J. Ewen, D. Armstrong, K. Parker and P. Seddon (eds.). *Reintroduction Biology: Integrating Science and Management*, pp. 33–72. Oxford: Blackwell Publishing Ltd.

Jones, H.P. and Kress, S.W. (2012). 'A review of the world's active seabird restoration projects'. *Journal of Wildlife Management* 76: 2–9.

Jones, H.P., Holmes, N.D., Butchart, S.H.M., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E., Burbridge, A.A., Campbell, K., Courchamp, F., Cowan, P., Cuthbert, R.J., Ebbert, J., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R. and Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences* 113 (15): 4033–4038.

Kaiser-Bunbury, C.N., Traveset, A. and Hansen, D.M. (2010). 'Conservation and restoration of plant–animal mutualisms on oceanic islands'. *Perspectives in Plant Ecology and Evolution Systematics*. 12: 131–143.

Kaiser-Bunbury, C.N., Mougal, J., Valentin, T., Gabriel, R. and Blüthgen, N. (2015). 'Herbicide application as a habitat restoration tool: Impact on native island plant communities'. *Applied Vegetation Science* 8 (4): 650–660.

Kappes, P. and Jones, H.P. (2014). 'Integrating seabird restoration and mammal eradication programs on islands to maximize conservation gains'. *Biodiversity Conservation* 23: 503–509.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

Kömdeur, J. and Pels, M.D. (2005). 'Rescue of the Seychelles warbler on Cousin Island, Seychelles: The role of habitat restoration'. *Biological Conservation* 124 (1): 15–26.

Kueffer, C., Beaver, K. and Mougal, J. (2013). 'Management of Novel Ecosystems in the Seychelles'. In: R.J. Hobbs, E.S. Higgs and C. Hall. *Novel Ecosystems: Intervening in the New Ecological World Order*, pp. 228–238. Oxford, UK: Wiley-Blackwell.

Kueffer, C. and Vos, P. (2004). *Case Studies on the Status of Invasive Woody Plant Species in the Western Indian Ocean: 5. Seychelles*. Forest Health & Biosecurity Working Papers FBS/4-5E. Rome, Italy: Forestry Department, Food and Agriculture Organization of the United Nations.

MacArthur, R.H. and Wilson, E.O. (1967). *The Theory of Island Biogeography*. Princeton, NJ: Princeton University Press.

Merton, D., Climo, G., Laboudallon, V., Robert, R. and Mander, C. (2002). 'Alien mammal eradication and quarantine on inhabited islands in the Seychelles'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 182–198. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Mulder, C., Anderson, W., Towns, D. and Bellingham, P. (2011). *Seabird Islands. Ecology, Invasion, and Restoration*. New York: Oxford University Press.

Nevill, J. (2001). 'Case Study 5.34. Ecotourism as a Source of Funding to Control Invasive Species'. In: R. Wittenberg and M.J.W. Cock (eds.). *Invasive Alien Species: A Toolkit of Best Prevention and Management Practices,* p. 202. Wallingford, Oxon, UK: CAB International.

Nevill, J. (2004). 'Ecotourism as a source of funding to control invasive species: the case of Seychelles'. *Insula, International Journal of Island Affairs*. *Special Issue February 2004*: 99–102.

Nevill, J. (2009). *Mainstreaming Prevention and Control Measures for Invasive Alien Species into Trade, Transport and Travel across the Production Landscape. National IAS Baseline Report*. GOS-UNDP – GEF. Victoria, Seychelles: Government of Seychelles/UNDP/GEF.

Nevill, J. (2011). 'Case Study 6. Ecosystem Rehabilitation. The Case of Small Islands in Seychelles – A Government, NGO and Private Sector Partnership'. In: *Government of Seychelles. Fourth National Report to the United Nations Convention on Biological Diversity*, pp. 84–87. Victoria, Republic of Seychelles: Department of Environment.

Richardson, D.S., Bristol, R. and Shah, N.J. (2006). 'Translocation of the Seychelles warbler *Acrocephalus sechellensis* to establish a new population on Denis Island, Seychelles'. *Conservation Evidence* 3: 54–57.

Rocamora, G. (1997). *Rare and Threatened Species, Sites and Habitats Monitoring Programme in Seychelles. Project G1 EMPS. Volume 1. Monitoring methodologies and recommended priority actions.* Victoria, Seychelles & Cambridge, UK: Ministry of Environment/BirdLife International/EU.

Rocamora, G. and Payet, R. (2002). 'Sustainable Ecotourism as a Financing Mechanism for the Conservation of Cosmoledo Atoll'. *Conference on Sustainable Development of Ecotourism in Small Islands Developing States (SIDS) and Other Small Islands*. Preparatory Conference for the International Year of Ecotourism, Mahé (Seychelles), 8–10 December 2001. Final Report. Annexed Case Study (unpaged). Madrid & Victoria: World Tourism Organisation & Seychelles Ministry of Tourism.

Rocamora, G. and Henriette-Payet, E. (2008). 'Conservation Introductions of the Seychelles White-eye on Predator-free Rehabilitated Islands of the Seychelles Archipelago, Indian Ocean'. In: P.S. Soorae. *Global Re-introduction Perspectives: Re-introduction Case Studies from Around the Globe*, pp. 121–129. Abu Dhabi, UAE: IUCN/SSC Re-introduction Specialist Group.

Rocamora, G. (2010a). 'Island Restoration'. In: A. Skerrett, P. Thelma and J. Skerrett. *Outer Islands of Seychelles*. *Zil Elwannyen Sesel*, pp. 178–183. Nairobi, Kenya: Camerapix.

Rocamora, G. (2010b). 'Oiseau-lunettes et autres espèces menacées: Bilan des actions menées aux Seychelles'. Conservation des Espèces et Populations Animales (*CEPA) Magazine* 2: 19–23.

Rocamora, G. (2013). '*Zosterops modestus* Seychelles White-eye'. In: R.J. Safford and A.F.A. Hawkins (eds.) *The Birds of Africa. Volume VIII: The Malagasy Region*, pp. 791–795. London: Christopher Helm.

Rocamora G. (2015). 'Restoring Islands to Save Species'. In: G. Rocamora and E. Henriette. *Invasive Alien Species in Seychelles: Why and How to Eliminate Them? Identification and Management of Priority Species*, pp. 55–95. Biotope, Mèze : Island Biodiversity & Conservation Centre, University of Seychelles and Paris : Muséum National d'Histoire Naturelle.

Rocamora G. and Henriette E. (2015). *Invasive Alien Species in Seychelles: Why and How to Eliminate Them? Identification and Management of Priority Species***.** Biotope, Mèze: Island Biodiversity & Conservation Centre, University of Seychelles and Paris: Muséum National d'Histoire Naturelle (Inventaires & Biodiversité series).

Rocamora G. and Laboudallon V. (2013). Seychelles Black Parrot *Coracopsis barklyi*. In: R. Safford and F. Hawkins (eds.). The birds of Africa, vol. 8: The Malagasy region: Madagascar, Seychelles, Comoros, Mascarenes, pp. 529–531. London: Christopher Helm.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Russell J., Cole N., Zuel N. and Rocamora G. (2016). 'Introduced mammals on Western Indian Ocean islands: Species distribution, eradication attempts, and long-term island restoration prospects'. *Global Ecology and Conservation* 6: 132–144.

Samways, M., Hitchins, P., Bourquin, O. and Henwood, J. (2010a). *Tropical Island Recovery. Cousine Island, Seychelles*. Chichester, UK: Wiley-Blackwell.

Samways, M.J., Hitchins, P.M., Bourquin, O. and Henwood, J. (2010b). 'Restoration of a tropical island: Cousine Island, Seychelles'. *Biodiversity and Conservation* 19(2): 425–434.

Samaniego-Herrera, A., Aguirre-Muñoz A., Méndez-Sánchez, F., Rojas-Mayoral, E. and Cárdenas-Tapia, A. (2015). 'Pushing the Boundaries of Rodent Eradications on Tropical Islands: Ship Rat Eradication on Cayo Centro, Banco Chinchorro, Mexico'. Poster presented at the 27th International Congress for Conservation Biology. Montpellier, France, 2–6 August 2015.

Samaniego-Herrera, A., Aguirre-Muñoz, A. Bedolla-Guzmán, Y., Cárdenas-Tapia A., Félix-Lizárraga, M., Méndez-Sánchez, F., Reina-Ponce, O., Rojas-Mayoral, E. and Torres-García, F. (2017). 'Eradicating invasive rodents from wet and dry tropical islands in Mexico'. *Oryx* 52(3): 559–570.

Senterre, B., Henriette, E., Chong Seng, L., Gerlach, J., Mougal, J. and Rocamora, G. (2013). *Seychelles Key Biodiversity Areas - Output 6: Patterns of Conservation Value in the Inner Islands*. Unpublished consultancy report, Victoria, Seychelles: Ministry of Environment-UNDP-GEF project.

Shah, N.J. (2001). 'Eradication of alien predators in Seychelles: An example of conservation action on tropical islands'. *Biodiversity and Conservation* 10(7): 1219–1220.

Shah, N.J. (2006). 'Ecological restoration of islands in the Seychelles'. *Conservation Evidence* 3: 1–2.

Shah, N.J. (2008). 'Managing expectations and outcomes for successful re-introductions and multiple stakeholders in Seychelles'. *Avian Biology Research* 1(1): 31.

Skerrett, A. (2010). 'Outer Island Conservation'. In: A. Skerrett, P. Thelma and J. Skerrett. *Outer Islands of Seychelles. Zil Elwannyen Sesel*, pp. 174–175. Nairobi, Kenya: Camerapix.

van Dinther, M., Bunbury, N. and Kaiser-Bunbury, C.N. (2015). 'Trial of herbicide control methods for sisal (*Agave sisalana*) in the arid island environment of Aldabra Atoll, Seychelles'. *Conservation Evidence* 12: 14–18.

Warman, S. and Todd, D. (1984). 'A biological survey of Aride Island Nature Reserve, Seychelles'. *Biological Conservation* 28: 51–71.

Varnham, K. (2010). *Invasive Rats on Tropical Islands: Their History, Ecology, Impacts and Eradication*. RSPB Research Report No. 41. Sandy, Bedfordshire, UK: Royal Society for the Protection of Birds.

Wegmann, A., Buckelew, S., Howald, G., Helm, J. and Swinnerton, K. (2011). 'Rodent eradication campaigns on tropical islands: novel challenges and possible solutions'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 239–243. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Veitch, C.R., Clout, M.N. and Towns, D.R. (2011). *Island invasives: eradication and management*. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Von Brandis, R.G. (2012). 'Rehabilitation of abandoned coconut plantations at D'Arros Island, Republic of Seychelles'. *Ocean & Coastal Management* 69: 1–7.

No detection of brodifacoum residues in the marine and terrestrial food web three years after rat eradication at Palmyra Atoll, Central Pacific

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Abstract Invasive alien species represent one of the greatest threats to native plants and animals on islands. Rats (*Rattus* spp.) have invaded most of the world's oceanic islands, causing lasting or irreversible damage to ecosystems and biodiversity. To counter this threat, techniques to eradicate invasive rats from islands have been developed and applied across the globe. Eradication of alien rats from large or complex island ecosystems has only been successful with the use of bait containing a rodenticide. While effective at eradicating rats from islands, rodenticide can persist in the ecosystem longer than the time required to eradicate the target rat population and can potentially harm non-target species. However, the persistence of rodenticides in ecosystems following rat eradication campaigns is poorly understood, though predictions can be made based on the chemical properties of the rodenticide and the environment it is applied in. Brodifacoum, a relatively persistent second-generation anticoagulant, was used to successfully eradicate rats from Palmyra Atoll. With this study, we evaluated the persistence of brodifacoum residues in terrestrial and marine species at Palmyra Atoll (Northern Line Islands) three years after rat eradication. We collected 44 pooled samples containing 121 individuals of the following: mullet (*Moolgarda engeli*), cockroaches (*Periplaneta* sp.), geckos (*Lepidodactylus lugubris*), hermit crabs (*Coenobita perlatus*), and fiddler crabs (*Uca tetragonon*). Despite detection of brodifacoum residue in all five of the species sampled in this study 60 days after the application of bait to Palmyra Atoll in 2011, brodifacoum residue was not found in any of the pooled samples collected three years after bait application. Our study demonstrates how brodifacoum residues are unlikely to persist in the marine and terrestrial food web, in a wet tropical environment, three years after rat eradication.

Keywords: aerial rodenticide broadcast, best practice, brodifacoum anticoagulant rodenticide, land crabs, *Rattus rattus*, risk assessment, tropical island

INTRODUCTION

Invasive alien species represent a key threat to native plants and animals on islands (Tershy, et al., 2015). In particular, invasive rodents are known to have widespread negative impacts following introduction to islands (Towns, et al., 2006), and rodents have been introduced to most of the world's island groups (Atkinson, 1985). In prior decades, techniques to eradicate invasive rodents from islands have been developed and applied across the globe, most using anticoagulant rodenticides (Howald, et al., 2007). Demonstrable conservation benefits are common following successful eradication (Jones, et al., 2016; Brooke, et al., 2017).

To date, rat (*Rattus* spp.) eradications on tropical islands experience a lower success rate than those in temperate regions (Russell & Holmes, 2015). Lack of seasonality and warm temperatures in tropical latitudes can provide year-round breeding opportunities and a consistent abundance of alternative food sources that rodents may choose instead of the offered bait. Tropical regions also host land crab populations which readily compete with rats for bait (Wegmann, et al., 2011; Holmes, et al., 2015). In 2011, Palmyra was the site of a successful eradication of *R. rattus* (US Fish and Wildlife Service, 2011). The planning and implementation of the rat eradication required novel techniques, including direct baiting of the tree canopy, and two aerial broadcast applications, each at rates of 75 and 85 kg/ha, of bait containing brodifacoum (0.0025%) (Wegmann, et al., 2012). Ecotoxicology monitoring undertaken during and after the project detected residual brodifacoum in soil, water and biota (Pitt, et al., 2015). Sampling ceased 60 days after the bait application before undetectable levels of brodifacoum were reached (Pitt, et al., 2015). Resources to continue the monitoring were not secured until three years after the bait application for rat eradication, providing the opportunity to investigate longer-term persistence of brodifacoum within the Palmyra food web.

METHODS

Study site and animals

Palmyra Atoll (5°53' N, 162°05' W) is located at the northern end of the Line Islands in the Central Pacific Ocean. Palmyra is a wet atoll containing approximately 235 ha of emergent land primarily covered in thick rainforest. The atoll is an incorporated, unorganised territory of the United States that is managed in partnership by The Nature Conservancy (TNC) and the US Fish and Wildlife Service (USFWS). TNC's preserve includes Cooper/Menge (94.3 ha) and Barren (4.6 ha) islands. Most of the remaining emergent land is owned and managed by USFWS as Palmyra Atoll National Wildlife Refuge, which includes all marine habitats to 12 nm offshore.

Palmyra's islets support a regional flora that is typical of Central Pacific wet forests (Wester, 1985). Heavily influenced by the Intertropical Convergence Zone, Palmyra receives an average of 450 cm of rain each year. Palmyra is a refuge for 11 species of seabirds and is home to a robust community of land crabs comprised of nine species. Black rats (*Rattus rattus*) were inadvertently brought to Palmyra during WWII. In 2011, Palmyra's rat population was eradicated through two strategic applications of compressed-grain bait containing the second-generation anticoagulant rodenticide, brodifacoum, at 0.0025% (25 ppm) (Wegmann, et al., 2012). Pitt et al. (2015) collected and analysed fifty-one animal samples representing 15 species of birds, fish, reptiles, and invertebrates for brodifacoum residue out to 60 days after the initial bait application.

Environmental monitoring methodology

We followed the sampling methods outlined in Pitt et al. (2015) to assess brodifacoum residue concentrations three years after bait application in cockroaches

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(*Periplaneta* sp.), fiddler crabs (*Uca tetragonon*), hermit crabs (*Coenobita perlatus*), and geckos (*Lepidodactylus lugubris*). Limited time and resource restrictions did not allow sampling of black-spot sergeant fish (*Abudefduf sordidus*) or ants, as undertaken in 2011; however, we harvested mullet (*Moolgarda engeli*)), which were opportunistically collected as carcasses in 2011 following the eradication and their tissues were found to contain brodifacoum (Pitt, et al., 2015). All biological samples were collected at Palmyra Atoll between 4 and 19 June 2014. Biological samples were frozen immediately after collection.

Sampling site selection (Fig. 1) was determined by ease of access to the target species. All emergent land at Palmyra has relatively similar characteristics and vegetation and was treated with the same baiting prescription during the 2011 eradication campaign. We therefore assumed that site location would not be an influential factor in brodifacoum residue concentrations three years after bait was applied. Biological samples were collected at least 500 m from The Nature Conservancy's research station where rodenticide bait is maintained in bait stations for biosecurity when planes and ships arrive.

All biological samples were collected with gloved-hands and segregated in sterile sample bags. Captured hermit crabs were placed in a freezer (-4 C) for 24 hours and then removed from their gastropod shells and stored in sterile sample bags. Mullet were collected by dip-nets and fence-nets from several shoreline locations around Palmyra's central lagoon. Geckos and cockroaches were captured at night from the leaves of *Scaevola taccada* shrubs, and fiddler crabs were collected from lagoon flats at low tide. American Veterinarian Medical Association guidelines for euthanasia were followed with all collections. All samples were pooled (Table 1) to increase probability of detecting brodifacoum within the funding limits of this project and to ensure minimum amounts of sample material were provided for analysis (e.g. cockroach samples required two individuals to achieve the 2 g minimum for brodifacoum residue analysis). Samples were shipped frozen to US Department of Agriculture's National Wildlife Research Center (NWRC) in Fort Collins, Colorado, for brodifacoum residue analysis. Samples were prepared and analysed according to methods established by USDA NWRC for detection of brodifacoum in animal tissue, and these methods, as well as the laboratory conducting the analyses, were the same as those used in Pitt et al. (2015). Same-species pooled carcasses' samples were homogenised for analysis.

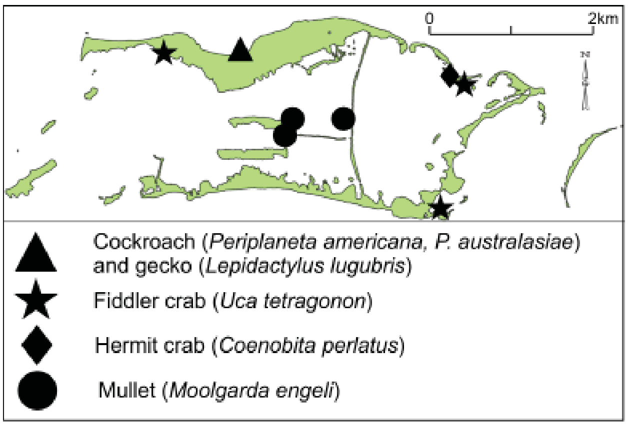

Fig. 1 Locations of sample collections that were used to investigate persistence of residual brodifacoum three years after the implementation of the 2011 eradication of rats from Palmyra Atoll.

Brodifacoum residue analysis methodology

The whole bodies of geckos and fish were homogenised. Cockroaches (whole bodies), as well as fiddler crabs and hermit crabs were homogenised in a liquid nitrogen freezer mill and 0.25 g of homogenate was placed into 25 ml glass tubes for further extraction and analysis following methods of Pitt, et al. (2015). Aliquots (0.5 g) of each homogenised gecko and fish sample were placed in MARS vessels for microwave extraction (Pitt, et al., 2015). Samples were clarified by centrifugation prior to HPLC analysis.

Brodifacoum analyses were performed with Agilent 1100 and 1200 HPLC systems (Pitt, et al., 2015). Brodifacoum concentrations were determined from the peak area ratio of brodifacoum to surrogate in each extracted sample and were compared to the average peak area ratio from replicate injections of a working standard. Samples with analytical concentrations above the linear range were re-diluted into the linear region.

RESULTS

We collected 44 pooled samples containing 121 total individuals (Table 1). Brodifacoum residues were not detected (detection levels reported in Table 1) in any of the pooled samples of mullet, geckos, cockroaches, hermit crabs, or fiddler crabs.

DISCUSSION

Ecotoxicology monitoring is uncommon for rodent eradication projects using rodenticides, but future projects are dependent on the collective knowledge gained from toxicological monitoring efforts. The Palmyra rat eradication used substantially higher rodenticide application rates compared to other rodenticide-based rodent eradication projects on islands and provided a unique opportunity to follow residue persistence in the environment over time. Brodifacoum residues were detected in soil, water and biota up to 60 days after the first aerial broadcast application (Pitt, et al., 2015) but were no longer detectable in the range of biota studied three years later, indicating rodenticides break down in this ecosystem over time. Resource availability did not allow complete repetition of the 2011 sampling, thus we chose to sample animals with known residue concentrations, as this had the most biologically useful outcome for management.

The use of second generation anticoagulant rodenticides can pose significant risks to non-target species (Howald, et al., 2007), particularly birds and mammals. However, knowledge gaps exist, particularly for taxa less sensitive to rodenticides, such as reptiles and invertebrates (Hoare & Hare, 2006). The distribution and longevity of rodenticide residue within a food web will be a function of rodenticide properties and how it is applied, environmental

Table 1 Biological samples analysed in 2014 for brodifacoum residue analysis following the 2011 eradication of rats from Palmyra Atoll. "Pooled" represents the number of individuals contained in each sample; "MLOD" is the mean level of brodifacoum detection.

Organism	Samples analysed	Pooled	MLOD (µg/g)
Mullet	9	2–3	0.013
Gecko	5	5	0.011
Cockroach	15	1–2	0.011
Hermit crab	5	3	0.0057
Fiddler crab	10	3	0.0057

compartments it ultimately resides within (e.g. soil, animals), open pathways to transfer residue (e.g. scavenger consumption of poisoned carcasses), and exposure to environmental conditions (e.g. temperature, precipitation, ultraviolet radiation, and fungi) that impact its persistence. Ultimately, the breakdown of rodenticides is believed to be accelerated in soil rich in organic matter with healthy populations of microbiological organisms. Different island ecosystems can be expected to have different timescales of residue longevity, and we expect our results will transfer most closely to other wet tropical atolls and low islands, rather than dry and/or temperate island environments.

Rodenticides are known to temporarily infiltrate the food web when undertaking rat eradications as happened with the Palmyra rat eradication. Brodifacoum residues were found in ocean water, soil, and marine and terrestrial biota within 60 days of the initial baiting, indicating diverse food web integration (Pitt, et al., 2015). Other studies document brodifacoum residues in various compartments of the food web after brodifacoum bait was applied to eradicate rats from islands (e.g. Dowding, et al., 1999; Masuda, et al., 2014; Masuda, et al., 2015; Pitt, et al., 2015; Siers, et al., 2015; Rueda, et al., 2016; Shiels, et al., 2017). Although few studies include long-term (>1 year) sampling for residues after brodifacoum application, there are three recent studies that report residues in animals two years (Rueda, et al., 2016), three years (Siers, et al., 2015), and four years (Shiels, et al., 2017) post-application. Brodifacoum persisted in lava lizards (*Microlophus duncanensis*) in the Galápagos Islands for 2.1 years (Rueda, et al., 2016), where liver residue levels were <0.200 μg/g (mean level of detection [MLOD] = 0.010 μg/g). On Wake Island in the Pacific Ocean, three years after rat eradication (Siers, et al., 2015), two out of 69 fish samples had detectable levels of brodifacoum in their livers, with concentrations 0.0038 μg/g and 0.0086 μg/g (MLOD = 0.0035 μg/g); the two fish were caught within an intermittently land-locked pond. Finally, on Desecheo Island, Puerto Rico, detectable levels of brodifacoum were found in seven animal samples (three endemic lizards, two black rats, one forest bird, and one cockroach sample [18 individuals]) four years after bait application (Shiels, et al., 2017). The range of brodifacoum residues in these seven samples was 0.027-0.134 μg/g (MLOD = 0.0054-0.012 μg/g, depending on species. Desecheo, Wake, and the Galápagos islands receive less rainfall than Palmyra (e.g. Desecheo = 1,020 mm/yr, Wake = 906 mm/yr; Pinzon, Galápagos = <1,100 mm/yr; Palmyra = 3,500 mm/yr), and this may contribute to the lack of detectable levels of brodifacoum in the Palmyra food web three years after bait application. We hypothesise that warmer and wetter environments, and soils with more diverse microbiological communities support microbiological processes breaking down residues faster. This remains an important research avenue, including decomposition experiments in a laboratory setting.

Undertaking eradications of invasive species from islands should only proceed where expected benefits outweigh expected costs (Broome, et al., 2014), including consideration of the environmental impacts of the method used (Empson & Miskelly, 1999). Potential non-target impacts were anticipated as part of the environmental impact assessment for the Palmyra rat eradication, but the decision to proceed was based on negative impacts ceasing shortly after the bait application and positive benefits accruing over a longer time-span (US Fish and Wildlife Service, 2011). Immediately following bait application, brodifacoum residues were detected within multiple levels of the food web, and were attributed to mortality of birds, fish, and crabs (Pitt, et al.., 2015). Our results show undetectable levels of residue three years later, suggesting this short-term impact is no longer present. Longer-term changes to native species populations following the removal of rat impacts are emerging, including increased seedling recruitment of several native tree species and the non-native coconut palm (*Cocos nucifera*), the elimination of a non-native mammal-biting mosquito population (*Aedes albopictus*), as well as the discovery of two new-to-Palmyra land crab species (*Geograpsus grayi* and *Ocypode cordimanus*). These short and long-term changes are consistent with management expectations, and the rat eradication has proven to be a baseline restoration activity to advance natural resource management goals.

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REFERENCES

Atkinson, I.A.E. (1985). 'The Spread of Commensal Species of *Rattus* to Oceanic Islands and their Effects on Island Avifaunas'. In*:* P.J. Moors (ed.) *Conservation of Island Birds.* pp. 35–81. Cambridge: International Council for Bird Preservation.

Brooke, M.d.L., Bonnaud, E., Dilley, B.J., Flint, E.N., Holmes, N.D., Jones, H.P., Provost, P., Rocamora, G., Ryan, P.G., Surman, C. and Buxton, R.T. (2017). 'Seabird population changes following mammal eradications on islands'. *Animal Conservation* 21(1): 3–12.

Broome, K., Cox, A., Golding, C., Cromarty, P., Bell, P. and McClelland, P. (2014). *Rat Eradication Using Aerial Baiting. Current Agreed Best Practice Used in New Zealand (Version 3.0).* Wellington, New Zealand: New Zealand Department of Conservation internal document.

Dowding, J.E., Murphy, E.C. and Veitch, C.R. (1999). 'Brodifacoum residues in target and non-target species following an aerial poisoning operation on Motuihe Island, Hauraki Gulf, New Zealand'. *New Zealand Journal of Ecology* 23: 207–214.

Empson, R.A. and Miskelly, C.M. (1999). 'The risks, costs and benefits of using brodifacoum to eradicate rats from Kapiti Island, New Zealand'. *New Zealand Journal of Ecology* 23(2): 241–254.

Hoare, J.M. and Hare, K.M. (2006). 'The impact of brodifacoum on non-target wildlife: Gaps in knowledge'. *New Zealand Journal of Ecology* 30: 157–167.

Holmes, N.D., Griffiths, R., Pott, M., Alifano, A., Will, D., Wegmann, A.S. and Russell, J.C. (2015). 'Factors associated with rodent eradication failure'. *Biological Conservation* 185: 8–16.

Howald, G., Donlan, C.J., Galvan, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21(5): 1258–1268.

Masuda, B.M., Fisher, P. and Jamieson, I.G. (2014). 'Anti-coagulant rodenticide brodifacoum detected in dead nestlings of an insectivorous passerines'. *New Zealand Journal of Ecology* 38: 110–115.

Masuda, B.M., Fisher, P. and Beaven, P. (2015). 'Residue profiles of brodifacoum in coastal marine species following an island rodent eradication'. *Ecotoxicology and Environmental Safety* 113: 1–8.

Jones, H.P., Holmes, N.D., Butchart, S.H.M., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P.E., Cuthbert, R.J., Ebbert, S., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R. and Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences* 113: 4033–4038.

Pitt, W.C., Berentsen, A.R., Shiels, A.B., Volker, S.F., Eisemann, J.D., Wegmann, A.S. and Howald, G.R. (2015). 'Non-target species mortality and the measurement of brodifacoum rodenticide residues after a rat (*Rattus rattus*) eradication on Palmyra Atoll, tropical Pacific'. *Biological Conservation* 185: 36–46.

Rueda, D., Campbell, K.J., Fisher, P., Cunninghame, F. and Ponder, J.B. (2016). 'Biologically significant residual persistence of brodifacoum in reptiles following invasive rodent eradication, Galapagos Islands, Ecuador'. *Conservation Evidence* 13: 38.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Shiels, A.B., Witmer, G.W., Samra, C., Moulton, R.S., Ruell, E.W., O'Hare, J.R., Eisemann, J.D., Volker, S.F. and Goldade, D.A. (2017). *Assessment of Bait Density, Bait Availability, and Non-target Impacts During an Aerial Application of Rodenticide to Eliminate Invasive Rats on Desecheo Island, Puerto Rico*. Final Report QA 2588. Ft Collins, CO: USDA, APHIS, WS, NWRC.

Siers, S.R., Shiels, A.B., Goldade, D.A., Volker, S.F., McAuliffe, T.W., Coad, H.L. and Pitt, W.C. (2015). *Wake Atoll Fish Tissue Sampling and Analysis Three Years After an Island-wide Rodenticide Application*. Final Report QA 2241. Hilo, HI: USDA, APHIS, WS, NWRC.

Tershy, B.R., Shen, K.-W., Newton, K.M., Holmes, N.D. and Croll, D.A. (2015). 'The importance of islands for the protection of biological and linguistic diversity'. *Bioscience* 65(6): 592–597.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?'. *Biological Invasions* 8(4): 863–891.

US Fish and Wildlife Service (2011). *Palmyra Atoll National Wildlife Refuge Rat Eradication Project Final Environmental Impact Statement*. Honolulu, Hawaii: US Fish and Wildlife Service.

Wegmann, A., Buckelew, S., Howald, G., Helm, J. and Swinnerton, K. (2011). 'Rat eradication campaigns on tropical islands: novel challenges and possible solutions'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 239–243. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Wegmann, A., Flint, E., White, S., Fox, M., Howald, G., McClelland, P., Alifano, A. and Griffiths, R. (2012). 'Pushing the Envelope in Paradise: A Novel Approach to Rat Eradication at Palmyra Atoll'. In: R.M. Timm (ed.) *Twenty Fifth Vertebrate Pest Conference*, pp. 48–53. Davis, CA: University of California.

'Island' eradication within large landscapes: the remove and protect model

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Abstract New Zealand has been the world leader in the eradication of invasive mammalian predators from offshore islands. Today, the focus for invasive predator management is shifting to larger landscapes; big inhabited islands or the mainland itself. The most cost-effective approach in the long term will be to eradicate the predators from those areas, ensuring permanent freedom for vulnerable and threatened native biodiversity to recover or be reintroduced. Island eradication technologies cannot always be employed on the mainland (e.g. aerial brodifacoum), so a new approach is required. Zero Invasive Predators Ltd (ZIP) is a not-for-profit research and development entity, established in New Zealand through public, private, and philanthropic funding, to pioneer a novel predator management model for landscape-scale application – a model known as 'Remove and Protect'. ZIP is developing the tools and technologies to both enable the complete removal of rats, possums, and stoats from large areas of mainland New Zealand, and then protect those areas from reinvasion. Among the innovations being tested is the 'virtual barrier', essentially converting large peninsulas into islands without the use of traditional predator fencing (which is expensive and impractical in some terrain); and a 'minimal infrastructure' detection system for automated early warning of any predator incursions. We review the transformative predator management model ZIP is developing and how it could help to pave the way towards large-scale predator-free landscapes.

Keywords: detection, invasive predators, mainland, *Rattus*, response, *Trichosurus vulpecula*, virtual barrier, 1080

INTRODUCTION / CONTEXT

New Zealand is a global biodiversity hotspot (Myers, et al., 2000), yet more than 3,000 native taxa are threatened or at risk of extinction (Hitchmough, 2013). It is generally agreed that there are three mammalian predator species that cause most of the ecological damage in New Zealand: possums (*Trichosurus vulpecula*), ship rats (*Rattus rattus*), and stoats (*Mustela erminea*) (Brown, et al., 2015). From here on, the term 'predators' refers to these three species plus Norway rats (*Rattus norvegicus*). The house mouse (*Mus musculus*) is specifically excluded as a predator in the context of this paper and is not a target species for ZIP. Aside from the estimated 25 million native birds they kill each year (Russell, et al., 2015), predators cost New Zealand hundreds of millions of dollars annually, both in terms of revenue lost and in control costs (Clout, 2011), and they impact the country's primary production base through the transmission of diseases such as bovine tuberculosis (Coleman & Caley, 2000).

New Zealand has an impressive track record in the eradication of invasive mammalian predators from offshore islands for the protection of native biodiversity. Since the first successful eradication in 1964 (Towns & Broome, 2003), 134 islands have been completely freed from invasive mammals (Parkes, et al., 2017a). Although costs vary widely from island to island, the initial eradication cost is in the order of NZ$300/ha (Parkes et al., 2017b); and the ongoing biosecurity surveillance costs of these islands typically ranges from NZ$17 to NZ$160/ha per annum (New Zealand Department of Conservation (DOC) unpublished data, 2017). These costs *exclude* incursion response. For example, the stoat incursion response on Kapiti Island in 2010–2011 cost approximately NZ$600,000 (NZ$305/ha) (King, et al., 2014). These predator-free islands are considered to be the 'jewels' of the conservation crown; however, they represent only 58,921 ha, or <0.01% of the land area of New Zealand (Parkes, et al., 2017a).

For most of the New Zealand mainland, where restricting the reinvasion of predators is currently not possible, the management model used is the ongoing suppression of predator populations. Currently the main tool used by the major predator management agencies (DOC, TB Free New Zealand, Regional Councils) for large scale (up to 100,000 ha) predator control is repeated pulsing of aerially applied sodium fluoroacetate (1080) toxin, typically every three to five years (Brown, et al., 2015; Elliott & Kemp, 2016). The current annualised cost of this is approximately NZ$10/ha (Brown, et al., 2015). The benefits of this technique are of limited duration without ongoing sustained control, because not all individuals are removed from the treatment area, and immigration is uncontrolled so predator populations are able to recover (Griffiths & Barron, 2016).

The alternative, ground-based predator control methods rely on either a knockdown of the resident predator population, followed by ongoing suppression to low levels, or seasonal control to realise biodiversity benefits (e.g. for the native bird breeding period). This work is relatively labour intensive (via trapping or toxins in bait stations) and is presently undertaken over areas of up to 50,000 ha (e.g. Murchison mountains stoat trapping; Hegg, et al., 2013). The current annualised cost of this work is in the order of NZ$25 to NZ$60/ha depending on the scale and intensity of the control efforts and target predator species (Brown, et al., 2015).

Predator exclusion fencing, a physical mesh fence with a solid steel capping, is also used to recreate eradication-like conditions on the mainland (colloquially, New Zealand's North and South Islands) by providing a physical barrier to halt reinvasion (Burns, et al., 2012). Predator fencing is scale-limited by terrain and cost, with the cost of recently constructed fences ranging from NZ$253–NZ$461/linear metre (Curnow & Kerr, 2017), with ongoing maintenance costs estimated to be 4% of capital costs per annum for the life of the fence (Norbury, et al., 2014), and eradication costs additional. Debate continues on the ecological, social and financial return on investment for predator fencing (Scofield, et al., 2011; Scofield & Cullen, 2012; Innes, et al., 2012; Norbury, et al., 2014).

To dramatically improve the status of New Zealand's biodiversity, a step change is required in the ability to manage predators, and the cost of doing so. The New Zealand Government has declared the goal of a predator-free New Zealand by 2050 (Cabinet, 2016). In order to

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achieve this ambitious goal, the country will need to heed the call of The Royal Society of New Zealand (2014), for urgent action to develop novel approaches and to improve existing tools to protect the country's environment and economy.

Remove and protect model

One novel approach being investigated is the 'remove and protect' model, entailing complete removal of predators from an area and then protection against reinvasion. In essence, this creates permanent 'island' eradications within large landscapes of the New Zealand mainland. A research and development entity, Zero Invasive Predators Ltd (ZIP; founded in 2015), has been established with the purpose of developing the 'toolbox' to enable this model.

The remove and protect model involves three streams of research and development:

Initial removal of target predators

The most common and most successful technique for island eradication has been the aerial application of the toxin brodifacoum (Howald, et al., 2007; Parkes, et al., 2011). However, the use of this technology in New Zealand is governed by a Code of Practice (Epro Ltd. 2006) that limits its current use to offshore islands and stock-free areas of the mainland behind predator fences – preventing its immediate application in the remove and protect model. As a result, eradicating predators on the New Zealand mainland will likely require new techniques to be developed or novel refinement of the application of existing tools – refer to Case Study 1 for one such example.

Defending a line to protect against reinvasion

Implementing a campaign of the scale of predator-free New Zealand by 2050 (Cabinet, 2016) will require the ability to divide the country up into manageable land parcels for progressive removal operations. Predator fencing has allowed small areas to be treated as 'islands' on the mainland but has limited application because of rugged terrain and/or social acceptance (Clapperton & Day, 2001; Burns, et al., 2012). Dividing up the country will require additional approaches; the creation of a virtual barrier is one such approach – refer to Case Study 2.

Detecting and removing invaders before they significantly impact on the predator-free area

Traditionally, in the island eradication context, biosecurity surveillance consists of intensive networks of passive devices to find individual invaders (Russell, et al., 2008). In order to ensure the remove and protect model is scalable, and to protect any significant predator-free investment, there is a need to develop a minimal infrastructure detection system that can facilitate timely incursion response before significant ecological damage is incurred – refer to Case Study 3 for detection concepts being explored.

Changing the cost model

Eradication is the most cost-effective methodology for predator management (Pascal, et al., 2008), so long as long-term biosecurity costs are manageable, as the upfront costs of removal only need to be found once. However, on the mainland, where reinvasion into management sites is typically not controllable, the most cost-efficient technique at present is to aim for predator suppression over as large a land area as affordable, in the knowledge that it will need to be repeated *ad infinitum* to maintain the gains achieved. In New Zealand, where a relatively modest budget for predator control (given the scale of the issue at hand) is largely static year-on-year, the cyclical pattern of suppression means that only a limited land area can be managed and that cannot expand without increased investment.

The remove and protect model seeks to change that cost structure. By treating blocks of land like island eradications, i.e. removing all predators and managing reinvasion to zero, those gains can be secured, and the predator management programme can be expanded to treat new land areas. Due to the greater expected biodiversity outcomes derived from complete predator absence in the long term (Ismar, et al., 2014; Towns, et al., 2016), i.e. a larger ecological return on investment, the initial management costs can be greater than those currently afforded for suppression, especially as they are a one off cost. However, for this cost structure to be feasible, the remove and protect model must achieve similar cost profiles to those of island or fenced sanctuary eradications in both the removal and maintenance phases. The initial targets ZIP is currently working to are: initial predator removal costs of NZ$100/ha (cf. NZ$300/ha for island eradications; Parkes, 2017b); NZ$200/m for installation of a virtual barrier (cf. NZ$253−NZ$461/m for predator fencing; Curnow & Kerr, 2017); and NZ$50/ha/annum for detection and response (cf., for example, NZ$160/ha per annum for biosecurity surveillance on Ulva Island; DOC unpublished data, 2017). All costs exclude Goods and Services Tax (GST).

A focussed approach: Zero Invasive Predators Ltd (ZIP)

The opportunity to establish a public-philanthropic partnership presented itself when the NEXT Foundation approached DOC to invest in 'transformative change' for conservation. In what is a first for DOC, the decision was made to 'spin out' of Government and establish ZIP as a limited liability company (with NEXT Foundation as the sole shareholder). Founded in 2015, the intention was that ZIP would be tightly focussed on the core challenge of developing a new model for predator management; the equivalent of taking a specialist research and development unit and sheltering it from the rest of a business until the problem is 'solved'. It was further considered that freedom from Government would provide the best environment in which to remain agile and innovative.

While ZIP has a business structure, it does not have commercial motives. Any self-generated Intellectual Property is held for New Zealand, effectively making it openly available to those in New Zealand who want to use or build upon it. The founding constitution confirms this 'not for profit' stance, with any products to be sold at the most accessible price point in New Zealand (while reserving the right to profit from international sales), with any profit to be reinvested in conservation, rather than returned as a dividend to shareholders. ZIP is also recognised as a Registered Charity by the Charities Commission (the governing body in NZ). This charitable status has aided in securing further philanthropic investment (beyond NEXT Foundation) as donations, which are tax deductible in New Zealand.

Some of the high-level goals of ZIP, such as removal of possums and a reduced reliance on cyclic toxin applications, have also attracted support from New Zealand dairy companies, who share those intentions (F. Eggleton, Fonterra Co-operative Group, pers. comm). This support includes non-shareholding investment in the research and development programme, thereby further enhancing the unique public-philanthropic-private investment positioning of ZIP.

Operating culture – try, sense, respond

Ecological systems are usually complex and therefore the development approach of ZIP is to 'try, sense, and respond'. Potential solutions are suggested, techniques and tools are rapid-prototyped and placed in the field as soon as possible, impacts are measured, and prototypes are refined as soon as required. The 'try, sense, respond' approach allows rapid learning about real world constraints, which in turn informs the next iteration of development and testing.

This operating style aims to recognise failure quickly, to expose what we don't know, and to maximise the return on effort and resources. Supporting this 'fast fail' approach, field trials of prototypes typically begin at small scale, i.e. less than five units, in the expectation that limitations will be exposed and the prototype redesigned. Once the prototype shows sufficient promise, the trial is scaled-up in stages, going from, for example, 50 to 100 units, then many hundreds of units, etc. to test if the statistical performance holds as the scale increases. Alternatively, if the prototype fails catastrophically at the small scale, and no practicable alternatives are found, the trial is shut down to minimise loss of investment.

This operating culture is strengthened by a diverse, highly-skilled team, purpose-built for research and development. Scientists and engineers co-design field trials and technologies, field rangers actively test prototypes, with timely data analysis by a specialist modeller. Input from all aspects of the team feeds into each step of the development process, enabling rapid evolution of the project. All team members spend time at the field site(s) to remain grounded in the challenge.

Development in the field

ZIP, under permission from DOC (the land manager), has established a 391 ha forested site at Bottle Rock Peninsula, Queen Charlotte Sound, Marlborough (41°06'30" S, 174°14'06" E) dedicated to field trialling the remove and protect system, and its component prototype parts. Remove and protect is well suited to peninsulas as they are easier to defend, with only one major exposed front (with the sea 'protecting' the remainder). Interception efforts can then be concentrated within a relatively small zone to protect a much larger area.

Bottle Rock Peninsula was selected as it offered the ideal initial size for rats and possums, and was a favourable shape (2 km narrow neck with a bulbous peninsula). Importantly, this peninsula is not a site of high biodiversity priority for DOC (unpublished data, 2015), therefore it is able to be manipulated without risk to vulnerable native species. However, it does enable a 'real world' assessment of new or modified technologies. [NB: the majority of the field trials carried out at Bottle Rock to date have excluded stoats on account of their home range size, mobility, and our current lack of sensitive detection devices rendering robust stoat research impracticable.]

Evaluation of the performance of the remove and defend model at Bottle Rock Peninsula uses a 'systems design' approach (Cabrera, et al., 2008), assessing the whole, as opposed to a reductionist approach which seeks to understand the role of the individual elements to explain the utility of the system. The goal is to prove the system works, not just some parts of it, hence multiple tools need to be tested simultaneously in the defence system. Individual considerations are secondary and are investigated by 'switching off' components to specifically test their relative impact on the system's performance.

REMOVE AND PROTECT CASE STUDIES

Case study 1: Removal – '1080 to Zero'

It is expected that an aerially applied tool will be required for the initial removal of predators at large-scale implementation sites. Some of the early work developing techniques for island eradications investigated sodium fluoroacetate (1080) as an option (McFadden & Towns, 1991; Moors, 1985). However, it was subsequently discounted because of its acute toxicity and the perception that some individuals of the target populations could detect it in the bait and avoid it (McFadden & Towns, 1991). There has been significant improvement since that work, namely prefeeding to increase toxicant uptake (Nugent, et al., 2011) and manufacturing quality control (Nugent, et al., 2010; Nugent, et al., 2012). Extensive use in suppression operations has refined aerial 1080 use, but those operations still do not remove all target individuals (Elliott & Kemp, 2016).

ZIP sought to test whether dual aerial 1080 operations, each using different bait (to overcome learnt aversion; Ross, et al., 2000) and coupled with multiple prefeed applications, could completely remove rats and possums. Success was deemed to be functional extinction. The thresholds for achieving functional extinction were set at ≤1 possum per 400 ha (OSPRI, 2014); and ≤1 rat per 100 ha (Innes, et al., 2011).

The trial was carried out on a 1,600 ha area (39°15'30" S, 174°07'45" E) on the north-eastern slope of Mt Taranaki. A 400 ha core, set back with a 1 km buffer to minimise reinvasion compromising the results (Griffiths & Barron, 2016), was intensively monitored for surviving rats and possums after treatment with toxin. The trial excluded stoats due to the scale being insufficient to account for stoat home range size and mobility (Murphy & Dowding, 1994; Murphy & Dowding, 1995).

Prior to commencing the trial, monitoring (using peanut-butter filled chew cards, self-manufactured using corflute supplied by Pest Control Research and Pic's peanut butter – Picot Productions Ltd) was deployed three times for between two and 10 nights using between 36 and 55 cards each time. The cards were placed every 50 metres on 2–3 randomly selected lines (of between 1.6 and 2 km in length) within the 400 ha core. The purpose of this monitoring was not to measure a relative abundance, but merely to confirm presence of target animals. 98% of total cards deployed were chewed by rats, 6% of total cards deployed were chewed by possums.

The first phase of baiting consisted of multiple prefeed baiting of non-toxic RS5, 6 g, cinnamon-masked cereal pellets (manufactured by Orillion, formerly Animal Control Products) applied by helicopter-slung bait-spreading bucket – at (on-ground application rates of) 4 kg/ha; 2 kg/ha (20 days later); 1 kg/ha (21 days later); 1 kg/ha (47 days later). Application of (on-ground rate) 4 kg/ha of RS5, 6 g, 0.15% 1080, cinnamon-masked cereal pellets (Orillion) followed 21 days later. Bait was flown with a 50% swath overlap, as per island eradication best practice (Broome, et al., 2014), to ensure no gaps in bait coverage. Baiting was intended to be completed in winter, when 1080 has been shown to be most effective (Veltman & Pinder, 2001; Gillies, et al., 2003); but adverse weather resulted in the toxin being applied on 1 December 2016.

In an effort to detect survivors, 835 chew cards were deployed on a 50 m × 100 m grid throughout the 400 ha core four nights after the toxin application, and checked every eight days, for a total of 42 days. In addition, 421 pre-weathered tracking tunnels installed on a 100 m × 100 m grid were baited 17 days into the detection period and maintained live until the same 42-day period post-toxin application had passed. Furthermore, 80 motion-activated cameras (Little Acorn, LTL5200 and LTL5300) were deployed in a 100 m × 100 m grid in the north-eastern corner of the ZIP block for the final 17 nights of the detection period to validate the performance of the other detection devices.

Functional extinction of possums was considered to be achieved, with only one possum detection (chew card) recorded across 36,430 detection nights across all applicable detection devices (chew cards and cameras). The same was not achieved for rats, with 42 detections (two chew cards; 25 tracking tunnels; 15 camera detections) recorded over 46,755 detection nights.

In light of the numbers of surviving rats, we attempted to individually test them for any learnt bait aversion (rather than undertake the second phase of toxic aerial baiting). Research by Morgan (2004) suggested that cereal pellets cannot overcome aversion if created by cereal pellets in the first place; however, that study did not include prefeeding. Morgan, in the same work, states that 'learnt food safety' (i.e. learnt through prefeeding) is a very strong behaviour once established. Ross et al. (2000) achieved 30% mortality in captive 1080 bait-shy possums when 'postfed' with cereal (compared with 0% of non-postfed possums). We sought to determine whether it is possible, in the wild, to overcome any bait aversion in the surviving rats through prefeeding with the different bait, even if it is cereal.

The 1,600 ha trial area was prefeed-baited twice, using non-toxic Wanganui #7, 6g, double orange-masked cereal pellets (Orillion) from a helicopter-slung bait-spreading bucket, seven days apart (58 and 65 days after the first toxin application). McGregor live-capture traps were set in areas of known detections and baited with a single Wanganui #7 0.15% 1080 6g double orange-masked cereal pellet (Orillion). Traps were baited in such a way that the rat had to interfere with the pellet to trigger the trap. Traps were in place for 270 trap nights across various detection sites.

Thirteen rats were caught that were deemed to be survivors based on the weight:age profile (Bentley & Taylor, 1965); animals that were very likely to have been present when the initial toxin application was carried out. Of those, six were found dead in the trap (following consumption of a lethal dose of the bait), while an additional two were alive but showed clear signs of toxicosis with bait consumed (with death expected). The remaining five animals were all alive and were subsequently euthanised. While those rats found alive suggest some level of aversion, the trap itself may have contributed to the aversion once triggered, or alternatively they may have received a sub-lethal dose and did not return to the bait. It is expected that some rats did not encounter the live capture traps or chose to avoid them (and the bait within).

If the second aerial toxic baiting had been carried out, the total cost of the novel prescription (including all prefeed and toxic baiting applications) is estimated at approximately NZ$90/ha, excluding costs associated with gaining regulatory approvals. There is potential for this cost to reduce further with economies of scale and reduced prefeed applications.

ZIP retested the hypothesis in a trial on the West Coast of the South Island during the second half of 2017. After the first phase of baiting (two prefeed applications, and one toxin application using Wanganui #7 0.15% 1080 6g double orange-masked cereal pellet (Orillion)), zero rats and possums were detected over 83,410 detection nights across 55 days post-toxin application (unpublished data). The trial was deemed a success, and ended here.

Case study 2: Protect – the 'virtual barrier'

The virtual barrier is a system that aims to exclude 99% of rats, and 95% of possums that attempt to enter a protected area. The virtual barrier being tested across the 2 km neck at Bottle Rock Peninsula consists of multiple defence lines, 100 m apart, comprising kill (for rats) and live capture (for possums) traps only, with no toxins currently deployed in the system. Devices are placed at high intensity along each defence line, one every 10 m, based on the assumption that this spacing would 'guarantee' no animals could breach the barrier without encountering a device, i.e. if the target animal is on the ground it is never more than five metres from a device as it passes through a line. Whether they choose to interact with that device is another matter entirely!

Possums

The most effective virtual barrier for possums tested to date consisted of four lines of leg hold traps (PCR #1, Pest Control Research) running across the peninsula and a 400 m long line of leg hold traps running along the central, prominent ridge through the barrier. The leg hold traps are set in a custom-made platform raised 1.2 metres above the ground (to avoid non-target captures of weka, *Gallirallus australis*, a ground dwelling endemic rail). The traps are visually lured with a plain white corflute card (Connovation Ltd) nailed to the tree approximately 30 cm above the platform. Each platform has a wooden ramp attached, at 60° to the horizontal. In addition to preventing weka access, alternating trials by ZIP have shown that ramps improve trap effectiveness by 18% (95% C.I. [2.5%, 29%]) compared with non-ramped traps.

Traditionally, live-capture leg hold traps must be physically inspected by the trapper every day in order to comply with New Zealand animal welfare legislation. ZIP has developed an automated, remote reporting system that uses a magnetically switched trap transmitter to advise that a trap has been sprung, via a 433 MHz 'daisy chain' and the Iridium satellite network. To date (May 2017), the remote reporting system has been in service for more than 580,000 trap nights and has remotely reported over 500 possum captures – there has not been a single false negative in this time. In conjunction, the NZ Ministry for Primary Industries has developed industry guidelines to allow the automated reporting of live-capture traps, while conforming to animal welfare standards as required by law (MPI, 2016). This innovation has reduced the labour cost of servicing the traps by 95%, with only sprung traps needing to be checked by the trapper.

During the period from 26 November 2016 to 17 May 2017, the virtual barrier caught 127 possums, with at least 11 possums breaching the barrier; i.e. 8% 'leakage' (95% confidence interval, [4%, 14%]). Leakage was determined from the number of possums killed in the protected area (beyond the barrier), using leg hold traps, set up as per the barrier, but placed on a one per 50 ha density, divided by the total number that attempted to breach the barrier (number killed in the barrier plus number killed beyond it). In addition, a detection network of 554 chew cards (self-filled as described in the removal case study), serviced every three weeks, confirms the ongoing absence or presence of possums in the protected area. On average, approximately 18 possums/month attempted to cross the 2 km wide barrier, with 1.5 possums/month succeeding. Improvements to the system have been identified, and therefore future versions of the barrier are expected to approach the target of ≤ 5% leakage.

Ship rats

The current virtual rat barrier at Bottle Rock consists of six lines of 'Tun200s' (two DOC200 single action stainless steel kill traps (CMI Springs), in custom built 'run-through tunnel' wooden trap box). The wooden tunnels have a 72 mm diameter entrance hole and 265 mm long tunnel leading to the kill plates from both ends, to avoid non-target captures of weka which cannot fit inside the entrance hole nor stretch out to reach the traps themselves (currently <1 kill every 35,000 trap nights).

From 26 June to 26 October 2015, the virtual barrier caught 160 ship rats, with at least nine rats breaching the barrier; i.e. 5% leakage (95% confidence interval, [2.5%, 10%]). Leakage was estimated as described for possums in the previous section (e.g. number of rats killed on 100 m x 60 m grid of single-set DOC150 kill traps (CMI Springs) in 'standard' wooden boxes, placed throughout the peninsula beyond the barrier), in conjunction with the detection network of chew cards (as described above for possums) confirming the absence or presence of rats in the protected area. On average, approximately 40 ship rats/month attempted to cross the 2 km wide barrier, with two rats per month succeeding. All Tun200 traps were lured with peanut butter (Goodnature Ltd) during this period. We found no evidence to suggest that the effectiveness of identically lured, multiple lines of Tun200 traps declined with repeated presentation (effectiveness 40%, 95% C.I [33%, 46%], for all Tun200 lines treated as samples from the same population, that is irrespective of the line placement).

A variety of alternative food lures have subsequently been trialled including Nutella (Ferrero Australia Pty Ltd), Colby cheese (Mainland Ltd), milk chocolate (J.H. Whittaker and Sons Ltd), and peanut butter (Goodnature Ltd, and Pic's - Picot Productions Ltd). These lures performed similarly and intercepted on average 36% (95% C.I. [33%, 39%]) of rats, as measured by the percentage of rats that breached each line.

Costs of the barrier

Including the cost of track cutting and installation, the current capital cost of a multiple line, ship rat and possum virtual barrier at Bottle Rock Peninsula is approximately NZ$250/m (excl. GST). This cost is for a 20-year life, and includes device replacement, remote reporting system, and an automated lure dispenser (in development to further reduce labour costs).

The annual operating cost is approximately NZ$20/m (8% of capital cost).

Case study 3: Detection – a 'minimal infrastructure' system

Ship rats

Considerable effort has gone into understanding the exploratory behaviour of invading rats in predator-free spaces, with substantial individual variation identified in the roaming behaviour (Russell, et al., 2005; Russell, et al., 2008; Russell, et al., 2010; Innes, et al., 2011). Not unexpectedly, the majority of this work has been focussed on the individual, as current biosecurity detection systems are tailored towards intensively targeting the individual invader.

ZIP is conceptualising an alternative approach that looks beyond the individual, and rather focusses on the emergent population (if it happens). So long as the incursion events are infrequent, if the invading rat is alone and non-pregnant, then the scale of their individual impact is expected to be small and impacts only begin to have significance once a new population emerges (Norbury, et al., 2015; Elliott & Kemp, 2016). This is the point of intervention ZIP proposes to target.

Targeting the first generation (Generation One) of a pregnant female provides up to 11 individuals, 10 juveniles plus mother (Innes, 2005) to trigger detection devices, rather than the sole invader, greatly increasing the chances of interaction. Furthermore, the anticipated dispersal footprint of Generation One is likely to lend itself to a minimal infrastructure network spacing (perhaps one detection device every 20 ha, based on emerging data from ZIP trials such as that below). This network could be further tailored to be predominantly coastal and waterway biased, to maximise the probability of encounter. In addition, we estimate that we could have up to 100 days to detect and remove the first generation of invaders, before those juveniles reach sexual maturity and begin breeding themselves (based on reproductive biology; Innes, 2005). Conversely, this approach will require bigger treatment areas to remove the entire emerging population. The response could well be aerially based, rather than the ground-based responses traditionally deployed for island incursions.

A ZIP field trial is currently underway (during the drafting of this paper) at the confluence of the Jackson and Arawhata Rivers, South Westland (44°03'00" S, 168°43'32" E) whereby a mother ship rat and her offspring have been released into an area of very low rat abundance to observe their dispersal footprint. Early indications, based on the distance between release point and subsequent trap capture points, are that some individual offspring dispersed at least 650 m from the natal den location by the time they were 86 days old.

If the concept works, the capital cost of installing this system today would be NZ$20/ha. The annual surveillance cost would be NZ$4/ha (using an automated reporting kill trap as the 'sentinel' detection device), with an annual response cost of NZ$5/ha (assuming a leakage rate of 0.5%).

Possums

Possums, once isolated, roam over considerable ranges, in the order of 50–100 ha (Sweetapple & Nugent, 2009; OSPRI, 2014), presumably looking for other possums. If possum incursions are infrequent, their slow breeding rates (Cowan, 2005) and curiosity (Carey et al., 1997) suggest that delayed detection and response may be all that is necessary to prevent possum re-establishment.

ZIP is currently trialling a minimal 'lethal detection' network for possums at Bottle Rock Peninsula. Six leg-hold traps, deployed as in the virtual barrier (excluding ramp) but spaced at approximately one per 50 ha, have been established beyond the virtual barrier. In the 12 months since its deployment, this network has prevented possum reestablishment; with 17 possums caught to June 2017 (and no sustained detections on the 'background' chew card network, as described in case study 2). The capital cost of installing this system today would be NZ$10/ha, with a current operating cost of detection and response of approximately NZ$5/ha/annum.

Automated reporting system

To support these minimal infrastructure detection networks, development of an automated system for near real-time updates on the status of remove and protect sites is continuing. ZIP has already developed the ability to use daisy chain communication for short range data transmission, e.g. trap lines in a barrier setting. However, a landscape scale network will require a different transmission technology – one that can transmit reliably over large distances, in rugged or forested terrain (Jones, et al., 2015). Recent advances in the international telecommunications industry are seeing the emergence of low powered, long range radio technology (LoRa). A small number of sensitive receiving stations allows the use of many battery-powered transmitters across a landscape. LoRa, used in combination with satellite-based communications, is likely to be the platform technology on which to build an incursion notification system for these remote networks.

CONCLUSION

The New Zealand Government has announced the goal of being predator-free by 2050. Momentum is building on this goal, with the Predator Free 2050 Ltd company established with a board of directors to guide strategic investment into projects of significance (Anon., 2016). While New Zealand has an internationally enviable track record in island eradications and developed the predator fenced sanctuary approach, these methodologies cannot be scaled on the mainland.

It is widely acknowledged that new technologies, along with a shift in operating model and cost structure, will be required to completely eradicate predators from the mainland. Such a shift from the suppression paradigm could utilise the remove and protect model, where peninsulas are able to be converted into 'islands' for eradication operations. Zero Invasive Predators (ZIP), a not-for-profit research and development company founded in 2015, is helping to develop the techniques required to enable this model on the mainland.

Further trials are underway to use a novel prescription of dual aerial 1080 operations to drive initial removal at a cost of less than NZ$100/ha (with no more than two prefeed applications per toxin application). In-forest capability exists now to intercept over 95% of all rats and possums using a virtual barrier at a capital cost of approximately NZ$250/m and an annual operating cost of less than NZ$40/m. The initial testing of a minimal infrastructure detection system shows promising signs of success. Large social strides are still required to make predator-free New Zealand a reality, but the first tentative technical steps are being taken now.

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REFERENCES

Anon. (2016). *Predator Free 2050 Ltd. Board Appointed.* Press release 30 November 2016. Wellington: Office of the Minister of Conservation.

Bentley, E.W. and Taylor, E.J. (1965). 'Growth of laboratory-reared ship rats (*Rattus rattus* L.)'. *Annals of Applied Biology* 55: 193–205.

Broome, K., Cox, A., Golding, C., Cromarty, P., Bell, P. and McClelland, P. (2014). *Rat Eradication Using Aerial Baiting: Current Agreed Best Practice Used in New Zealand (Version 3.0)*. Wellington: Department of Conservation.

Brown, K., Elliott, G., Innes, J. and Kemp, J. (2015). *Ship Rat, Stoat and Possum Control on Mainland New Zealand: An Overview of Techniques, Successes and Challenges.* Wellington: Department of Conservation.

Burns, B., Innes, J. and Day, T. (2012). 'The Use and Potential of Pest-proof Fencing for Ecosystem Restoration and Fauna Conservation in New Zealand'. In: M. Somers and M. Hayward (eds.) *Fencing for Conservation: Restriction of Evolutionary Potential or a Riposte to Threatening Processes?* pp. 65–90. New York: Springer-Verlag.

Cabinet (2016). *Accelerating Predator Free New Zealand*. CAB-16-MIN-0335. Wellington: Cabinet Economic Growth and Infrastructure Committee.

Cabrera, D., Colosi, L. and Lobdell, C. (2008). 'Systems thinking'. *Evaluation and Program Planning* 31: 299–310.

Carey, P.W., O'Connor, C.E., McDonald, R.M. and Matthews, L.R. (1997). 'Comparison of the attractiveness of acoustic and visual stimuli for brushtail possums'. *New Zealand Journal of Zoology* 24: 273–276.

Clapperton, B.K. and Day, T.D. (2001). 'Cost-effectiveness of exclusion fencing for stoat and other pest control compared with conventional control'. *DOC Science Internal Series* 14: 5–19.

Clout, M.N. (2011). 'Ecological and Economic Costs of Alien Vertebrates in New Zealand'. In: D. Pimentel (ed.) *Biological Invasions: Economic and Environmental Costs of Alien Plant, Animal, and Microbe Species*, pp. 283–291. Boca Raton: CRC Press.

Coleman, J. and Caley, P. (2000). 'Possums as a Reservoir of Bovine TB'. In: T. L. Montague (ed.) *The Brushtail Possum. Biology, Impact and Management of an Introduced Marsupial*, pp. 92–104. Lincoln: Manaaki Whenua Press.

Cowan, P.E. (2005). 'Brushtail Possum'. In: C.M. King (ed.) *The Handbook of New Zealand Mammals*, pp. 56–80. Auckland: Oxford University Press.

Curnow, M. and Kerr, G.N. (2017). *Predator Free Banks Peninsula: Scoping Analysis*. LEaP Research Report No. 44. Lincoln: Lincoln University.

Elliott, G. and Kemp, J. (2016). 'Large-scale pest control in New Zealand beech forests'. *Ecological Management and Restoration* 17: 200–209.

Epro Ltd. (2006). *Code of Practice: Aerial and Hand Broadcast of Pestoff Rodent Bait 20R for the Intended Eradication of Rodents from Specified Areas of New Zealand*, Wellington: New Zealand Food Safety Authority.

Gillies, C., Campbell, J., Marsh, N. and Gembitsky, M. (2003). 'Seasonal Differences in Bait Acceptance by Forest-dwelling Rats Following Simulated Aerial 1080 Possum Control Operations in New Zealand: Interim Results'. In: G. Singleton, L. Hinds, C. Krebs and D. Spratt (eds.) *Rats, Mice and People: Rodent Biology and Management. ACIAR Monograph 96*, pp. 343–345. Canberra: Australian Centre for International Agricultural Research.

Griffiths, J.W. and Barron, M.C. (2016). 'Spatiotemporal changes in relative rat (*Rattus rattus*) abundance following large-scale pest control'. *New Zealand Journal of Ecology* 40: 371–380.

Hegg, D., MacKenzie, D.I. and Jamieson, I.G. (2013). 'Use of Bayesian population viability analysis to assess multiple management decisions in the recovery programme for the endangered takahe *Porphyrio hochstetteri*'. *Oryx* 47: 144–152.

Hitchmough, R. (2013). 'Summary of Changes to the Conservation Status of Taxa in the 2008–11 New Zealand Threat Classification System Listing Cycle'. *New Zealand Threat Classification Series 1*. Wellington: Department of Conservation.

Howald, G., Donlan, C.J., Galvan, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Innes, J. (2005). 'Ship Rat'. In: C. M. King (ed.) *Handbook of New Zealand Mammals*, pp. 187–203. Auckland: Oxford University Press.

Innes, J., Watts, C., Fitzgerald, N.L., Thornburrow, D., Burns, B., MacKay, J. and Speedy, C. (2011). 'Behaviour of invader ship rats experimentally released behind a pest-proof fence, Maungatautari, New Zealand'. In: C.R. Veitch, M.N. Clout, and D.R. Towns (eds.) *Island invasives: eradication and* management, pp. 437–440. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Innes, J., Lee, W.G., Burns, B., Campbell-Hunt, C. and Watts, C. (2012). 'Role of predator-proof fences in restoring New Zealand's biodiversity: a response to Scofield, et al. (2011)'. *New Zealand Journal of Ecology* 36: 232–238.

Ismar, S.M.H., Baird, K.A., Gaskin, C.P., Taylor, G.A., Tennyson, A.J.D., Rayner, M.J., Bettesworth, D., Fitzgerald, N., Landers, T.J. and Imber, M.J. (2014). 'A case of natural recovery after the removal of invasive predators – community assemblage changes in the avifauna of Burgess Island'. *Notornis* 61: 188–195.

Jones, C., Warburton, B., Carver, J. and Carver, D. (2015). 'Potential applications of wireless sensor networks for wildlife trapping and monitoring programs'. *Wildlife Society Bulletin* 39: 341–348.

King, C.M., Veale, A., Patty, B. and Hayward, L. (2014). 'Swimming capabilities of stoats and the threat to inshore sanctuaries'. *Biological Invasions* 16: 987–995.

McFadden, I. and Towns, D. (1991). *Eradication Campaigns Against Kiore (*Rattus exulans*) on Rurima Rocks and Korapuki, Northern New Zealand. Science and Research Internal Report No 97.* Wellington: Department of Conservation.

Moors, P.J. (1985). 'Eradication campaigns against *Rattus norvegicus* on the Noises Islands, New Zealand, using brodifacoum and 1080'. *ICBP Technical Publication* 3: 143–155.

Morgan, D.R. (2004). 'Maximising the Effectiveness of Aerial 1080 Control of Possums (*Trichosurus vulpecula*)'. PhD thesis. Lincoln: Lincoln University.

MPI (2016). *Remote Monitoring of Live Capture Traps for Vertebrates: Guidelines for Good Practice*. Wellington: Ministry for Primary Industries.

Murphy, E.C. and Dowding, J.E. (1994). 'Range and diet of stoats (*Mustela erminea*) in a New Zealand beech forest'. *New Zealand Journal of Ecology* 18: 11–18.

Murphy, E.C. and Dowding, J.E. (1995). 'Ecology of the stoat in *Nothofagus* forest: home range, habitat use and diet at different stages of the beech mast cycle'. *New Zealand Journal of Ecology* 19: 97–109.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B. and Kent, J. (2000). 'Biodiversity hotspots for conservation priorities'. *Nature* 403: 853–858.

Norbury, G., Hutcheson, A., Reardon, J. and Daigneault, A. (2014). 'Pest fencing or pest trapping: a bio-economic analysis of cost-effectiveness'. *Austral Ecology* 39: 795–807.

Norbury, G.L., Pech, R.P., Byrom, A.E. and Innes, J. (2015). 'Density-impact functions for terrestrial vertebrate pests and indigenous biota: guidelines for conservation managers'. *Biological Conservation* 191: 409–420.

Nugent, G., Morgan, D., Clayton, R. and Warburton, B. (2010). 'Improving the efficacy of aerial poisoning of brushtail possums (*Trichosurus vulpecula*) through reduced fragmentation of bait'. *International Journal of Pest Management* 57: 51–59.

Nugent, G., Warburton, B., Thomson, C., Sweetapple, P. and Ruscoe, W.C. (2011). 'Effect of prefeeding, sowing rate and sowing pattern on efficacy of aerial 1080 poisoning of small-mammal pests in New Zealand'. *Wildlife Research* 38: 249–259.

Nugent, G., Twigg, L.E., Warburton, B., McGlinchy, A., Fisher, P., Gormley, A.M. and Parkes, J.P. (2012). 'Why 0.02 %? A review of the basis for current practice in aerial 1080 baiting of rabbits in New Zealand'. *Wildlife Research* 39: 89–103.

OSPRI (2014). *Our First Year|2013/2014 OSPRI Annual Report and TBfree Research and Disease Control Report*. Wellington: OSPRI.

Parkes, J., Fisher, P. and Forrester, G. (2011). 'Diagnosing the cause of failure to eradicate introduced rodents on islands: brodifacoum versus diphacinone and method of bait delivery'. *Conservation Evidence* 8: 100–106.

Parkes, J.P., Nugent, G., Forsyth, D.M., Byrom, A.E., Pech, R.P., Warburton, B. and Choquenot, D. (2017a). 'Past, present and two potential futures for managing New Zealand's mammalian pests'. *New Zealand Journal of Ecology* 41: 151–161.

Parkes, J.P., Byrom, A. and Edge, K.-A. (2017b). 'Eradicating mammals on New Zealand island reserves: what is left to do?' *New Zealand Journal of Ecology* 41: 1–8.

Pascal, M., Lorvelec, O., Bretagnolle, V. and Culioli, J-M. (2008). 'Improving the breeding success of a colonial seabird: a cost-benefit comparison of the eradication and control of its rat predator'. *Endangered Species Research* 4: 267–276.

Ross, J.G., Hickling, G.J., Morgan, D.R. and Eason, C.T. (2000). 'The role of non-toxic prefeed and postfeed in the development and maintenance of 1080 bait shyness in captive brushtail possums'. *Wildlife Research* 27(1): 69–74.

Royal Society of New Zealand (2014). 'Challenges for Pest Management in New Zealand'. *Emerging Issues March 2014.*

Russell, J.C., Towns, D.R., Anderson, S.H. and Clout, M.N. (2005). 'Intercepting the first rat ashore'. *Nature* 437: 1107.

Russell, J.C., Beaven, B.M., MacKay, J.W.B., Towns, D.R. and Clout, M.N. (2008). 'Testing island biosecurity systems for invasive rats'. *Wildlife Research* 35: 215–221.

Russell, J.C., McMorland, A.J.C. and MacKay, J.W.B. (2010). 'Exploratory behaviour of colonizing rats in novel environments'. *Animal Behaviour* 79: 159–164.

Russell, J.C., Innes, J.G., Brown, P.H. and Byrom, A.E. (2015). 'Predator-free New Zealand: Conservation country'. *BioScience* 65: 520–525.

Scofield, R.P., Cullen, R. and Wang, M. (2011). 'Are predator-proof fences the answer to New Zealand's terrestrial faunal biodiversity crisis?' *New Zealand Journal of Ecology* 35: 312–317.

Scofield, R.P. and Cullen, R. (2012). 'Fenced sanctuaries need critical evaluation: a reply to Innes, et al. (2012)'. *New Zealand Journal of Ecology* 36: 239–242.

Sweetapple, P. and Nugent, G. (2009). 'Possum demographics and distribution after reduction to near-zero density'. *New Zealand Journal of Zoology* 36: 461–471.

Towns, D.R. and Broome, K.G. (2003). 'From small Maria to massive Campbell: forty years of rat eradications from New Zealand islands'. *New Zealand Journal of Zoology* 30: 377–398.

Towns, D.R., Borrelle, S.B., Thoreson, J., Buxton, R.T. and Evans, A. (2016). 'Mercury Islands and their role in understanding seabird island restoration'. *New Zealand Journal of Ecology* 40: 235–249.

Veltman, C.J. and Pinder, D.N. (2001). 'Brushtail possum mortality and ambient temperatures following aerial poisoning using 1080'. *Journal of Wildlife Management* 65: 476–481.

Multi island, multi invasive species eradication in French Polynesia demonstrates economies of scale

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Abstract Eradication of invasive vertebrates on islands has proven to be one of the most effective returns on investment for biodiversity conservation. To recover populations of the critically endangered Polynesian ground dove (*Gallicolumba erythroptera*), the endangered white-throated storm-petrel (*Nesofregetta fuliginosa*), the endangered Tuamotu sandpiper (*Prosobonia cancellata*) as well as other native plant and animal species, a project was undertaken to eradicate five species of invasive alien vertebrates: Pacific rat (*Rattus exulans*), ship rat (*R. rattus*), feral cat (*Felis catus*), rabbit (*Oryctolagus cuniculus*) and goat (*Capra hircus*), on six islands spanning 320 km of open ocean in the Tuamotu and Gambier Archipelagos of French Polynesia. Using a ship to deliver supplies and equipment, a helicopter for offloading and bait application, and ground teams for follow up trapping and hunting, invasive vertebrates were successfully removed from five of the six islands. Pacific rats survived at one site. The project was planned and executed by a partnership consisting of international and local conservation NGO's, working together with local communities. Combining the different eradication operations into one expedition added complexity to project planning and implementation and increased the risk of the operation failing on any one island but generated greater returns on investment allowing six islands to be targeted at significantly less cost than if each island had been completed individually. An extensive and thorough planning effort, effective relationships with local stakeholders and communities, a good operational strategy and a partnership of stakeholders that each brought complementary capacities to the project contributed to its success.

Keywords: cat, conservation, goat, rabbit, rat, restoration, threatened species recovery

INTRODUCTION

The removal of alien species from islands, especially invasive vertebrates, offers one of the best returns on investment for the protection of indigenous biodiversity (Donlan & Wilcox, 2008; Genovesi, 2011). There is now a growing list of island species no longer regarded as endangered because a key invasive species threat has been lifted (Russell, et al., 2016). The San Nicolas island night lizard (*Xantusia riversiana*) (Rice & Clark, 2016), the Seychelles magpie robin (*Copsychus sechellarum*) (Burt, et al., 2016) and the northern tuatara (*Sphenodon punctatus*) (Towns, et al., 2016) are just three examples of the many species whose threat status has been downgraded to a more secure category as a consequence (IUCN, 2010).

Eradication projects can be expensive (Simberloff, 2002). The remote nature of many islands and the necessity to target every individual within a population requires extensive planning effort, meticulous execution (Cromarty, et al., 2002) and resourcing that often exceeds the means of a single organisation. For many island nations, eradication projects are simply unaffordable and for some projects, the cost may exceed the annual environmental expenditure of an entire country.

French Polynesia is an overseas collectivity (political unit) of the French Republic. It is composed of 118 geographically dispersed islands and atolls scattered over an expanse of more than 5,030,000 km^2 in the South Pacific Ocean. Like many other tropical island archipelagos, French Polynesia is biologically rich and its remoteness has led its flora and fauna to be characterised by high levels of endemism (Gillespie, et al., 2008; Meyer & Butaud, 2009). Sixty three percent of its plants and 72% of its birds are found nowhere else (Gillespie, et al., 2008; Meyer & Butaud, 2009). As witnessed elsewhere, French Polynesia has been severely affected by habitat loss and invasive species. Nineteen of its bird species have become extinct since the 16th century and of the 25 surviving endemic birds, 18 are listed as threatened and five as critically endangered (Zarzoso-Lacoste, 2013).

Invasive vertebrates are widely considered the most significant threat to French Polynesia's avifauna (Zarzoso-Lacoste, 2013). Interventions, to remove invasive vertebrates, could be made to improve security for many species. However, investment within the collectivity for the management of invasive alien species remains small and a national invasive species strategy has not yet been developed. The collectivity does not appear to have the financial mechanisms to undertake vertebrate eradications and outside financial support will be required if species extinctions are to be avoided.

In 2015, four species of invasive alien vertebrates, Pacific rat (*Rattus exulans*), ship rat (*R. rattus*), feral cat (*Felis catus*) and rabbit (*Oryctolagus cuniculus*), were successfully removed from five of six islands spanning 320 km of open ocean in the Tuamotu and Gambier Archipelagos of French Polynesia. The project failed to remove rats from one project site and completion of goat (*Capra hircus*) eradication from another was delayed until 2017.

Here we describe the methods used to remove invasive vertebrates from the project sites and the logistics associated with the project. We explain how cost efficiencies were gained by combining operations and targeting multiple islands and define how the project partnership was instrumental to the project's success.

METHODS

Site description

Six islands in the south-east of French Polynesia were targeted for the removal of invasive vertebrates (Fig. 1). These included the two atolls of Vahanga and Tenania (Tenarunga) that, together with Tenararo and Matureivavao, make up the Acteon Island Group. Vahanga and Tenararo are identified as a Key Biodiversity Area (Atherton, 2007) and as an Important Bird Area (Raust & Sanford, 2007). Tenararo is one of four islands in French Polynesia never

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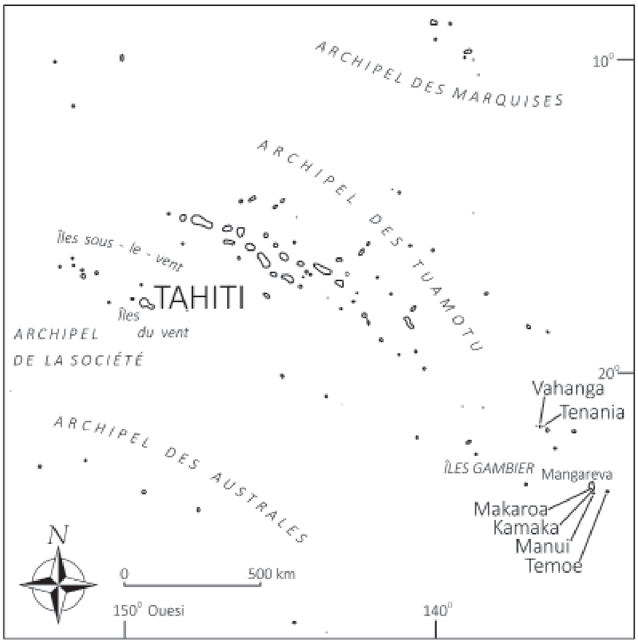

Fig. 1 Location of the six project sites within French Polynesia.

to have had invasive vertebrates and the atoll remains a stronghold for the critically endangered Polynesian ground dove and the endangered Tuamotu sandpiper as well as other native species (Blanvillain, et al., 2002). The operation also included Makaroa, Kamaka and Manui, three of the higher elevation islands that form Mangareva atoll complex and Temoe atoll, which lies to the south of Mangareva. Makaroa, Kamaka and Manui, together with the pest-free Motu Teiku, are classified as an IBA (Raust & Sanford, 2007). The endangered white-throated storm petrel breeds on both Manui and Motu Teiku.

All three of the atolls included in the operation were planted with coconut (*Cocos nucifera*) and used historically for copra production although only Tenania continues to be used for this purpose. Consequently, although areas of indigenous vegetation remain, *C. nucifera* dominates many of the forested parts of the atolls. Makaroa, Kamaka and Manui have also been extensively modified by burning, and the introduction of herbivores such as goats and rabbits. At the time of the project, little ground cover existed on Makaroa and large areas of Kamaka and Manui were covered in the invasive molasses grass (*Melinus minutiflora*). Of the six targeted sites, only the islet of Kamaka in the Gambier group is permanently inhabited. The atoll of Tenania in the Acteon group is occupied for part of the year for copra harvesting and Temoe is regularly visited by local fisherman. Table 1 summarises the general characteristics of the six sites.

Project feasibility and planning

Planning for rat eradication on Vahanga began in 2006 after a previous attempt undertaken in 2000–2001 was confirmed as having failed (Pierce, et al., 2006). Research was completed on the atoll to quantify the impact of terrestrial crabs on rodent bait availability, assess bait uptake by rats and quantify the effort required to hand broadcast bait across Vahanga (Griffiths, et al., 2011). Following this, an operational plan for rat eradication on the island was prepared (Broome, et al., 2011), but lack of funding delayed the project's implementation. A feasibility assessment for the removal of rats from Kamaka and Makaroa, completed in 2008 (Faulquier, 2008), stipulated the need for a helicopter due to the steep topography of these sites.

Following several high profile rat eradication failures on tropical islands, a global review of eradication methods was undertaken in 2013 to increase success rates (Russell & Holmes, 2015). New best practice guidelines were published, recommending higher bait application rates and longer periods of bait availability (Keitt, et al., 2015). The new guidelines meant it would be extremely challenging logistically to complete a ground-based operation for Vahanga and the use of a helicopter was recommended.

Funding for rat eradication on Vahanga was eventually obtained in 2014. However, due to the costs associated with transporting a helicopter to the south-east corner of French Polynesia and the relatively low cost of including additional sites and invasive species, a decision was made by project partners to target five additional, high conservation value islands in the area. This decision was facilitated by broadening the project partnership and securing additional funding. An operational plan was devised that prescribed the aerial application of rodent bait containing brodifacoum to target rats followed by trapping and hunting to target cats, rabbits and goats across the six project sites (Derand, et al., 2015).

The target bait application rate for rat eradication was derived using the methods described by Pott et al. (2015) to interpret bait availability data collected by Griffiths et al. (2011). The proposed application rate, coupled with reported island sizes and areas derived from available satellite imagery and a 15% contingency for lost or damaged bait, were then used to estimate the total amount of bait required. Immediately before the project's implementation, higher resolution satellite imagery acquired from the Millennium Coral Reef Mapping Project (Andréfouët, et al., 2005)

Table 1 Characteristics of the six sites targeted for invasive vertebrate removal.

Island	Area (ha)	Elevation (m)	Location	Native threatened species expected to benefit	Targeted invasives
Vahanga	380	5	Acteon	Pacific ground-dove, Tuamotu sandpiper, atoll fruit-dove (*Ptilinopus coralensis*), Murphy's petrel (*Pterodroma ultima*), bristle-thighed curlew (*Numenius tahitiensis*), green turtle (*Chelonia mydas*)	*Rattus exulans*
Tenania	425	5	Acteon	Pacific Ground-dove, Tuamotu sandpiper, bristle-thighed curlew, green turtle	*R. exulans, R. rattus, Felis catus*
Kamaka	58	166	Gambier	Polynesian storm petrel, Murphy's petrel	*R. exulans*
Makaroa	22	136	Gambier	Polynesian storm petrel	*R. exulans, Capra hircus*
Manui	8	54	Gambier	Polynesian storm petrel, Murphy's petrel	*Oryctolagus cuniculus*
Temoe	431	5	Gambier	Murphy's petrel	*R. exulans*

and EVS-Islands digital earth imagery showed that initial estimates of island areas had been overestimated, in some cases by as much as 22%. Consequently, this made more bait available for distribution at each site than had been planned.

Project implementation

Staging

Ninety-two tonnes of rodent bait, 30,000 l of Jet A1 helicopter fuel, three bait-spreading buckets, and equipment and supplies necessary for the project were shipped from the port of Papeete to the project sites by the coastal freighter 'Nuku Hau'. Rodent bait was transported in 22.7 kg paper-walled sacks stacked inside Ox Boxes (waxed cardboard pods) (hereafter referred to as pods) and the fuel in 200 l drums. A single-engine Squirrel AS350 B2 supplied by Tahiti Helicopters was flown from Papeete by 'island hopping' between four intermediate islands (a total distance of 1500 km) before converging with the Nuku Hau at Vahanga to commence the offloading process.

Immediately prior to unloading, the island's coastal boundaries were flown to confirm the size of the area to be treated and revalidate the amount of bait and fuel to be unloaded at each site. All equipment and supplies were first offloaded from the Nuku Hau to a small barge which was then unloaded by helicopter in separate sling loads. Bait and fuel, sufficient for each atoll, were staged on Vahanga, Tenania, and Temoe with unloading taking between 4–6 hours for each atoll. Supplies for the three closely grouped Gambier Islets were staged in less than four hours on Kamaka. To minimise flying between island groups, two bait spreading buckets were offloaded in the Acteon Group, one for use on Vahanga and the other for Tenania. One bucket and a range of spare parts were stationed in the Gambier Islands for use on both Temoe and the Gambier Islets.

Project team members, 24 in total, were also deployed at this time. Team members were stationed on Vahanga (6), Tenania (5), Temoe (3) and Kamaka (1). The project manager, GIS analyst, baiting team (3), pilots (2) and helicopter mechanic travelled by helicopter between the islands to complete bait applications and one person, stationed in Mangareva, provided logistical support. In between bait applications, the project manager, GIS analyst and members of the baiting team deployed to different islands to provide support for monitoring, trapping and hunting.

On Tenania, large piles of broken coconut husks containing coconut flesh were found across the atoll. These byproducts of the recent copra harvest represented a signficant alternative food source for rats and a risk to the project's likelihood of success. To reduce risk, members of the project team systematically burned piles of coconuts. This laborious activity greatly reduced the amount of coconut available to rats but did not eliminate it.

Bait application

After staging was complete, bait application took place sequentially beginning on Temoe followed by the Gambier Islets, Vahanga and finally Tenania. Each of the three atolls took more than one day to complete due to the amount of bait applied and the requirement to break the circular atolls into multiple blocks. Dividing the operational area into blocks maximised the length of flight lines that could be flown thereby simplifying the operation for the pilot. Adjacent baiting swaths were overlapped by 50% to reduce the risk of gaps in coverage (e.g. Fig. 2). In addition to parallel flight lines across each island's interior, a swath with a deflector bucket (which spreads bait in one direction only) was completed along the edge of both coastal and lagoon vegetation. Additional bait was applied by helicopter over areas considered to be higher risk, such as areas of human habitation or sites known to support the highest crab densities. At the same time as bait was applied by helicopter, rodent bait was placed in small dishes within all buildings still in use and scattered by hand underneath buildings and inside all derelict or abandoned structures.

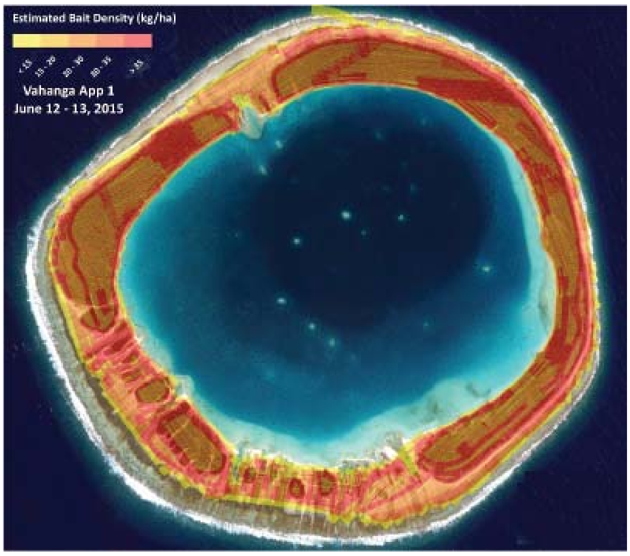

Fig. 2 First bait application completed on Vahanga.

Following an 18 day interval, a second application of bait was completed at the project sites in the same sequence. The length of the interval was dictated by a desire to ensure that all individuals (including juveniles) within the targeted rat populations were exposed to bait, as discussed in Keitt, et al. (2015), and also by the resource limitations of the project partnership. Operational specifications for the second bait application were the same, except for the exclusion from bait application of barren storm-washed coral habitat across all three atolls. Bait availability monitoring and anecdotal observations suggested negligible disappearance of bait from these atolls and thus no advantage in re-treating these areas. This action was also seen as a means of reducing risk to non-target species such as Tuamotu sandpiper (*Prosobonia cancellata*). Operational areas treated were thus smaller in the second application for Vahanga, Tenania and Temoe (Table 2). Dates of bait application and the application rates achieved are provided in Table 2. No significant delays because of sustained rainfall or excessive winds were encountered.

Loading of bait spreading buckets was undertaken from platforms constructed from an 18 mm thick plywood sheet set atop two pods. A second plywood sheet was placed on the ground in front of these pods to ensure a level footing for the spreader bucket. The helicopter was fitted with a VHF radio for ground to air communications with the bait loading team.

The pod and pallet containment system withstood crushing (some pods were stacked up to seven high in the hold of the Nuku Hau), being dipped in saltwater (as they were airlifted onto the islands), tropical temperatures and periods of heavy rain. Water was found inside the internal plastic bag (used to protect sacks of bait) in just four pods and of these pods only the bags at the bottom of the pod were affected. Only four of the 4,065 bags (<0.1%) of bait shipped were considered unfit for application. Water ingress into pods was primarily a result of damage incurred to the cardboard during shipping and unloading, coupled

Table 2 Bait application summary.

Date	Island	Application	Bait used (kg)	Island area treated (ha)	Average bucket sow rate (kg/ha)*	Operational hours	Bait spread (T/hr)	Average ground application rate (kg/ha)+
June 8–9	Temoe	1st	15,111.9	429.1	24.4	11	1.37	35.2
June 26–27	Temoe	2nd	16,433.5	341.5	29.9	9	1.83	48.1
			31,545.4	**429.1**	**26.8**	**20**	**1.57**	**73.5**
June 10	Gambier	1st	3,797.8	88.2	32.3	3	1.27	43.1
June 28	Gambier	2nd	2,986.9	86.6	21.1	2.75	1.09	34.5
			6,784.7	**88.2**	**26.8**	**5.75**	**1.18**	**76.9**
June 12–13	Vahanga	1st	11,715.5	382.6	21.8	7.5	1.56	30.6
July 3	Vahanga	2nd	14,272.8	333.3	27.7	6.5	2.20	42.8
			25,988.3	**382.6**	**24.5**	**14**	**1.86**	**67.9**
June 14	Tenania	1st	13,479.3	419.6	24.3	9	1.50	32.1
July 4–5	Tenania	2nd	14,315.3	394	23.5	8	1.79	36.3
			27,794.6	**419.6**	**23.9**	**17**	**1.64**	**66.2**

*Average rate at which bait was spread from the bucket (bait used/TracMap recorded area). +Average rate at which bait was available on the ground (bait used/island area treated).

with water pooling on the lid of the pod. Intact pods showed no sign of water ingress despite water pooling.

Rabbits

Based on the results of other projects (e.g. Griffiths, et al., 2014), most rabbits were expected to consume rodent bait and succumb to poisoning on Manui. This proved to be the case, with just four survivors found and one of these appeared to be close to death at the time it was shot. Two staff began follow-up work targeting surviving rabbits, nine days after the first application of bait, to eliminate survivors before the team departed French Polynesia.

All accessible areas of the island offering apparently suitable habitat were searched for sign and surviving rabbits during the day and at night, using powerful head lamps (see Fig. 3). Some inaccessible parts of the island such as cliff faces were searched with spotlights at night, but comprehensive searching of these areas was not possible. To manage search effort and spatial data, the island was divided into zones. Generally, the same zone was searched during the day and then again at night. Areas where fresh sign was found, or a live rabbit sighted were searched more intensively. Search effort was logged using handheld GPS and a map used to identify areas that had not yet been visited. Waypoints were recorded for any fresh sign found and live rabbits sighted.

Two trail cameras were established on the island from 26 April 2015 and seven added from 10 June. Three cameras were kept at the same locations throughout the operation while the remaining four were moved to locations where fresh sign was found or where rabbit presence was suspected. Old rabbit sign (faeces and chewed vegetation) was found in most parts of the island except within the molasses grass sward and the coastal littoral zone. Ten freshly dead rabbits (presumed poisoned) were found. Carcasses were found in three discrete locations and were generally associated with areas of concentrated old sign.

Fresh rabbit sign was found in three discrete locations during the period of follow up searching. In each case the discovery of fresh sign led to the location of freshly dead or surviving rabbits within the same area. One adult female was found during the day on 19 June and shot. This individual superficially appeared to be in good condition but was presumed to be in the last stages of anticoagulant

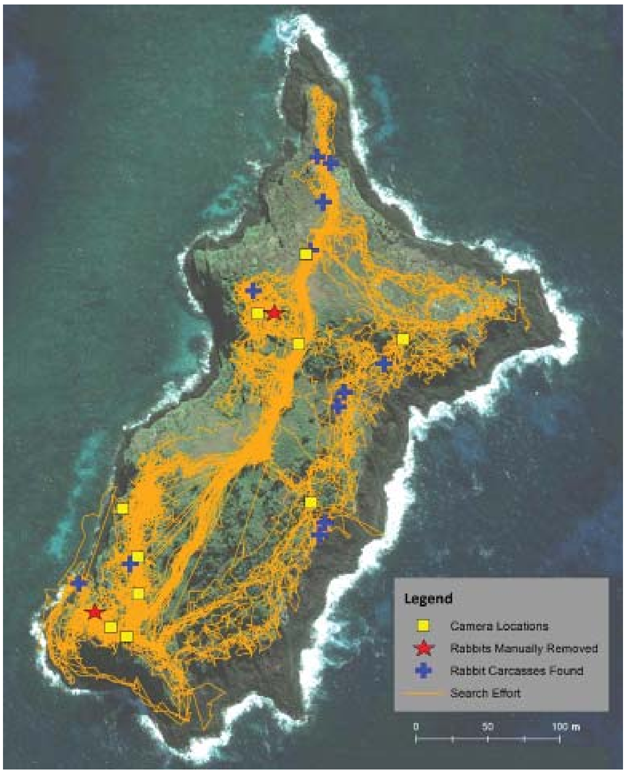

Fig. 3 Search effort and location of surviving rabbits on Manui 10 June to 3 July 2015.

poisoning as it did not move when approached and, although not evident in the gut, the lower intestine was full of blood.

Two young rabbits (a male and female) were found together while spotlighting and shot.. These individuals appeared healthy and a necropsy indicated no evidence of bait ingestion or anticoagulant poisoning. The last surviving rabbit was shot 100 m further north also after finding fresh sign. This individual, an adult female, was in excellent condition and showed no sign of bait ingestion or anticoagulant poisoning. Following removal of this individual, no further fresh sign or images on cameras were found during five more days of search effort.

Cats

Cat trapping on Tenania began on 6 June prior to the 1st bait application and continued until 4 July when the team departed. A total of 564 trap nights (sum of the number of active traps for each trap night) were achieved. Trapping was conducted primarily with leg hold traps, No. 2 Bridger padded, and No. 1.5 Oneida/Victor unpadded leg hold traps set in a combination of cubby, trail and bucket sets. Traps were baited with either canned or fresh fish, lured with a commercial lure or left un-baited in the case of some trail sets. Every 2–3 days, traps that had sprung were triggered and reset. Trap locations were changed as necessary when sign (tracks, scat, and visual observation) was encountered in the field and camera data collected.

Traps were raised (i.e. positioned on top of sand filled buckets) or left unlured to minimise crab interference. However, traps were often triggered by what remaining sign indicated was crabs, mostly *Coenobita perlatus*. Spotlight surveys were undertaken but only one cat detected using this method and this method was not pursued. A total of 10 remote trail cameras were installed on 5 June and data collected daily, in most cases, to inform trap placement. Monitoring with cameras continued until 4 July. Camera data were also used to verify the identity of captured cats. Ten distinct individual cats were detected with trail cameras and, of these, nine were caught: five female and four male. All were mature adults and one female was pregnant with three foetuses at the time of capture. Feral cat captures were made exclusively with leg-hold traps; three were caught in cubby sets, five in trail sets and one in a bucket set. The last cat captured displayed signs of internal haemorrhaging, likely due to secondary exposure to brodifacoum. The last feral cat detected by trail camera on 29 June was not captured, despite concentrated trapping in the vicinity of detection, and is assumed to have succumbed to secondary poisoning. An individual with distinct black and white fur patterns, seen during the first spotlight survey, was also never observed again despite follow-up surveys.

All cats captured appeared to be in excellent body condition. When the captured cats' stomach contents were examined, the only prey remains observed were rodents. Interestingly, the stomachs of two cats captured contained coconut flesh. Rat remains encapsulating the observed coconut within one individual, indicated rats to be the source; however, the other had its stomach completely full of coconut.

Goats

Despite local reports to the contrary, eight goats were still present on Makaroa in 2015 at the time of the project's implementation. One of these (a young female) was shot, but further hunting effort was abandoned due to insufficient capacity, a lack of suitable firearms and the remaining goats being extremely wary due to having been hunted recently. Two experienced hunters returned in 2017, each with a .308 calibre rifle and thermal imaging equipment. Eighteen goats were removed during the first four days and no more were seen in the subsequent six days of intensive search effort (Table 3). It is unknown whether goats ate rodent bait, but its application to remove rats had no apparent impact on the population, and the presence of goats did not impact the success of the rat eradication.

Non-target species mitigation

The proposed application of rodent bait posed a potential risk to non-target native species such as the Polynesian ground dove and the Tuamotu sandpiper. Tuamotu sandpiper were considered at high risk based on observations made on Tahanea Atoll during a rat eradication in 2011 (Pott, et al., 2014). Concerns were also held for Polynesian ground dove although other projects had targeted rats in the presence of conspecifics without apparent losses (Griffiths, 2014). Both species were recorded in very low numbers at just one of the project sites (Vahanga) and, because of their conservation status, mitigation was undertaken.

Prior to bait application, efforts were made to catch all Polynesian ground dove on Vahanga and translocate them to Tenararo. Of the five to six birds observed, two were captured and transferred. The others evaded capture and were monitored over the course of the project's implementation, along with two individuals sighted on Tenania. Transferred birds had two of the outermost primaries of each wing removed to lessen the chances of them flying back to Vahanga.

Efforts were also made to capture and transfer Tuamotu sandpiper. More birds were found on Vahanga than had been anticipated and five of the six birds present were caught. One escaped, but four were translocated to Tenararo with outermost primaries plucked on both wings (1–3 per wing, depending on bird condition) to prevent their return to Vahanga.

Table 3 Monitoring completed to confirm eradication success at the six project sites.

Island	Invasive species	Monitoring effort				Outcome
		Corrected trap nights	Spotlighting (hrs)	Sign searches (hrs)	Trail cameras (hrs)	
Vahanga	Pacific rat	345	16	112	0	Successful
Tenania (Tenarunga)	Pacific rat, ship rat, cat	213	17	112	420	Successful
Temoe	Pacific rat	455	25	128	0	Successful
Kamaka	Pacific rat	612	-	-	0	Failed
Makaroa	Pacific rat, goat	210	8	188	440	Successful
Manui	Rabbit		20	232	1,230	Successful

These interventions were partially effective for both species with 90% of captured birds resighted in 2017 (R. Pierce pers. comm.). Ground dove that remained on Vahanga and Tenania were resighted throughout the period of implementation, suggesting any risks to this species were low. Sightings of an uncaptured Tuamotu sandpiper displaying symptoms of poisoning were made on Vahanga. This individual was not seen again highlighting the vulnerability of this species.

ERADICATION SUCCESS

Trapping, spotlight searches and searches for sign of invasive vertebrate presence, conducted in April and May 2017 nearly two years after the project was implemented, confirmed the project was successful at removing invasive vertebrates at five of the six sites. No rats were found on Vahanga, Tenania, Temoe or Makaroa. No cats were found on Tenania or rabbits on Manui and goats were finally removed from Makaroa. The monitoring effort expended for each site to confirm eradication success is provided in Table 3. Despite Kamaka being inhabited, rats were not detected until monitoring was instigated nearly 12 months after the project was implemented. Rats are now widespread on the island. Analysis of DNA confirmed that some rats survived the operation.

OUTCOMES

In removing invasive species from five islands, the project increased the total number of islands free of invasive vertebrates within French Polynesia from four to nine and created an additional 1,426 ha of secure habitat, effectively tripling the area available for Polynesian ground dove and Tuamotu sandpiper recovery. Early signs of recovery were observed in 2017 with more individuals of Polynesian ground dove seen on both Vahanga and Tenania in 2017 and Tuamotu sandpiper recorded on Tenania for the first time. Recovery of native vegetation was observed on both Manui and Makaroa. Longer term monitoring is required to confirm trends.

With the removal of rats, the risk of rodent-borne leptospirosis has been eliminated from Tenania and the quality and quantity of copra produced appears to have increased, although the increase in income generated for the local community has yet to be quantified. Local skills to undertake future eradication projects were developed and support from policy-makers, funders and the public for future rodent eradications on other atolls/islands generated.

PROJECT COST

The operational cost of the project was estimated based on expenditure records kept by project partners. The total cost of the project, from when concerted planning began in 2014 to completion of the operation in 2015, was €1.4M with the largest costs being the helicopter, shipping, rodent bait and personnel. The cost efficiency of the project gained by targeted all six islands was assessed by comparing the total cost of the project with estimates completed separately for eradicating invasive mammals independently at each site (Table 4). Costs such as helicopter, shipping and staff travel would all have added significantly to cost if each island had been completed as a standalone project. Postponement of goat eradication on Makaroa increased costs for this component of the project but only by a relatively small margin as the cost of goat eradication was small (<€20,000).

DISCUSSION

Implementing the project described in this paper was challenging due to the remote nature of the islands, the number of sites, the range of invasive species targeted, and the lack of infrastructure and resources available within French Polynesia. Overcoming these challenges required an extensive and thorough planning effort. An added benefit of the time taken for project planning was the clear identification of roles and responsibilities for each project partner. Each of the project partners provided capabilities that could not readily have been supplied by the other partners.

An operational strategy, informed by a contemporary review of rat eradications on tropical islands (Keitt, et al., 2015) contributed to project success although, as noted, rats survived on Kamaka despite the application of best practice guidelines. Reasons why rats survived on Kamaka are unknown but an investigation to determine causal factors is currently underway. The project also benefited from generally favourable weather through the implementation phase. In hindsight, sufficient time, effort and resources were put in place to ensure successful cat and rabbit removal from Tenania and Manui. However, more time spent on each of these islands would have increased the level of confidence held by departing teams that surviving individuals had been removed. Local reports that goats were no longer present on Makaroa proved incorrect and eradication of this species had to be postponed.

The cost efficiencies gained in this project through removing invasive species from multiple islands are evident. Completing each of the islands as a standalone project would have increased the total cost of removing invasive species from the six sites by a factor of three. Resources for conservation are scarce and similar approaches will need to be considered for many projects to make them economically viable. The proposed removal of rats and cats from five uninhabited islands in the Marquesas archipelago is one such example. The high costs of shipping and helicopters would rule out doing any one of the islands as a standalone project.

Interventions to mitigate the impacts of the operation to non-target species were largely effective (Pierce, et al., 2015) and the level of mortality sustained will be outweighed by the anticipated benefits to populations following the removal of rats from Vahanga. Although it is too early to measure the full impact of this conservation intervention, Polynesian ground dove and Tuamotu sandpiper should increase in abundance on both Vahanga and Tenania, eventually forming self-sustaining populations. The number of populations of Polynesian ground dove will increase from three to five and for

Table 4 Projected standalone project costs and the actual costs incurred for removing invasive vertebrates from the six project sites.

Island	Projected cost	Actual cost[1]
Vahanga	€1.1M	€0.3M
Tenania	€1.1M	€0.3M
Temoe	€1.1M	€0.3M
Kamaka	€0.4M	€0.15M
Makaroa	€0.5M	€0.2M
Manui	€0.4M	€0.15M
Total	€4.6M	€1.4M

[1] Costs such as flying the helicopter from Tahiti were divided equally between project sites.

Tuamotu sandpiper from six to eight. Polynesian storm petrel (*Nesofregetta fuliginosa*) along with other sea birds are expected to recolonise Makaroa increasing the number of breeding sites for this species from at least six to seven. Translocations of these species and others are also now possible.

Completion of the project provided greater security from extinction for a number of plant and animal species but most importantly for bird species listed as critically endangered or endangered by the IUCN (IUCN, 2010), Polynesian ground dove, white-throated storm petrel and Tuamotu sandpiper. The project also delivered socio-economic benefits to local communities through increased production from a coconut plantation on Tenania and greater resilience for harvested seabird populations on Temoe. In doing so, the project provides a precedent for further action within French Polynesia to protect endemic biodiversity and livelihoods.

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REFERENCES

Andréfouët, S., Chauvin, C., Spraggins, S., Torrespuzilla, D. and Kranenbourg, C. (2005). *Atlas des Recifs Coralliens de Polynesie Francaise*. Nouvelle-Caledonie: Centre IRD de Noumea.

Atherton, J. (2007). *Polynesia-Micronesia Biodiversity Hotspot. Critical Ecosystem Profile*. Arlington, Virginia, USA: Ecosystem Partnership Fund, Conservation International–Melanesia Center for Biodiversity Conservation.

Blanvillain, C., Florent, C. and Thenot, V. (2002). 'Land birds of Tuamotu Archipelago, Polynesia: Relative abundance and changes during the 20th century with particular reference to the critically endangered Polynesian ground-dove (*Gallicolumba erythroptera*)'. *Biological Conservation* 103(2): 139–149.

Broome, K., Boudjelas, S. and Gouni, A. (2011). *Feasibility Study for the Eradication of Pacific Rats from Vahanga Atoll, French Polynesi*a. Unpublished Report, Tahiti, French Polynesia: Societe d'Ornithologie de Polynesie, Manu.

Burt, A.J., Gane, J., Olivier, I., Calabrese, L., De Groene, A., Liebrick, T., Marx, D. and Shah, N. (2016). 'The history, status and trends of the endangered Seychelles magpie-robin *Copsychus sechellarum'*. *Bird Conservation International* 26(4): 505–523.

Cromarty, P.L., Broome, K.G., Cox, A., Empson, R.A., Hutchinson, W.M. and McFadden, I. (2002). 'Eradication planning for invasive alien animal species on islands – the approach developed by the New Zealand Department of Conservation'. In*:* C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species.* pp. 85–91. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Derand, G.-D., Griffiths, R., Ghestemme, T. and Cranwell, S. (2015). *Operational Plan to Eradicate Rats, Cats, Rabbits, Goats, and Control Lantana on Three Islands and Three Atolls In French Polynesia*. Unpublished Report. Suva, Fiji: BirdLife International, Pacific Secretariat.

Donlan, C.J. and Wilcox, C. (2008). 'Integrating invasive mammal eradications and biodiversity offsets for fisheries bycatch: Conservation opportunities and challenges for seabirds and sea turtles'. *Biological Invasions* 10(7): 1053–1060.

Faulquier, L. (2008). *Feasibility Study for the Eradication of Rats from Islands of the Gambier Archipelago (French Polynesia*). Suva, Fiji: BirdLife International, Pacific Secretariat.

Genovesi, P. (2011). 'Are we turning the tide? Eradications in times of crisis: how the global community is responding to biological invasions'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 5–8. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Gillespie, R.G., Claridge, E.M. and Goodacre, S.L. (2008). 'Biogeography of the fauna of French Polynesia: Diversification within and between a series of hot spot archipelagos'. *Philosophical Transactions of the Royal Society B: Biological Sciences* 363(1508): 3335–3346.

Griffiths, R. (2014). *Feasibility and Risk Assessment for the Removal of Invasive Species from Vahanga, Tenarunga, Temoe, Kamaka, Manui and Makaroa*. Unpublished Report. Santa Cruz, CA, USA: Island Conservation.

Griffiths, R., Miller, A. and Climo, G. (2011). 'Addressing the impact of land crabs on rodent eradications on islands'. *Pacific Conservation Biology* 17(4): 347–353.

Griffiths, R., Buchanan, F., Broome, K., Neilsen, J., Brown, D. and Weakley, M. (2014). 'Successful eradication of invasive vertebrates on Rangitoto and Motutapu Islands, New Zealand'. *Biological Invasions* 16(11): 1–15.

IUCN. (2010). 'IUCN Red List of Threatened Species. Version 2010.1' <www.iucnredlist.org>. Accessed 19 May 2010.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

Meyer, J.-Y. and Butaud, J.-F. (2009). 'The impacts of rats on the endangered native flora of French Polynesia (Pacific Islands): Drivers of plant extinction or coup de grâce species?'. *Biological Invasions* 11(7): 1569–1585.

Pierce, R.J., Boudjelas, S., Broome, K., Cox, A., Denny, C., Gouni, A. and Raust, P. (2006). *Ecological Restoration of Vahanga Atoll, Acteon Group, Tuamotu Archipelago – Draft Operational Plan.* Tahiti, French Polynesia: Société d'Ornithologie de Polynesie.

Pierce, R., Blainvillain, C. and Burle, M.-H. (2015). *Bird Research and Monitoring Associated with SOP Manu Pest Eradications in Acteon–Gambier Archipelagos, June 2015.* Unpublished Report. Suva, Fiji: BirdLife International, Pacific Secretariat.

Pott, M., Griffiths, R., Burle, M.-H. and Wegmann, A. (2014). 'Protecting the Tuamotu Sandpiper (*Prosobonia cancellata*) One Island at a Time'. In*:* R.M. Timm and J.M. O'Brien (eds.) *Proceedings of the 26th Vertebrate Pest Conference*, pp. 166–171. Davis, CA: University of California.

Pott, M., Wegmann, A., Griffiths, R., Samaniego-Herrera, A., Cuthbert, R., Brooke, M., Pitt, W., Berentsen, A., Holmes, N., Howald, G. and Russell, J.C. (2015). 'Stacking the deck: Assessing bait availability before rodent eradications to inform bait application rates'. *Biological Conservation* 185: 27–35.

Raust, P. and Sanford, G. (2007). *Les Zones Importantes pour la Conservation des Oiseaux en Polynésie Française*. Papeete, Tahiti, French Polynesia: Société d'Ornithologie de Polynésie.

Rice, S.E. and Clark, R.W. (2016). 'Factors affecting intra-island genetic connectivity and diversity of an abundant and widespread island endemic, the San Clemente Island night lizard'. *bioRχiv*: 060038.

Russell, J.C., Cole, N.C., Zuël, N. and Rocamora, G. (2016). 'Introduced mammals on western Indian Ocean islands'. *Global Ecology and Conservation* 6: 132–144.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Simberloff, D. (2002). 'Today Tiritiri Matangi, tomorrow the world. are we aiming too low in invasives control?'. In*:* C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 4–12. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Towns, D., Miller, K., Nelson, N. and Chapple, D. (2016). 'Can translocations to islands reduce extinction risk for reptiles? Case studies from New Zealand'. *Biological Conservation* 204: 120–127.

Zarzoso-Lacoste, D. (2013). 'Vers une Meilleure Compréhension des Interactions Trophiques Directes et Indirectes entre Prédateurs Invasifs et Espèces Natives au Sein des Écosystèmes Insulaires'. PhD thesis. Marseille, France: Aix-Marseille University.

Trialling gene drives to control invasive species: what, where and how?

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Abstract The control of invasive species would be enhanced through the addition of novel, more effective and sustainable pest management methods. One control option yet to be trialled in the field is to deploy transgene-based 'Gene Drives': technologies which force the inheritance of a genetic construct through the gene pool of a wild population, suppressing it or replacing it with a less harmful form. There is considerable interest in applying gene drives to currently intractable invasives across a broad taxonomic range. However, not all species will make efficient or safe targets for these technologies. Additionally, the safety and efficacy of these systems will vary according to where they are deployed, the specific molecular design chosen, and how these factors interact with the ecology of the target pest. Given the transformative but also controversial nature of gene drives, it is imperative that their first field trials are able to successfully demonstrate that they can be used safely and efficiently. Here, we discuss how to maximise the probability of this outcome through considering three important questions: *What* types of invasive species should we use to trial gene drives? *Where* should we be trialling them? and *How* should these trials be conducted? In particular, we focus on the ecological, genetic and geographic features of small, isolated islands which make them ideal locations for these initial trials. A case study of an island invasive that is deemed highly appropriate for gene drive intervention, and for which gene drive development is currently underway (*Mus musculus*), is used to further explore these concepts.

Keywords: biodiversity conservation, CRISPR, *Culex quinquefasciatus*, gene drive, island invasive, *Mus musculus*, population eradication, restoration

INTRODUCTION

Molecular advancements have made feasible a new range of Genetic Pest Management (GPM) strategies – the transgene-based gene drives (Sinkins & Gould, 2006). These technologies aim to introduce DNA sequences (the gene drive transgene) into the genome of a wild pest population through the release of genetically engineered individuals which go on to mate with conspecifics in the field. Once introduced, the inheritance of the gene drive is forced – driven – through the target population gene pool along with its control phenotype. This driving effect can be achieved, for example, by biasing inheritance of the transgene above normal mendelian levels, or through placing an evolutionary advantage on inheritance of the transgene at the population level. Proposed control phenotypes aim either to reduce/eradicate a pest population – "population suppression" strategies – or to leave a population intact but modify it so that it is less harmful (e.g. by spreading a transgene which makes a mosquito population less able to transmit a particular disease) – "population replacement" strategies (Alphey, 2014). Within population suppression, current proposals aim to spread either a sex ratio bias (usually in favour of males) or a genetic load, e.g. female sterility (Deredec, et al., 2008).

Theoretically, gene drives could be engineered that are capable of spreading to every member of an interbreeding population from one or several relatively small initial releases (Deredec, et al., 2008). This autonomous nature is appealing for invasive species control, where programmes often extend into remote/inaccessible areas and less than total eradication may be viewed as failure. Indeed, there is increasing interest in applying gene drives to currently intractable invasive species that threaten biodiversity (Alphey, 2002; Gould, 2008; Esvelt, et al., 2014; Simberloff, 2014; Thresher, et al., 2014; Campbell, et al., 2015; NASEM, 2016; Harvey-Samuel, et al., 2017; Piaggio, et al., 2017). However, two primary concerns arise from their proposed use. Firstly, that a gene drive transgene could unintentionally spread beyond a target geographic area (e.g. from an invasive population into the native range of the invader) or into a non-target species through hybridisation/horizontal-gene transfer – here collectively termed 'transgene escape'. Secondly, that their persistence, once released, could cause unintended ecological effects that are difficult to reverse (Sutherland, et al., 2014; Webber, et al., 2015; NASEM, 2016).

Previous field testing of gene drives is limited to non-transgenic population replacement utilising artificial infections of *Aedes aegypti* mosquitoes with the intracellular bacterium *Wolbachia* (Hoffmann, et al., 2011; Schmidt, et al., 2017). *Wolbachia* technologies are considered non-transgenic as they do not, deliberately, involve the introduction of DNA sequences into the target pest genome. Proposed application of transgene-based gene drives to invasive species differs from *Wolbachia* in that the systems available are, potentially, significantly more powerful and flexible and their taxonomic scope is broader, encompassing groups as divergent as plants, mammals, fish and molluscs, in addition to insects (Gould, 2008; Hodgins, et al., 2009; Thresher, et al., 2014; Campbell, et al., 2015; Sytsma, et al., 2015; Webber, et al., 2015). The first open-field trials of transgene-based gene drive technologies will thus represent a precedent-setting milestone. As recommended by the USA National Academy of Sciences (NASEM, 2016) , these trials will seek to examine whether the efficacy (e.g. its ability to invade a target population and induce a desired control phenotype therein) and safety (e.g. our ability to constrain its spread to the target population using molecular or experimental designs) of a gene drive system conform with theoretical expectations, themselves informed by preliminary laboratory experiments and mathematical modelling (Benedict, et al., 2008; Brown, et al., 2014). As such, open-field trials can be considered extensions of initial highly biocontained laboratory experiments where artificial biocontainment (Akbari, et al., 2015) is 'relaxed' because aspects of efficacy and safety have previously been demonstrated. Both these aspects – efficacy and safety – are important in order to convince a potentially sceptical public that they may have confidence in the wider use of these technologies.

Here we summarise the primary considerations involved in conducting the precedent-setting open-field trials of transgene-based gene drives (henceforth 'gene

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drives') in invasive species through posing three questions (1) What types of invasives are appropriate targets for these trials? (2) Where should the trials of these systems be located? (3) How should these trials be conducted? These questions are considered with the aim of exploring how these technologies could be trialled against invasive species as efficaciously as possible, whilst minimising the risk of transgene escape. In order to increase the value of this discussion these points are addressed in a general, rather than taxon-specific manner. Additionally, we explore their implications for a specific invader currently being targeted for control using gene drives – the house mouse, *Mus musculus* (See case study: GBIRd and Table 1). We bring this forward with the purpose of encouraging dialogue and improving criteria for such trials.

WHAT CHARACTERISTICS ARE IMPORTANT WHEN CHOOSING A TARGET ORGANISM?

General characteristics of a gene drive target

Minimum *requirements* for gene drive development are that the target pest is sexually reproductive, is amenable to laboratory rearing/germ-line transgenesis and is genetically well characterised.

As barriers to gene-flow within a population will decrease the efficiency of a gene drive's spread (see: The importance of dispersal), target species should preferably be obligately sexually reproductive (Alphey, et al., 2010) and incapable of self-fertilisation, which may simultaneously reduce the potential for gene drive resistance evolution (Bull, 2016). As such, it is unlikely these systems will be broadly applicable to invasive plants, of which many propagate vegetatively or through self-fertilisation (Kolar & Lodge, 2001; Rambuda & Johnson, 2004). Regarding transgenesis, the ease with which the germ-line cells can be manipulated will influence the speed that new transgene designs can be tested. Insect transgenesis has predominantly been through microinjection of pre-blastoderm embryos which requires that the fertilised egg is accessible. Transformation of species which are viviparous (e.g. the tsetse fly) or whose embryos are laid in inaccessible protective structures (e.g. pods or cases) may prove more challenging (Bourtzis, et al., 2016). Finally, as gene drives require the expression of various genetic components in highly temporal or spatially explicit patterns, often to target precise genomic loci, a good knowledge of the genetics of a target, e.g. a high-quality genome/transcriptome sequence and an understanding of the molecular-genetic basis of sex determination, is imperative.

Desirable characteristics are not absolutely necessary for gene drive development but, in practice, species whose biology diverged significantly from these characteristics would be deemed as inappropriate targets for these technologies.

Chief amongst desirable characteristics is a short generation time. This will minimise the time taken for strain development, and for these vertically transmitted systems to spread through and control a target population. Similarly, species with complex mating systems (e.g. the synchronised and ephemeral mating events of termites or ants) or where subsets of the population can remain dormant and inaccessible (e.g. long-term seed banks) effectively

Table 1 Idealised ecological selection criteria proposed as an initial filter for potential trial islands for potential gene drive constructed mice trials within Australia, New Zealand, USA. Additional steps will be required prior to any potential field trial, including engagement with stakeholders (e.g. land managers, local communities) and regulators to determine final approval.

Criteria	Rationale
1 Island is biosecure Desktop assessment indicates: a. Closed to public or infrequent/controlled visitation b. Remote enough (>1 km from other land masses) to avoid unassisted immigration or emigration	● Mice typically invade remote islands through human mediated transport, not through swimming (Russell & Clout, 2005). ● *M. musculus* are known to have swum up to 500 m between land masses (Harris, et al., 2012). ● Closed population required for proof-of-concept ● After desktop assessment. If the island passes other filters and is tentatively selected, conduct a biosecurity risk assessment. Island biosecurity plans for individual islands or island groups should be developed and implemented if island is selected (Fritts, 2007; Russell, et al., 2008; AAS, 2017)
2. No significant challenges exist to treatment using traditional methods to eradicate mice, e.g.: a. Uninhabited (besides research station or similar) b. No livestock c. No native rodents d. No non-target species of concern e. Regulatory environment allows the use of brodifacoum bait products and no rodenticide resistance alleles present f. Island size <300 ha g. Single land manager	● Provides a means to terminate experiments (i.e. exit strategy) using traditional methods without known complicating factors.
3. *M. musculus* are the only rodent present *or* could be introduced.	● Mouse behaviour is known to change significantly in the presence of rats (Harper & Cabrera, 2010). ● There may be man-made or other islands that are suitable that don't currently have *M. musculus* present.
4. Reasonably economical and feasible to visit the island year-round.	● Some islands are cost prohibitive to visit. ● Seasonal conditions may impact safe access to the island.

extend the generation time and may limit transgene introgression into or through a wild population (Alphey, et al., 2010). Furthermore, it is critical that there is a good knowledge of the ecology (e.g. mating systems, population dynamics and community interactions) of the target and in the case of vectors, the ecology and epidemiology of the pathogen and disease. The importance of this knowledge when developing a GPM strategy – from choosing the most appropriate/effective system, to predicting the impact of a strategy on a target population and community – cannot be overstated (Yakob, et al., 2008; Bax & Thresher, 2009; Yakob & Bonsall, 2009; Bonsall, et al., 2010; Thresher, et al., 2013; Piaggio, et al., 2017). Finally, it is desirable that the target is the dominant and ideally, sole, cause of an impact. As these strategies are vertically transmitted, they are extremely species-specific, making scenarios where there are multiple contributors to an impact (e.g. the spread of avian pox in Hawaii, where there are both mechanical and vector-based disease transmission routes) less appropriate.

The importance of dispersal

Gene-flow between populations

In limiting transgene escape into non-target areas, two important and interacting considerations are the level of gene-flow between a target and non-target population and the invasion threshold of the gene drive deployed (Figs 1 and 2) (Marshall & Hay, 2012). The invasion threshold is the theoretical frequency a gene drive transgene must be present at in a population before it will begin to spread. Highly invasive gene drives spread from very low invasion thresholds (e.g. the introduction of a few individuals into a target population) while less invasive systems may require significant levels of introduction before they begin to spread (high invasion threshold). Transgene escape may be considered an issue if gene-flow occurs at a frequency which makes it probable that a gene drive will exceed its invasion threshold in a non-target population

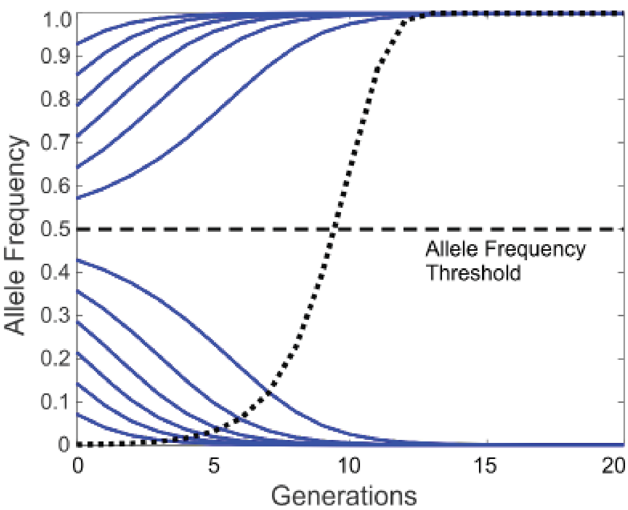

Fig. 1 Gene drives may be classified by their level of invasiveness, which is defined as the frequency they must reach in a target population before they begin to spread (the invasion threshold).Relatively non-invasive gene drives such as underdominance-based systems (Reeves, et al., 2014) (solid lines) require a high minimum allele frequency (dashed line) to be exceeded before they will begin to spread (50% of the population in this simulation). This differs from highly invasive (also known as "global") gene drives such as homing-based systems (Deredec, et al., 2008; Unckless, et al., 2015) (dotted) that will theoretically spread throughout a population even from a very low initial allele frequency, at least in the absence of resistant alleles.

within the time-frame of a trial (Akbari, et al., 2013). An 'acceptable' level of gene-flow between target and non-target populations will therefore be significantly higher for less invasive gene drives (Fig. 2). As the choice of gene drive may be constrained by the desired outcome (less invasive systems are generally more suited to replacement rather than suppression), it may not always be possible to choose less invasive designs to prevent transgene escape in species which are capable of long-distance gene-flow. A more flexible option is to trial gene drives in species which show limited ability to disperse and where human-mediated dispersal pathways can be managed. As previously noted (NASEM, 2016), important considerations here are the distance, frequency and life-stage of dispersal. Generally, species which disperse as juveniles/adults will show lower rates of gene-flow between populations than those which disperse as fertilised embryos (seeds or spores) or gametes (e.g. wind-borne pollen) (NASEM, 2016). Furthermore, dispersal via gametes may be more likely to result in interspecific hybrids, potentially increasing the risk of transgene escape into non-target species (NASEM, 2016). Consideration of these dispersal issues may make terrestrial animals more attractive targets than plants or marine species. As social interactions can strongly influence adult/juvenile dispersal events, it is important to consider how the predicted outcome of a particular gene drive may interact with these species-specific behavioural cues. For example, mate-limitation or increased inbreeding at low population densities or highly skewed sex ratios (both expected outcomes of proposed suppression gene drive designs) could in some species/scenarios result in increased levels of dispersal (Clobert, et al., 2012; Matthysen, 2012) and potentially also transgene escape.

Prior knowledge of the dispersal behaviour of an invasive population is therefore a prerequisite to safely deploying a gene drive. Fortunately, for many important invaders details of their dispersal mechanisms, invasion rates and levels of gene-flow within their invaded range already exist – in addition to other useful details such as the observed variance in their population size. Potential target species and populations could be short-listed based on the existence of this historical information, which could then be used to inform models predicting the potential for transgene escape during the expected time-frame of a trial.

Gene-flow within a population

Reaction-diffusion models have shown that dispersal rates will affect the speed that a gene drive travels through a target population (Beaghton, et al., 2016). Under more realistic scenarios, barriers to gene-flow within a population may have a more qualitative effect on whether a gene drive will spread or persist (North, et al., 2013). This concern could be reduced by avoiding targets whose populations show strong local spatial structuring, e.g. those which engage in high levels of sib-sib mating (Hamilton, 1967). However, even less extreme levels of spatial structuring resulting from limited life-time dispersal can significantly affect the ability of a gene drive to spread through and collapse a target population (Huang, et al., 2011; Eckhoff, et al., 2017). In particular, species whose population dynamics are significantly affected by seasonality may provide more fragmented landscapes for a gene drive to attempt to traverse. Models comparing gene drive dynamics in spatially explicit and homogenous mosquito populations suggest that increased structuring of a target population decreases the parameter space under which the target population is successfully eradicated (Eckhoff, et al., 2017). In these models, sub-populations became explicit annually in response to lowered population densities during the dry season. If sub-populations became explicit prior to arrival of the spreading transgene, these areas could act as a source for wild-type reinvasion into areas where the drive had eradicated the pest the previous season. Although limited within-population dispersal can be overcome through increasing the 'patchiness' or number of transgenic releases (Huang, et al., 2011; Eckhoff, et al., 2017), this tactic partially negates the primary advantage of employing gene drives. In choosing a target it is thus critical to have evaluated whether, given their population spatial-structure and the gene drive chosen, the release effort required to efficiently eradicate or replace a population is low enough to justify intervention with this technology.

Relatedness to important pests

Development and trialling of gene drives against invasives will proceed most efficiently if target species impact multiple values (e.g. human or animal health, agriculture, conservation). If these 'dual-target' species can be identified then the financial burden of developing gene drive strategies could be shared amongst different funding agencies, efficient designs/components shared between different researchers and the benefits of, and motivation for gene drive deployment shared amongst varied stakeholders. If a target invasive did not impact multiple values, gene drive development would still benefit if they were closely related to species in which GPM technology had previously been investigated, due to the transferability of many underlying molecular designs and components (Harvey-Samuel, et al., 2017). Examples of dual-target species are the mosquito (*Culex quinquefasciatus*) – a vector for multiple human diseases (Eldridge, 2005) and an invasive vector of avian malaria in Hawaii (LaPointe, et al., 2012) – and rodents including the house mouse (*Mus musculus*) and rats (e.g. *Rattus exulans, R. norvegicus and R. rattus*) which collectively are serious economic pests of agriculture (Aplin, et al., 2003; Pimentel, et al., 2005), impact infrastructure, are hosts for human, domestic animal and wildlife disease (Banks & Hughes, 2012), and amongst the most damaging invasives of island ecosystems (Angel, et al., 2009; Harper & Bunbury, 2015). Encouragingly, germ-line transgenesis and genome sequences already exist for *C. quinquefasciatus* (Allen, et al., 2001; Arensburger, et al., 2010), *M. musculus* (Waterston, et al., 2002; Ivics, et al., 2014) and *R. norvegicus* (Gibbs, et al., 2004; Ivics, et al., 2014). Moreover, all these species are invasive in isolated, uninhabited areas where there are no closely related species: desirable characteristics for a gene drive trial location (see next section).

WHERE SHOULD TRIALS BE CONDUCTED?

In order to maximise containment and efficacy, small, isolated islands are ideal locations for the first trials of gene drives (WHO/TDR, 2014).

Advantages of island locations to trial safety

Limiting intraspecific transgene escape

Gene-flow from an invasive population to conspecifics in its native range will decrease with increasing inter-population distance, the ecological inhospitality of the intervening area and the size of the invasive 'source' population. Locating trials on small, isolated islands can therefore act as an ecological containment strategy (WHO/TDR, 2014; NASEM, 2016), reducing the risk of intraspecific transgene escape. The effectiveness of this containment will depend on the proximity of a trial island to the native range of an invader, its natural and human-

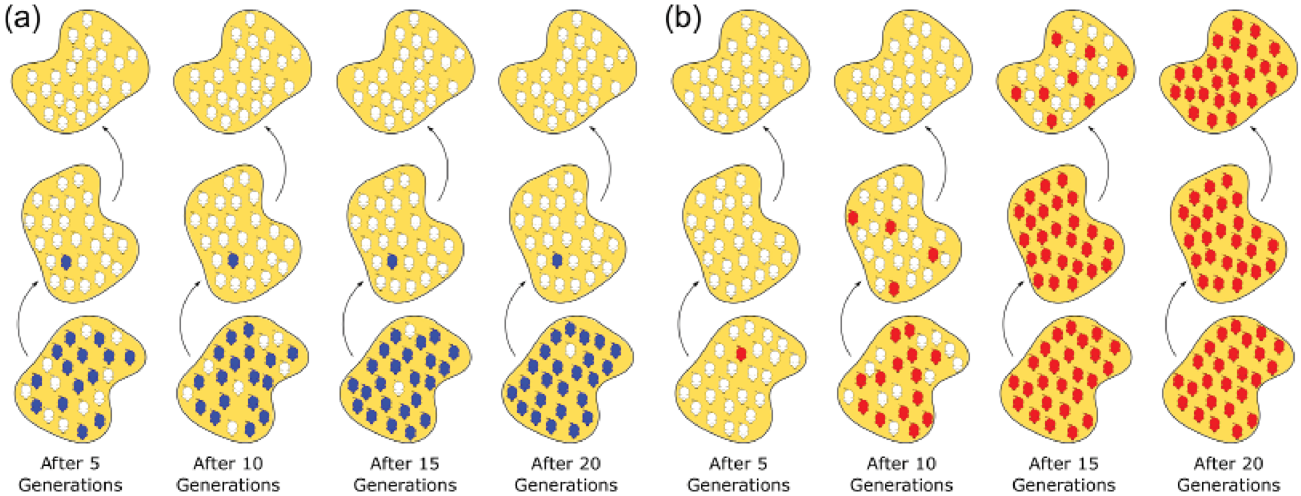

Fig. 2 The invasiveness of gene drive systems affects their containment ability once deployed in the field. Relatively non-invasive systems require large initial introductions before the gene drive will begin to spread and therefore migration alone is unlikely to exceed their invasion threshold. Highly invasive gene drives, on the other hand, can spread from only a few initial colonists and are predicted to spread through all linked populations. This is illustrated above using a three deme population genetics mathematical models. In each case we assume that a target (bottom) island and a nearby neighbour (middle) exchange 2% of their respective populations by migration in each generation while the nearby neighbour and a more remote (top) island exchange just 1%. It is assumed that no direct migration occurs between the target and remote islands due to the distance between them. Resulting transgene frequencies for each island at various times after the transgenic release are represented diagrammatically by a series of 25 mice (each representing a transgene frequency of 4%). White and shaded mice respectively represent wild-type and underdominance/homing drive transgenic allele frequencies, rounded to the nearest 4% (i.e. to the nearest whole mouse). Panel (a) shows results for a frequency dependent, single locus haploinsufficient underdominance-based system (Reeves, et al., 2014). This is a relatively non-invasive system with a high invasion threshold of 50% (See Fig. 1, solid lines). Here it is assumed that wild-type and transgene homozygotes suffer no fitness cost while 50% of heterozygous offspring are non-viable. For an initial transgene frequency of 55% it can be seen that the system spreads throughout the target population but does not reach significant levels in the neighbouring populations. Panel (b) shows results from a homing-based gene drive (Deredec, et al., 2008; Unckless, et al., 2015) which imparts no fitness cost on individuals and converts heterozygotes to homozygotes with 100% efficiency, introduced with an initial transgene frequency of 0.1%. The population genetics of this gene drive are shown in Fig. 1, (dotted line). Even this low initial frequency allows this highly invasive gene drive to spread throughout the target, and in time, the neighbouring populations also.

mediated dispersal ability and the invasiveness of the gene drive being trialled. A set of case-studies illustrating the interplay between these factors is the open-field releases of artificial *Wolbachia* infections aimed at local replacement of *A. aegypti* mosquito populations in Australia. After deliberate establishment in relatively isolated trial *A. aegypti* populations (Hoffmann, et al., 2011), it was found that long-distance dispersal was taking *Wolbachia* infected mosquitos into areas beyond the trial site (up to 1.86 km away) but that migration rates were insufficient over this distance to overcome the relatively high invasion threshold of the *Wolbachia* system (>30%) which remained largely contained to the trial site (Hoffmann, et al., 2014). Conversely, in subsequent releases where the trial site formed part of a larger, continuous *A. aegypti* population, *Wolbachia* was capable of spreading, albeit slowly, to high frequency beyond release sites and into the wild target population (Schmidt, et al., 2017). Gene drives with lower invasion thresholds than *Wolbachia* will require significantly greater isolation and/or molecular safeguard designs to limit transgenes to target populations/areas (discussed in the **How** section). This concept is illustrated for transgene-based gene drives in Figs 1 and 2.

In the context of island trial locations, the potential for a gene drive to cover large geographic distances, potentially back to mainland populations, through 'island-hopping' should not be overlooked (Bellemain & Ricklefs, 2008). For suppression drive designs, this island-hopping would require the existence of viable populations extending back to a native range and for the drive to escape each invaded 'stepping-stone' population before that population was itself eliminated by the drive. However, for replacement drives these aspects would not be a pre-requisite.

Limiting interspecific transgene escape

Transgene escape between species could take place either through horizontal gene transfer (HGT – acquisition of genetic material from an organism other than a direct ancestor) or introgression following hybridisation. Signals of HGT in metazoans can be seen by sequence comparisons between species (e.g. Crisp, et al., 2015). However, even the most frequent of these HGT events are rare, seen in nature on timescales of millions of years (e.g. Ortiz, et al., 2015). Therefore, as discussed generally for mosquitoes (Besansky, 2015) and specifically for homing-drives (Burt, 2003), HGT of a gene drive is held to be unlikely to occur at a frequency which will make it a realistic concern.

Regular gene-flow between native and invasive species through introgressive hybridisation, however, is well documented (Mooney & Cleland, 2001). Here, island locations provide both benefits and disadvantages in terms of limiting transgene escape. A benefit is that, given a frequency of fertile hybridisation events, stochastic elimination of an escaped transgene prior to its spread in a non-target species is more likely in small, island populations than at continental scales. However, hybridisation between closely related invasive and native species may be higher in insular compared to continental communities (Rhymer & Simberloff, 1996), potentially allowing transgenes to introgress into native populations at increased rates on islands. The potential genetic homogeneity of an island invasive population and simplicity of island communities (reducing the number of hybridising congeners) may prove advantageous in designing sequence-specific molecular safeguards to limit this risk.

Advantages of island locations to trial efficacy

Geographic isolation

Trials of gene drives will seek to achieve a series of pre-defined scientific endpoints (Brown, et al., 2014; NASEM, 2016). These will include evidence that the transgene is able to spread efficiently in the wild population, as well as endpoints specific to individual designs (e.g. reduced population density or reduced number of fully-competent vectors for suppression and replacement strategies, respectively). As immigration of wild-type individuals into a target population effectively dilutes the frequency of the transgene, unanticipated immigration will cause drive rates to be estimated inaccurately; this has been a frequently-observed problem in trials of sterile insects for population suppression (Klassen & Curtis, 2005) and is assumed to have prevented fixation of artificial *Wolbachia* infected mosquitoes in open-field trials (Hoffmann, et al., 2014). A sufficiently isolated island trial site will reduce this concern through minimising wild-type immigration. What constitutes 'sufficient' geographic isolation could be considered in conjunction with estimating outward gene-flow from a proposed trial island, acknowledging that migration rates between populations may not be symmetrical (Kawecki, 2004) and may only occur during infrequent events (e.g. El Niño, hurricanes).

Small population size

For equivalent release numbers/resources, introductions can be made at a higher population allele frequency on small islands than at larger, continental scales. This is primarily advantageous in testing gene drives with high invasion thresholds. However, even for more invasive systems, test releases would likely take place at frequencies well above the estimated minimum to protect against stochastic loss of the transgene in initial generations. Increased introduction rates will also allow the transgene to reach fixation (or a stable internal equilibrium) more rapidly (Deredec, et al., 2008). Moreover, for population-suppression strategies, smaller target populations may mean that density-dependent processes such as Allee effects (Tobin, et al., 2011) and environmental stochasticity (Eckhoff, et al., 2017) can be leveraged to more rapidly drive populations to extinction.

Genetically distinct

Small, insular populations arising from recent single invasion events are likely to be relatively genetically homogenous (Dlugosch & Parker, 2008). Assuming that heritable resistance to gene drives is possible (Bull, 2015), but founder individuals did not carry resistance alleles, this would provide target populations initially entirely susceptible to a released drive. Furthermore, given a constant mutation rate, such a gene drive resistance allele is less likely to arise in smaller, isolated populations within the time-frame of a trial. Conversely, however, if founder individuals did display pre-existing resistance it may occur at high frequencies. Target island populations should be screened prior to a trial for the presence of pre-existing resistance mutations; a relatively simple task for sequence-specific homing-drives, but potentially less straightforward for other technologies.

Which islands?

Islands that are small and sufficiently isolated to provide effective ecological containment could provide ideal locations for trialling gene drives. However, there are a number of biological, geographic and social criteria which will, in general, make a location more or less suitable for trialling GPM strategies (Benedict, et al., 2008; Lavery, et al., 2008; Brown, et al., 2014) and which can be extended to identify particularly promising examples within this group.

Biological criteria

If sufficiently isolated, invasive populations will be allopatric from conspecifics in their native range but sympatric with native congenic populations with which they might hybridise. If it occurs at an appreciable frequency, interspecific gene-flow may therefore be considered the more likely of the two risks when trialling gene drives in these locations. The most effective solution would be to avoid locations where there are closely related native species. For example, targeting invasive rodents on off-shore islands in New Zealand (which has no native terrestrial mammals) would pose low/no risk of transgene escape into native species, whereas deployment of the same technology in areas with diverse endemic rodent fauna such as south-east Asian archipelagos (Amori, et al., 2008) would likely require extensive pre-trial risk assessment. A further point to consider is that hybridisation events may be unidirectional with regards to sex (Rhymer & Simberloff, 1996). Molecular designs such as Y-drive which are transmitted exclusively through the paternal line would not be introgressed into a native population if hybrids formed via crosses between native males and invasive females.

If sufficient safety measures are taken, gene drives are expected to act in an extremely species-specific manner and are thus highly suitable for deployment in ecologically sensitive locations. However, a precautionary approach would suggest that precedent-setting trials be conducted in locations devoid of endangered/threatened flora or fauna (Brown, et al., 2014). This is particularly relevant if broader spectrum conventional control methods are used to terminate the trial at a pre-defined endpoint (see Table 1).

Geographic criteria

Barriers to gene-flow will decrease the efficiency of a released gene drive. Islands with relatively simple geographies and a resulting homogenous invasive population, for example low-lying oceanic islands, will therefore be most amenable to initial trials of these technologies. Where multiple islands occur in close proximity, these areas could be used to test assumptions on the spread of a drive technology within/between populations depending on the dispersal of the target (e.g. coral atolls/archipelagos for short/longer distance dispersal, respectively).

Social criteria

Challenges associated with invasive species control in inhabited areas are well-documented (Oppel, et al., 2011; Glen, et al., 2013). The novel and controversial nature of gene drives means that these challenges are likely to be exacerbated during their first trials. Levels of regulatory/engagement costs, risk assessment and societal objection are all likely to be more favourable if initial trials take place in uninhabited areas which are not of great cultural value. At least as importantly, restricting traffic off an island during a trial will substantially reduce the likelihood of transgene escape via intraspecific gene-flow. Employing modified biosecurity measures currently employed during conventional eradication efforts (Russell, et al., 2008), this would be far more feasible for uninhabited areas.

Previous experience in choosing sites for self-limiting GPM mosquito trials suggest that two social criteria critical for site identification are the existence of a credible regulatory structure and an enthusiastic local participant (e.g. academic researcher or wildlife management agency) with expertise regarding the invasive being targeted (Brown, et al., 2014). The regulatory framework in operation is relevant at multiple stages during planning and implementing a gene drive trial, from granting importation permits for gene drive organisms (Brown, et al., 2014) to determining appropriate risk assessment (NASEM, 2016) and public engagement (Lavery, et al., 2008) activities and experimental design/biosecurity during and after a trial (Benedict, et al., 2008). A robust and defensible regulatory framework allows public confidence in approved trials and reduces the likelihood of a trial being halted prematurely due to previously unvoiced concerns (Brown, et al., 2014). As regulation of gene drive trials is expected to take place on a case-by-case basis (Oye, et al., 2014) a local participant with knowledge of the regional ecological, social, economic, political and cultural context of deployment is invaluable. Additionally, due to the relative complexity and large scale (both temporal and spatial) expected of a gene drive trial, access to experienced research teams provided by a local collaborator would likely be necessary.

How should trials be conducted?

Practical guidance on how to conduct field-trials of self-limiting GPM mosquitoes (e.g. aspects of experimental design, safety and efficiency endpoints) is available (Benedict, et al., 2008; Brown, et al., 2014) and has been extended to the case of gene drives (WHO/TDR, 2014; NASEM, 2016). We will not replicate this discussion, but instead focus on how molecular designs can be utilised to increase the safety of a gene drive trial.

Proactive approaches

Proactive designs aim to limit the probability of transgene escape in the first instance. 'Precision' CRISPR-Cas9 gene drives (Esvelt, et al., 2014), which have been demonstrated in yeast (DiCarlo, et al., 2015) target the Cas9 endonuclease to cut a fixed DNA sequence in the genome unique to the specific target population, with the gene drive transgene then copied across into the cut site. The occurrence of such unique targeting sites is more probable in isolated populations derived recently from small numbers of initial founders and therefore may be particularly useful against island invasives. Alternatively, a 'daisy-chain' drive design could be employed (Noble, et al., 2016). Here a CRISPR-Cas9 gene drive is divided into a linear series of sub-components where each component will only drive in the presence of the component directly beneath it in the series. Critically, the basal component in a daisy-chain cannot drive and will be subject to loss over time through purifying selection. These components are then integrated at independent loci in a release strain meaning that the system is constrained spatially and taxonomically (multiple, sequential, components must escape an island population in the same individual or be combined again through interbreeding in order to continue driving) and temporally (selection will erode each basal component of the daisy-chain in turn until it is flushed from the population). Daisy-chain drives are currently being investigated for the island invasives *C. quinquefasciatus* and *M. musculus*, however analysis so far is theoretical, with – to our knowledge – no prototype strains reported in any metazoan.

An alternative proactive approach is to place inherent fitness costs on a gene drive such that it will persist for a time in a target population, potentially suppressing it, but not increase in frequency. Proposed examples include utilising a gene drive to spread a dominant female-lethal transgene, as proposed for mosquitoes (RIDL-with-drive) (Thomas, et al., 2000), and the endogenous *t*-haplotype meiotic-drive system to spread the male-determining *Sry* gene in mice. Although these systems utilise independent technologies and gene targets, their effects are the same: the transgene doubles in frequency each generation but half those individuals inheriting it (females) are non-viable. If transgenic individuals suffer from reduced fitness, or the drive is less than 100% efficient at biasing its inheritance – both of which are likely in the field – these systems will decrease in frequency over time once deployed (Backus & Gross, 2016), reducing the risk of transgene escape from a trial site but also their efficiency as suppression systems.

Responsive approaches

Responsive designs are complete or partial genetic systems, likely themselves gene drives, designed to be deployed in the event of an escaped drive in order to curtail its spread and potentially remove it or its phenotypic effects from a non-target population. These can include for example a 'reversal-drive' designed to target, spread into and disrupt the DNA sequence of an escaped drive, or the 'immunisation-drive' designed to spread into a non-target population and recode the wild-type target locus, making it unrecognisable to an escaped drive (Esvelt, et al., 2014). These designs can be combined into a single 'immunising-reversal' drive and be made less invasive through using the daisy-chain architecture. A more complex 'restoration-drive' design integrates a relatively non-invasive underdominance system (Figs 1 and 2) into this daisy-chain 'immunising-reversal' drive to theoretically allow the entire system to be flushed from the non-target population once the escaped drive has been halted (Min, et al., 2017).

Although reversal drives have been demonstrated in lab yeast colonies (DiCarlo, et al., 2015) and a non-driving equivalent in *Drosophila* (Wu, et al., 2016) it is unclear how effective these and other responsive approaches would be in the field. There is also concern that, in the event of an escaped gene drive, there may be considerable pressure against rectifying the situation through the release of another gene drive. A more realistic, but not mutually exclusive, approach would be to integrate a high level of conventional control methods at all potential transgene escape points (e.g. connected docking areas/airstrips) during and for a period after a trial. It is clear that responsive approaches should not be relied upon as critical containment methods during a gene drive trial.

Case study: Genetic Biocontrol of Invasive Rodents (GBIRd)

The Genetic Biocontrol of Invasive Rodents (GBIRd) programme aims to develop multiple gene drive systems in mice (*Mus musculus*) for simultaneous evaluation of their safety and efficacy using biosafety standards beyond those required by existing law, while carefully assessing the social, cultural and policy acceptability of such an approach (Campbell et al., 2019). The programme's first stage culminates in the potential submission of an application to a regulatory agency for release of gene drive constructed mice with a spatial control mechanism on a small, biosecure island to test eradication of the wild, invasive mouse population (Campbell et al., 2019). This step-wise approach follows recommendations from USA and Australian National Academies of Sciences (NASEM, 2016; AAS, 2017). Ecological criteria for selecting an appropriate trial island for this application have been proposed (Campbell et al., 2019; Table 1). However, these criteria are just an initial filter and additional steps will be required prior to any potential field trial, including engagement with stakeholders (e.g. land managers, local communities) and regulators to determine final approval (Campbell et al., 2019).

Mus musculus are non-native in countries within the GBIRd partnership (Australia, New Zealand, USA). Mice are not consumed as a food item by people in these countries; negatively impact native species, stored foods, crops, and infrastructure and can carry zoonotic diseases that impact the health of people and their livestock (Stenseth, et al., 2003; Meerburg, et al., 2009; Capizzi, et al., 2014), likely increasing socio-political acceptability. Further, these countries have (or are expected to have) appropriate regulatory capacity and systems established to evaluate a GBIRd proposal, if one is submitted (Campbell et al., 2019). Idealised island selection criteria for potential trials within these countries are provided in Table 1.

CONCLUSION

Gene drives hold enormous potential for application against invasive species and there is increasing interest in adapting them to this purpose. As a transformative but controversial set of technologies, it is important that the first instances of their use in the field are successful, both in terms of efficacy and safety. As discussed, the likelihood of a successful trial can be increased by making appropriate decisions at multiple stages of a gene drive's development and deployment. Making these decisions requires input from a broad range of scientific disciplines (Gould, 2008; Piaggio, et al., 2017) involving, for example, conservationists identifying potential targets, ecologists advising on the biological appropriateness of these targets and efficiencies of different gene drive strategies, molecular biologists advising on the feasibility of building proposed designs, mathematical modellers devising the most efficient means of deploying these systems and, finally, managers who will ultimately advise on the logistic feasibility of deployment. Although described in a linear series, in practice this will require informed dialogue between all these parties from the outset – there is no point in developing a system that performs well in computer models or in the lab if it is ultimately deemed impractical to deploy in the field. With proof-of-principle suppression (Hammond, et al., 2016) and replacement (Gantz, et al., 2015) drives functional in anopheline mosquitoes, it is critical that these conversations begin now to ensure these technologies are applied as safely, efficiently and rapidly as possible to the control of invasive species.

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REFERENCES

AAS. (2017). Synthetic gene drives in Australia: implications of emerging technologies. Australian Academy of Science. <www.science.org.au/gene-drives>. Accessed 17 November 2017.

Akbari, O.S., Matzen, K.D., Marshall, J.M., Huang, H.X., Ward, C.M. and Hay, B.A. (2013). 'A synthetic gene drive system for local, reversible modification and suppression of insect populations'. *Current Biology* 23(8): 671–677.

Akbari, O.S., Bellen, H.J., Bier, E., Bullock, S.L., Burt, A., Church, G.M., Cook, K.R., Duchek, P., Edwards, O.R., Esvelt, K.M., Gantz, V.M., Golic, K.G., Gratz, S.J., Harrison, M.M., Hayes, K.R., James, A.A., Kaufman, T.C., Knoblich, J., Malik, H.S., Matthews, K.A., O'Connor-Giles, K.M., Parks, A.L., Perrimon, N., Port, F., Russell, S., Ueda, R. and Wildonger, J. (2015). 'Safeguarding gene drive experiments in the laboratory'. *Science* 349(6251): 927–929.

Allen, M.L., O'Brochta, D.A., Atkinson, P.W. and Levesque, C.S. (2001). 'Stable, germ-line transformation of *Culex quinquefasciatus* (Diptera : Culicidae)'. *Journal of Medical Entomology* 38(5): 701–710.

Alphey, L. (2002). 'Re-engineering the sterile insect technique'. *Insect Biochemistry and Molecular Biology* 32(10): 1243–1247.

Alphey, L., Benedict, M., Bellini, R., Clark, G.G., Dame, D.A., Service, M.W. and Dobson, S.L. (2010). 'Sterile-insect methods for control of mosquito-borne diseases: An analysis'. *Vector-Borne and Zoonotic Diseases* 10(3): 295–311.

Alphey, L. (2014). 'Genetic control of mosquitoes'. *Annual Review of Entomology* 59: 205–224.

Amori, G., Gippoliti, S. and Helgen, K.M. (2008). 'Diversity, distribution, and conservation of endemic island rodents'. *Quaternary International* 182: 6–15.

Angel, A., Wanless, R.M. and Cooper, J. (2009). 'Review of impacts of the introduced house mouse on islands in the Southern Ocean: are mice equivalent to rats?'. *Biological Invasions* 11(7): 1743–1754.

Aplin, K.P., Chesser, T. and ten Haven, J. (2003). 'Evolutionary Biology of the Genus *Rattus*: Profile of an Archetypal Rodent Pest'. In: G.R. Singleton, L.A. Hinds, C.J. Krebs and D.M. Spratt (eds.) *Rats, Mice and People: Rodent Biology and Management,* pp. 487–498. Canberra: Australian Centre for International Agricultural Research.

Arensburger, P., Megy, K., Waterhouse, R.M., Abrudan, J., Amedeo, P., Antelo, B., Bartholomay, L., Bidwell, S., Caler, E., Camara, F., Campbell, C.L., Campbell, K.S., Casola, C., Castro, M.T., Chandramouliswaran, I., Chapman, S.B., Christley, S., Costas, J., Eisenstadt, E., Feschotte, C., Fraser-Liggett, C., Guigo, R., Haas, B., Hammond, M., Hansson, B.S., Hemingway, J., Hill, S.R., Howarth, C., Ignell, R., Kennedy, R.C., Kodira, C.D., Lobo, N.F., Mao, C.H., Mayhew, G., Michel, K., Mori, A., Liu, N.N., Naveira, H., Nene, V., Nguyen, N., Pearson, M.D., Pritham, E.J., Puiu, D., Qi, Y.M., Ranson, H., Ribeiro, J.M.C., Roberston, H.M., Severson, D.W., Shumway, M., Stanke, M., Strausberg, R.L., Sun, C., Sutton, G., Tu, Z.J., Tubio, J.M.C., Unger, M.F., Vanlandingham, D.L., Vilella, A.J., White, O., White, J.R., Wondji, C.S., Wortman, J., Zdobnov, E.M., Birren, B., Christensen, B.M., Collins, F.H., Cornel, A., Dimopoulos, G., Hannick, L.I., Higgs, S., Lanzaro, G.C., Lawson, D., Lee, N.H., Muskavitch, M.A.T., Raikhel, A.S. and Atkinson, P.W. (2010). 'Sequencing of *Culex quinquefasciatus* establishes a platform for mosquito comparative genomics'. *Science* 330(6000): 86–88.

Backus, G.A. and Gross, K. (2016). 'Genetic engineering to eradicate invasive mice on islands: modelling the efficiency and ecological impacts'. *Ecosphere* 7(12): 01589. 10.1002/ecs2.1589.

Banks, P.B. and Hughes, N.K. (2012). 'A review of the evidence for potential impacts of black rats (*Rattus rattus*) on wildlife and humans in Australia'. *Wildlife Research* 39(1): 78–88.

Bax, N.J. and Thresher, R.E. (2009). 'Ecological, behavioral, and genetic factors influencing the recombinant control of invasive pests'. *Ecological Applications* 19(4): 873–888.

Beaghton, A., Beaghton, P.J. and Burt, A. (2016). 'Gene drive through a landscape: Reaction-diffusion models of population suppression and elimination by a sex ratio distorter'. *TheoreticalPpopulation Biology* 108: 51–69.

Bellemain, E. and Ricklefs, R.E. (2008). 'Are islands the end of the colonization road?'. *Trends in Ecology and Evolution* 23(8): 461–468.

Benedict, M., D'Abbs, P., Dobson, S., Gottlieb, M., Harrington, L., Higgs, S., James, A., James, S., Knols, B., Lavery, J., O'Neill, S., Scott, T., Takken, W., Toure, Y. and Containe, C.W.G.G. (2008). 'Guidance for contained field trials of vector mosquitoes engineered to contain a gene drive system: Recommendations of a scientific working group'. *Vector-Borne and Zoonotic Diseases* 8(2): 127–166.

Besansky, N. (2015). Ecological & Evolutionary Conditions for Gene-flow in Mosquitoes. *Science, Ethics, and Governance Considerations for Gene Drive Research.* National Academies of Science, Engineering and Medicine. <http://nas-sites.org/gene-drives/2015/10/03/second-public-meeting/>. Accessed 17 November 2017.

Bonsall, M.B., Yakob, L., Alphey, N. and Alphey, L. (2010). 'Transgenic control of vectors: The effects of interspecific interactions'. *Israel Journal of Ecology and Evolution* 56(3–4): 353–370.

Bourtzis, K., Lees, R.S., Hendrichs, J. and Vreysen, M.J.B. (2016). 'More than one rabbit out of the hat: Radiation, transgenic and symbiont-based approaches for sustainable management of mosquito and tsetse fly populations'. *Acta Tropica* 157: 115–130.

Brown, D.M., Alphey, L.S., McKemey, A., Beech, C. and James, A.A. (2014). 'Criteria for identifying and evaluating candidate sites for open-field trials of genetically engineered mosquitoes'. *Vector-Borne and Zoonotic Diseases* 14(4): 291–299.

Bull, J.J. (2015). 'Evolutionary decay and the prospects for long-term disease intervention using engineered insect vectors'. *Evolution, Medicine and Public Health* 2015(1): 152–166.

Bull, J.J. (2016). 'OUP: Lethal gene drive selects inbreeding'. *Evolution, Medicine and Public Health* 2017(1): 1–16.

Burt, A. (2003). 'Site-specific selfish genes as tools for the control and genetic engineering of natural populations'. *Proceedings of the Royal Society B-Biological Sciences* 270(1518): 921–928.

Campbell, K.J., Beek, J., Eason, C.T., Glen, A.S., Godwin, J., Gould, F., Holmes, N.D., Howald, G.R., Madden, F.M., Ponder, J.B., Threadgill, D.W., Wegmann, A.S. and Baxter, G.S. (2015). 'The next generation of rodent eradications: Innovative technologies and tools to improve species specificity and increase their feasibility on islands'. *Biological Conservation* 185: 47–58.

Campbell, K.J. Saah, J.R., Brown, P.R., Godwin, J., Gould, F., Howald, G.R., Piaggio, A., Thomas, P., Tompkins, D.M., Threadgill, D., Delborne, J., Kanavy, D.M., Kuikin, T., Packard, H., Serr, M. and Shiels, A. (2019). 'A potential new tool for the toolbox: assessing gene drives for eradicating invasive rodent populations'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 6–14. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Capizzi, D., Bertolino, S. and Mortelliti, A. (2014). 'Rating the rat: Global patterns and research priorities in impacts and management of rodent pests'. *Mammal Review* 44(2): 148–162.

Clobert, J., Massot, M. and Le Galliard, J. (2012). 'Multi-determinism in Natal Dispersal: The Common Lizard as a Model System'. In: J. Clobert, M. Baguette, T.G. Benton and J.M. Bullock (eds.) *Dispersal Ecology and Evolution*, pp. 29–40. Oxford, UK: Oxford University Press.

Crisp, A., Boschetti, C., Perry, M., Tunnacliffe, A. and Micklem, G. (2015). 'Expression of multiple horizontally acquired genes is a hallmark of both vertebrate and invertebrate genomes'. *Genome Biology* 16: https://doi.org/10.1186/s13059-015-0607-3.

Deredec, A., Burt, A. and Godfray, H.C. (2008). 'The population genetics of using homing endonuclease genes in vector and pest management'. *Genetics* 179(4): 2013–2026.

DiCarlo, J.E., Chavez, A., Dietz, S.L., Esvelt, K.M. and Church, G.M. (2015). 'Safeguarding CRISPR-Cas9 gene drives in yeast'. *Nature Biotechnology* 33(12): 1250–1255.

Dlugosch, K.M. and Parker, I.M. (2008). 'Founding events in species invasions: Genetic variation, adaptive evolution, and the role of multiple introductions'. *Molecular Ecology* 17(1): 431–449.

Eckhoff, P.A., Wenger, E.A., Godfray, H.C.J. and Burt, A. (2017). 'Impact of mosquito gene drive on malaria elimination in a computational model with explicit spatial and temporal dynamics'. *Proceedings of the National Academy of Sciences* 114(2): E255–E264.

Eldridge, B.F. (2005). 'Mosquitoes, the Culicidae'. In: W.C. Marquardt and B.C. Kondratieff (eds.) *Biology of Disease Vectors*, pp. 95–111. London, UK: Elsevier Academic Press.

Esvelt, K.M., Smidler, A.L., Catteruccia, F. and Church, G.M. (2014). 'Concerning RNA-guided gene drives for the alteration of wild populations'. *eLife*: doi:10.7554/eLife.03401.

Fritts, E.I. (2007). *'Wildlife and People at Risk: A Plan to Keep Rats Out of Alaska*. Juneau, AK: Alaska Department of Fish and Game.

Gantz, V.M., Jasinskiene, N., Tatarenkova, O., Fazekas, A., Macias, V.M., Bier, E. and James, A.A. (2015). 'Highly efficient Cas9-mediated gene drive for population modification of the malaria vector mosquito *Anopheles stephensi'*. *Proceedings of the National Academy of Sciences* 112(49): E6736-E6743.

Gibbs, R.A., Weinstock, G.M., Metzker, M.L., Muzny, D.M., Sodergren, E.J., Scherer, S., Scott, G., Steffen, D., Worley, K.C., Burch, P.E., Okwuonu, G., Hines, S., Lewis, L., DeRamo, C., Delgado, O., Dugan-Rocha, S., Miner, G., Morgan, M., Hawes, A., Gill, R., Holt, R.A., Adams, M.D., Amanatides, P.G., Baden-Tillson, H., Barnstead, M., Chin, S., Evans, C.A., Ferriera, S., Fosler, C., Glodek, A., Gu, Z.P., Jennings, D., Kraft, C.L., Nguyen, T., Pfannkoch, C.M., Sitter, C., Sutton, G.G., Venter, J.C., Woodage, T., Smith, D., Lee, H.M., Gustafson, E., Cahill, P., Kana, A., Doucette-Stamm, L., Weinstock, K., Fechtel, K., Weiss, R.B., Dunn, D.M., Green, E.D., Blakesley, R.W., Bouffard, G.G., de Jong, J., Osoegawa, K., Zhu, B.L., Marra, M., Schein, J., Bosdet, I., Fjell, C., Jones, S., Krzywinski, M., Mathewson, C., Siddiqui, A., Wye, N., McPherson, J., Zhao, S.Y., Fraser, C.M., Shetty, J., Shatsman, S., Geer, K., Chen, Y.X., Abramzon, S., Nierman, W.C., Gibbs, R.A., Weinstock, G.M., Havlak, P.H., Chen, R., Durbin, K.J., Egan, A., Ren, Y.R., Song, X.Z., Li, B.S., Liu, Y., Qin, X., Cawley, S., Weinstock, G.M., Worley, K.C., Cooney, A.J., Gibbs, R.A., D'Souza, L.M., Martin, K., Wu, J.Q., Gonzalez-Garay, M.L., Jackson, A.R., Kalafus, K.J., McLeod, M.P., Milosavljevic, A., Virk, D., Volkov, A., Wheeler, D.A., Zhang, Z.D., Bailey, J.A., Eichler, E.E., Tuzun, E., Birney, E., Mongin, E., Ureta-Vidal, A., Woodwark, C., Zdobnov, E., Bork, P., Suyama, M., Torrents, D., Alexandersson, M., Trask, B.J., Young, J.M., Smith, D., Huang, H., Fechtel, K., Wang, H.J., Xing, H.M., Weinstock, K., Daniels, S., Gietzen, D., Schmidt, J., Stevens, K., Vitt, U., Wingrove, J., Camara, F., Schmidt, J., Stevens, K., Vitt, U., Wingrove, J., Camara, F., Alba, M.M., Abril, J.F., Guigo, R., Smit, A., Dubchak, I., Rubin, E.M., Couronne, O., Poliakov, A., Hubner, N., Ganten, D., Goesele, C., Hummel, O., Kreitler, T., Lee, Y.A., Monti, J., Schulz, H., Zimdahl, H., Himmelbauer, H., Lehrach, H., Jacob, H.J., Bromberg, S., Gullings-Handley, J., Jensen-Seaman, M.I., Kwitek, A.E., Lazar, J., Pasko, D., Tonellato, P.J., Twigger, S., Ponting, P., Duarte, J.M., Rice, S., Goodstadt, L., Beatson, S.A., Emes, R.D., Winter, E.E., Webber, C., Brandt, P., Nyakatura, G., Adetobi, M., Chiaromonte, F., Elnitski, L., Eswara, P., Hardison, R.C., Hou, M.M., Kolbe, D., Makova, K., Miller, W., Nekrutenko, A., Riemer, C., Schwartz, S., Taylor, J., Yang, S., Zhang, Y., Lindpaintner, K., Andrews, T.D., Caccamo, M., Clamp, M., Clarke, L., Curwen, V., Durbin, R., Eyras, E., Searle, S.M., Cooper, G.M., Batzoglou, S., Brudno, M., Sidow, A., Stone, E.A., Venter, J.C., Payseur, B.A., Bourque, G., Lopez-Otin, C., Puente, X.S., Chakrabarti, K., Chatterji, S., Dewey, C., Pachter, L., Bray, N., Yap, V.B., Caspi, A., Tesler, G., Pevzner, P.A., Haussler, D., Roskin, K.M., Baertsch, R., Clawson, H., Furey, T.S., Hinrichs, A.S., Karolchik, D., Kent, W.J., Rosenbloom, K.R., Trumbower, H., Weirauch, M., Cooper, D.N., Stenson, P.D., Ma, B., Brent, M., Arumugam, M., Shteynberg, D., Copley, R.R., Taylor, M.S., Riethman, H., Mudunuri, U., Peterson, J., Guyer, M., Felsenfeld, A., Old, S., Mockrin, S., Collins, F. and Cons, R.G.S.P. (2004). 'Genome sequence of the brown Norway rat yields insights into mammalian evolution'. *Nature* 428(6982): 493–521.

Glen, A.S., Atkinson, R., Campbell, K.J., Hagen, E., Holmes, N.D., Keitt, B.S., Parkes, J.P., Saunders, A., Sawyer, J. and Torres, H. (2013). 'Eradicating multiple invasive species on inhabited islands: The next big step in island restoration?'. *Biological Invasions* 15(12): 2589–2603.

Gould, F. (2008). 'Broadening the application of evolutionarily based genetic pest management'. *Evolution* 62(2): 500–510.

Hamilton, W.D. (1967). 'Extraordinary sex ratios'. *Science* 156(3774): 477–488.

Hammond, A., Galizi, R., Kyrou, K., Simoni, A., Siniscalchi, C., Katsanos, D., Gribble, M., Baker, D., Marois, E., Russell, S., Burt, A., Windbichler, N., Crisanti, A. and Nolan, T. (2016). 'A CRISPR-Cas9 gene drive system targeting female reproduction in the malaria mosquito vector *Anopheles gambiae'*. *Nature biotechnology* 34(1): 78–83.

Harper, G. and Cabrera, L. (2010). 'Response of mice (*Mus musculus*) to the removal of black rats (*Rattus rattus*) in arid forest on Santa Cruz Island, Galápagos'. *Biological Invasions* 12(6): 1449–1452.

Harper, G. and Bunbury, N. (2015). 'Invasive rats on tropical islands: Their population biology and impacts on native species'. *Global Ecology and Conservation* 3: 607–627.

Harris, D., Gregory, S.D., Bull, L. and Courchamp, F. (2012). 'Island prioritization for invasive rodent eradications with an emphasis on reinvasion risk'. *Biological Invasions* 14(6): 1251–1263.

Harvey-Samuel, T., Ant, T. and Alphey, L. (2017). 'Towards the genetic control of invasive species'. *Biological Invasions* 19(6): 1683–1703.

Hodgins, K.A., Rieseberg, L. and Otto, S.P. (2009). 'Genetic control of invasive plants species using selfish genetic elements'. *Evolutionary Applications* 2(4): 555–569.

Hoffmann, A.A., Montgomery, B.L., Popovici, J., Iturbe-Ormaetxe, I., Johnson, P.H., Muzzi, F., Greenfield, M., Durkan, M., Leong, Y.S., Dong, Y., Cook, H., Axford, J., Callahan, A.G., Kenny, N., Omodei, C., McGraw, E.A., Ryan, P.A., Ritchie, S.A., Turelli, M. and O'Neill, S.L. (2011). 'Successful establishment of *Wolbachia* in *Aedes* populations to suppress dengue transmission'. *Nature* 476(7361): 454–U107.

Hoffmann, A.A., Iturbe-Ormaetxe, I., Callahan, A.G., Phillips, B., Billington, K., Axford, J.K., Montgomery, B., Turley, A.P. and O'Neill, S.L. (2014). 'Stability of the *w*Mel *Wolbachia* infection following invasion into *Aedes aegypti* populations'. *PLOS Neglected Tropical Diseases* 8(9): e3115.

Huang, Y.X., Lloyd, A.L., Legros, M. and Gould, F. (2011). 'Gene drive into insect populations with age and spatial structure: A theoretical assessment'. *Evolutionary Applications* 4(3): 415–428.

Ivics, Z., Mates, L., Yau, T.Y., Landa, V., Zidek, V., Bashir, S., Hoffmann, O.I., Hiripi, L., Garrels, W., Kues, W.A., Bosze, Z., Geurts, A., Pravenec, M., Rulicke, T. and Izsvak, Z. (2014). 'Germline transgenesis in rodents by pronuclear microinjection of Sleeping Beauty transposons'. *Nature Protocols* 9(4): 773–793.

Kawecki, T. (2004). 'Ecological and Evolutionary Consequences of Source-Sink Population Dynamics'. In*:* Island Hanski and O.E. Gaggiotti (eds.) *Ecology, Genetics and Evolution of Metapopulations*, pp. 387–414. Boston: Elsevier.

Klassen, W. and Curtis, C.F. (2005). 'History of the Sterile Insect Technique'. In*:* V.A. Dyck, J. Hendrichs and A.S. Robinson (eds.) *Sterile Insect Technique: Principles and Practice in Area-Wide Integrated Pest Management*, pp. 3–36. Dordrect, Netherlands: Springer.

Kolar, C.S. and Lodge, D.M. (2001). 'Progress in invasion biology: Predicting invaders'. *Trends in Ecology and Evolution* 16(4): 199–204.

LaPointe, D.A., Atkinson, C.T. and Samuel, M.D. (2012). 'Ecology and conservation biology of avian malaria'. *Annals of the New York Academy of Sciences* 1249: 211–226.

Lavery, J.V., Harrington, L.C. and Scott, T.W. (2008). 'Ethical, social, and cultural considerations for site selection for research with genetically modified mosquitoes'. *American Journal of Tropical Medicine and Hygiene* 79(3): 312–318.

Marshall, J.M. and Hay, B.A. (2012). 'Confinement of gene drive systems to local populations: A comparative analysis'. *Journal of Theoretical Biology* 294: 153–171.

Matthysen, E. (2012). 'Multicausality of Dispersal: A Review'. In*:* J. Clobert, M. Baguette, T.G. Benton and J.M. Bullock (eds.) *Dispersal Ecology and Evolution*, pp. 3–18. Oxford, UK: Oxford University Press.

Meerburg, B.G., Singleton, G.R. and Kijlstra, A. (2009). 'Rodent-borne diseases and their risks for public health'. *Critical Reviews in Microbiology* 35(3): 221–270.

Min, J., Noble, C., Najjar, D. and Esvelt, K. (2017). 'Daisy quorum drives for the genetic restoration of wild populations'. *bioRχiv* http://dx.doi.org/101101/115618.

Mooney, H.A. and Cleland, E.E. (2001). 'The evolutionary impact of invasive species'. *Proceedings of the National Academy of Sciences* 98(10): 5446–5451.

NASEM. (2016). *Gene Drives on the Horizon: Advancing Science, Navigating Uncertainty, and Aligning Research with Public Values,* Washington (DC): The National Academies Press.

Noble, C., Min, J., Olejarz, J., Buchthal, J., Chavez, A., Smidler, A., DeBenedictis, E., Church, G., Nowak, M. and Esvelt, K. (2016). 'Daisy-chain gene drives for the alteration of local populations':. *bioRχiv*: https://doi.org/10.1101/057307.

North, A., Burt, A. and Godfray, H.C.J. (2013). 'Modelling the spatial spread of a homing endonuclease gene in a mosquito population'. *Journal of Applied Ecology* 50(5): 1216–1225.

Oppel, S., Beaven, B.M., Bolton, M., Vickery, J. and Bodey, T.W. (2011). 'Eradication of invasive mammals on islands inhabited by humans and domestic animals'. *Conservation Biology* 25(2): 232–240.

Ortiz, M.F., Wallau, G.L., Graichen, D.A.S. and Loreto, E.L.S. (2015). 'An evaluation of the ecological relationship between *Drosophila* species and their parasitoid wasps as an opportunity for horizontal transposon transfer'. *Molecular Genetics and Genomics* 290(1): 67–78.

Oye, K.A., Esvelt, K., Appleton, E., Catteruccia, F., Church, G., Kuiken, T., Lightfoot, S.B.Y., McNamara, J., Smidler, A. and Collins, J.P. (2014). 'Regulating gene drives'. *Science* 345(6197): 626–628.

Piaggio, A.J., Segelbacher, G., Seddon, P.J., Alphey, L., Bennett, E.L., Carlson, R.H., Friedman, R.M., Kanavy, D., Phelan, R., Redford, K.H., Rosales, M., Slobodian, L. and Wheeler, K. (2017). 'Is it time for synthetic biodiversity conservation?'. *Trends in Ecology and Evolution* 32(2): 97–107.

Pimentel, D., Zuniga, R. and Morrison, D. (2005). 'Update on the environmental and economic costs associated with alien-invasive species in the United States'. *Ecological Economics* 52(3): 273–288.

Rambuda, T.D. and Johnson, S.D. (2004). 'Breeding systems of invasive alien plants in South Africa: Does Baker's rule apply?'. *Diversity and Distributions* 10(5–6): 409–416.

Reeves, R.G., Bryk, J., Altrock, P.M., Denton, J.A. and Reed, F.A. (2014). 'First steps towards underdominant genetic transformation of insect populations'. *PLOS ONE* 9(5): e97557.

Rhymer, J.M. and Simberloff, D. (1996). 'Extinction by hybridization and introgression'. *Annual Review of Ecology and Systematics* 27: 83–109.

Russell, J. and Clout, M. (2005). 'Rodent Incursions on New Zealand islands'. In: J.Parkes, M.Statham and G Edwards,. (Comps.) *Proceedings of the 13th Australasian Vertebrate Pest Conference*, pp. 324–330. Lincoln, NZ: Landcare Research,.

Russell, J.C., Towns, D.R. and Clout, M. (2008). 'Review of Rat Invasion Biology: Implications for Island Biosecurity'. *Science for Conservation* 286, Wellington, NZ: Department of Conservation.

Schmidt, T.L., Barton, N.H., Rasic, G., Turley, A.P., Montgomery, B.L., Iturbe-Ormaetxe, I., Cook, P.E., Ryan, P.A., Ritchie, S.A., Hoffmann, A.A., O'Neill, S.L. and Turelli, M. (2017). 'Local introduction and heterogeneous spatial spread of dengue-suppressing *Wolbachia* through an urban population of *Aedes aegypti'*. *PLOS Biology* 15(5): e2001894.

Simberloff, D. (2014). 'Biological invasions: What's worth fighting and what can be won?'. *Ecological Engineering* 65: 112–121.

Sinkins, S.P. and Gould, F. (2006). 'Gene drive systems for insect disease vectors'. *Nature Reviews Genetics* 7(6): 427–435.

Stenseth, N.C., Leirs, H., Skonhoft, A., Davis, S.A., Pech, R.P., Andreassen, H.P., Singleton, G.R., Lima, M., Machang'u, R.S. and Makundi, R.H. (2003). 'Mice, rats, and people: The bio-economics of agricultural rodent pests'. *Frontiers in Ecology and the Environment* 1(7): 367–375.

Sutherland, W.J., Aveling, R., Brooks, T.M., Clout, M., Dicks, L.V., Fellman, L., Fleishman, E., Gibbons, D.W., Keim, B., Lickorish, F., Monk, K.A., Mortimer, D., Peck, L.S., Pretty, J., Rockstrom, J., Rodriguez, J.P., Smith, R.K., Spalding, M.D., Tonneijck, F.H. and Watkinson, A.R. (2014). 'A horizon scan of global conservation issues for 2014'. *Trends in Ecology and Evolution* 29(1): 15–22.

Sytsma, M.D., Phillips, S. and Counihan, T.D. (2015). 'Dreissenid Mussel Research Priorities Workshop'. *Center for Lakes and Resevoirs Publications and Presentations* 49: 11.

Thomas, D.D., Donnelly, C.A., Wood, R.J. and Alphey, L.S. (2000). 'Insect population control using a dominant, repressible, lethal genetic system'. *Science* 287(5462): 2474–2476.

Thresher, R.E., Canning, M. and Bax, N.J. (2013). 'Demographic effects on the use of genetic options for the control of mosquitofish, *Gambusia holbrooki'*. *Ecological Applications* 23(4): 801–814.

Thresher, R.E., Hayes, K., Bax, N.J., Teem, J., Benfey, T.J. and Gould, F. (2014). 'Genetic control of invasive fish: Technological options and its role in integrated pest management'. *Biological Invasions* 16(6): 1201–1216.

Tobin, P.C., Berec, L. and Liebhold, A.M. (2011). 'Exploiting Allee effects for managing biological invasions'. *Ecology Letters* 14(6): 615–624.

Unckless, R.L., Messer, P.W., Connallon, T. and Clark, A.G. (2015). 'Modeling the manipulation of natural populations by the mutagenic chain reaction'. *Genetics* 201(2): 425–431.

Waterston, R.H., Lindblad-Toh, K., Birney, E., Rogers, J., Abril, J.F., Agarwal, P., Agarwala, R., Ainscough, R., Alexandersson, M., An, P., Antonarakis, S.E., Attwood, J., Baertsch, R., Bailey, J., Barlow, K., Beck, S., Berry, E., Birren, B., Bloom, T., Bork, P., Botcherby, M., Bray, N., Brent, M.R., Brown, D.G., Brown, S.D., Bult, C., Burton, J., Butler, J., Campbell, R.D., Carninci, P., Cawley, S., Chiaromonte, F., Chinwalla, A.T., Church, D.M., Clamp, M., Clee, C., Collins, F.S., Cook, L.L., Copley, R.R., Coulson, A., Couronne, O., Cuff, J., Curwen, V., Cutts, T., Daly, M., David, R., Davies, J., Delehaunty, K.D., Deri, J., Dermitzakis, E.T., Dewey, C., Dickens, N.J., Diekhans, M., Dodge, S., Dubchak, I., Dunn, D.M., Eddy, S.R., Elnitski, L., Emes, R.D., Eswara, P., Eyras, E., Felsenfeld, A., Fewell, G.A., Flicek, P., Foley, K., Frankel, W.N., Fulton, L.A., Fulton, R.S., Furey, T.S., Gage, D., Gibbs, R.A., Glusman, G., Gnerre, S., Goldman, N., Goodstadt, L., Grafham, D., Graves, T.A., Green, E.D., Gregory, S., Guigo, R., Guyer, M., Hardison, R.C., Haussler, D., Hayashizaki, Y., Hillier, L.W., Hinrichs, A., Hlavina, W., Holzer, T., Hsu, F., Hua, A., Hubbard, T., Hunt, A., Jackson, I., Jaffe, D.B., Johnson, L.S., Jones, M., Jones, T.A., Joy, A., Kamal, M., Karlsson, E.K., Karolchik, D., Kasprzyk, A., Kawai, J., Keibler, E., Kells, C., Kent, W.J., Kirby, A., Kolbe, D.L., Korf, I., Kucherlapati, R.S., Kulbokas, E.J., Kulp, D., Landers, T., Leger, J.P., Leonard, S., Letunic, I., Levine, R., Li, J., Li, M., Lloyd, C., Lucas, S., Ma, B., Maglott, D.R., Mardis, E.R., Matthews, L., Mauceli, E., Mayer, J.H., McCarthy, M., McCombie, W.R., McLaren, S., McLay, K., McPherson, J.D., Meldrim, J., Meredith, B., Mesirov, J.P., Miller, W., Miner, T.L., Mongin, E., Montgomery, K.T., Morgan, M., Mott, R., Mullikin, J.C., Muzny, D.M., Nash, W.E., Nelson, J.O., Nhan, M.N., Nicol, R., Ning, Z., Nusbaum, C., O'Connor, M.J., Okazaki, Y., Oliver, K., Larty, E.O., Pachter, L., Parra, G., Pepin, K.H., Peterson, J., Pevzner, P., Plumb, R., Pohl, C.S., Poliakov, A., Ponce, T.C., Ponting, C.P., Potter, S., Quail, M., Reymond, A., Roe, B.A., Roskin, K.M., Rubin, E.M., Rust, A.G., Santos, R., Sapojnikov, V., Schultz, B., Schultz, J., Schwartz, M.S., Schwartz, S., Scott, C., Seaman, S., Searle, S., Sharpe, T., Sheridan, A., Shownkeen, R., Sims, S., Singer, J.B., Slater, G., Smit, A., Smith, D.R., Spencer, B., Stabenau, A., Strange-Thomann, N.S., Sugnet, C., Suyama, M., Tesler, G., Thompson, J., Torrents, D., Trevaskis, E., Tromp, J., Ucla, C., Vidal, A.U., Vinson, J.P., von Niederhausern, A.C., Wade, C.M., Wall, M., Weber, R.J., Weiss, R.B., Wendl, M.C., West, A.P., Wetterstrand, K., Wheeler, R., Whelan, S., Wierzbowski, J., Willey, D., Williams, S., Wilson, R.K., Winter, E., Worley, K.C., Wyman, D., Yang, S., Yang, S.P., Zdobnov, E.M., Zody, M.C., Lander, E.S. and Consor, M.G.S. (2002). 'Initial sequencing and comparative analysis of the mouse genome'. *Nature* 420(6915): 520–562.

Webber, B.L., Raghu, S. and Edwards, O.R. (2015). 'Opinion: Is CRISPR-based gene drive a biocontrol silver bullet or global conservation threat?'. *Proceedings of the National Academy of Sciences* 112(34): 10565–10567.

WHO/TDR (2014). *The Guidance Framework for Testing Genetically Modified Mosquitoes,* Geneva, Switzerland: World Health Organisation.

Wu, B., Luo, L. and Gao, X.J. (2016). 'Cas9-triggered chain ablation of cas9 as a gene drive brake'. *Nature Biotechnology* 34: 137–138.

Yakob, L., Alphey, L. and Bonsall, M.B. (2008). '*Aedes aegypti* control: The concomitant role of competition, space and transgenic technologies'. *Journal of Applied Ecology* 45(4): 1258–1265.

Yakob, L. and Bonsall, M.B. (2009). 'Importance of space and competition in optimizing genetic control strategies'. *Journal of Economic Entomology* 102(1): 50–57.

Tracking invasive species eradications on islands at a global scale

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Abstract Indicators for tracking conservation efforts at a global scale are rare but important tools for understanding trends and measuring progress towards global conservation targets. Eradication of invasive species from islands is an increasingly used conservation intervention in countries and territories around the world. With a goal of collating these efforts, the Database of Islands and Invasive Species Eradications (DIISE) holds records of the location, target species, year and outcome of invasive mammal and bird eradications on islands from around the world. The database is publicly available in Spanish and English, at <diiise.islandconservation.org>, and represents a partnership between the University of California at Santa Cruz, University of Auckland, IUCN Invasive Species Specialist Group, Landcare Research and Island Conservation. The database holds records for more than 1,200 eradication attempts. This database will continue to be added to and evolve as new opportunities for its application arise; thus, we expect these numbers to change over time as new events are added and knowledge about existing events improves. Updating the DIISE relies on contributions from experts and reporting from island restoration activities. Here we present database history, parameter definitions and database considerations. We also highlight additional studies the underlying data have contributed to, including evaluating the native species benefit from invasive mammal eradications on islands, and global indices to track progress towards the Convention of Biological Diversity Aichi target 9 (Invasive Alien Species), that explicitly requires an increased effort of eradication of priority invasive species.

Keywords: database, eradication, global, invasive, islands, mammal

INTRODUCTION

Eradication of invasive species from islands is an increasingly used conservation intervention in countries and territories around the world. Indicators for tracking conservation efforts at a global scale are rare but important tools for understanding trends, and measuring progress towards global conservation targets (McGeoch, et al., 2010). The number of eradications of invasive species on islands is one response indicator that contributes to measuring such progress. The number of eradications of invasive species on islands is a particularly good metric as these events tend to take place over discrete periods of time, occur in clearly defined spatial areas, and have a clear measure of success or failure (Niemeijer & de Groot, 2008).

With a goal of collating these efforts, the Database of Islands and Invasive Species Eradications (DIISE) holds records of, at a minimum, the location, target species, year and outcome of invasive mammal and bird eradications on islands around the world. Data within the database focus on terrestrial vertebrate species, primarily mammals and birds. Fish eradications are not included, nor are plant or invertebrate eradications (but see Tobin, et al., 2014; Hoffmann, et al., 2016). As of 2016, the database holds records for more than 1,200 eradication attempts. The database is publicly available in Spanish and English, at <diiise.islandconservation.org>, and represents an ongoing partnership between the University of California at Santa Cruz, University of Auckland, IUCN Invasive Species Specialist Group, Landcare Research and Island Conservation.

Here we present database history, parameter definitions and database considerations. During 2017, a major update to the data is underway with a goal of using the 2017 Island Invasives Conference as a venue to engage island restoration practitioners to help improve the dataset.

MATERIALS AND METHODS

Database history

The first synthesis of the database (then known as the Global Islands Invasive Vertebrate Eradication Database) was published in the proceedings of the Island Invasives: eradication and management conference in Auckland in 2010 (Keitt, et al., 2011). Data for this synthesis were gathered from published, grey and unpublished literature, with the majority of data from reviews of eradications for rodents (Howald, et al., 2007), goats (Campbell & Donlan, 2005) and cats (Nogales, et al., 2004; Campbell, et al., 2011). Following the conference, the database was shared with all of the attendees of the conference (240 topic experts from 20 countries) with the goals of checking facts and adding missing eradication events. Attendees were encouraged to share the database with their networks to help achieve these goals.

In 2013–2014 an update of the database was undertaken using additional review papers on invasive mice (MacKay, et al., 2007) and small Indian mongoose (Barun, et al., 2011), the two Island Invasives conference proceedings (Veitch & Clout, 2002; Veitch, et al., 2011), summaries of eradication on inhabited islands (Oppel, et al., 2011; Glen, et al., 2013) and regional summaries for New Zealand (Clout & Russell, 2006), Europe and overseas territories (Genovesi, 2005; Genovesi & Carnevali, 2011), USA and territories (Witmer & Fuller, 2011), Galapagos (Carrion, et al., 2011; Harper & Carrion, 2011; Phillips, et al., 2012), California Channel and north-western Baja California Islands (McChesney & Tershy, 1998), Mexico (Aguirre-Muñoz, et al., 2008; Aguirre-Muñoz, et al., 2011), Hawaii and Central Pacific (Hess & Jacobi, 2011), France and overseas territories (Lorvelec & Pascal, 2005) and Seychelles (Beaver & Mougal, 2009).

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Other resources reviewed include, but were not limited to, IUCN SSC Invasive Species Specialist Group Invasives listserv, Pacific Seabird Group listserv, Pacific Invasives Initiative listserv; new sites including Agreement for the Conservation of Albatrosses and Petrels <http://acap.aq/news>, Seychelles Island Foundation newsletter <http://www.sif.sc/index.php?langue=eng&rub=19>; industry sources including the Australian Invasive Animals Cooperative Research Centre <https://www.pestsmart.org.au/tag/invasive-animals-cooperative-research-centre/> and <https://invasives.com.au/about/our-legacy/>, Mediterranean Small Islands Initiative <http://initiative-pim.org/> and the Web of Science for the key words "island" and "eradication". Further, we were fortunate to benefit from communications with practitioners who maintain regularly updated databases for territories including the Falklands / Islas Malvinas (Falkland Islands Rat Eradication Register, S. Poncet pers. comm.), France and overseas territories (O. Lorvelec pers. comm.), Seychelles (G. Rocamura pers. comm.), and worldwide (J. Parkes pers. comm.). This effort also included an evaluation period where entries were cross-checked with experts, and review of emails sent to directly to database managers.

During 2017, a third update began, including review of regional assessments including Italy (Capizzi, et al., 2016), Australia (Gregory, et al., 2014), California (California Department of Fish and Wildlife (CDFW, 2015) and the Indian Ocean (Russell, et al., 2016). Additional listservs and new sites reviewed include the NZ Department of Conservation media <http://www.doc.govt.nz/news/media-releases/>, South Pacific Regional Environment Program media <http://www.sprep.org/news>, Pacific Invasives Learning Network soundbites <http://www.sprep.org/piln/soundbites-documents>, Battler resource base <https://piln.sprep.org/>, and BirdLife news <http://www.birdlife.org/news>. The keyword 'eradication' was used to search these sites, plus the word 'deratisation' for French language sites. This review is expected to continue through 2017 including an expert review to validate new or changed entries.

Parameter definitions

Keitt et al (2011) describe the general methods used to populate the DIISE for the first synthesis. Each eradication event is an attempt to eradicate an invasive vertebrate population from an island. Where multiple invasive species are eradicated from an island these are considered separate eradication events, even if using the same technique. Each eradication event has a unique identification number and can generally be identified by the combination of the key parameters of species removed + island + eradication end date + eradication status. Citations for each eradication event are recorded.

For the 2013–2014 update, the parameter definitions were expanded to also include data quality, primarily to classify how eradication events were verified for inclusion the database. We assessed the quality of data available for all eradication attempts within the database using criteria in Table 1. We encourage other users of DIISE data to use data classified as good or satisfactory data quality event only. We retain events classified as poor data quality in the online database in the hope others can help us further qualify or remove these events.

Each eradication event was linked to an island. Each island was given a unique ID based on the World Conservation Monitoring Centre (WCMC) Global Islands Database (GID) (Depraetere, 2007), a spatial dataset with 180,000 unique island locations of the world. Eradications on different islands were recorded as separate events, regardless of whether it was in the same archipelago or treated concurrently (e.g. Montebello islands in Western Australia). For coral atolls, if the project targeted individual motu these were treated as separate events and linked to individual motu accordingly. However, projects that occurred at the atoll scale were treated as one event. For islands that were not in the GID we allocated our own ID number and metadata. Locations were verified in Google Earth and corrected if necessary. Island names are standardised to the common proper noun within the larger country/territory, excluding frequently used words for 'island' (e.g. islets, rocks, etc.). Country or territory was based on International Standards Organization (ISO) 3166-1 alpha-2 codes. In 2016, the DIISE island locations were migrated to the GID2, a higher resolution product by WCMC that holds approximately 460,000 islands. Each polygon used for the DIISE was validated for island location and size against Google Earth and other satellite imagery.

Each invasive species has a unique ID code, and the common name, scientific name, family, trophic level (omnivore, herbivore, carnivore), and nominate type [amphibian, flying bird; non-flying bird; rodents (*Mus*); rodents (*Rattus*); cat; dogs or foxes, mongooses or weasels, rabbits or hares, reptiles (excluding snakes), snakes, ungulates, or other mammals] were recorded. Invasive species populations were either classified as feral, semi-feral, domestic, or a combination, with semi-feral defined as having some human care but not restricted in movement (e.g. fences).

We also sought to classify the eradication type, based upon the extent of the established invasive species population on the island and thus the scale of the operation necessary to achieve eradication. The aim of the database is to only include events where the goal was complete removal of an invasive species population from the island, and not removal from only part of an island such as fenced

Table 1 Data quality definitions

Data quality	Data quality definition
Good	We can verify the attempt; we have a copy of the primary reference (e.g. from a report, or peer reviewed publication) that details the effort, typically allowing us to populate almost all fields
Satisfactory	An expert practitioner has verified the event and/or we have limited information about an eradication but what we do have has come from a verifiable source (e.g. email from a reputable practitioner or cited in a review paper), and we can typically identify all of the following attributes: the island, end year (if applicable), invasive species type, eradication status, and primary eradication method
Poor	We cannot verify the attempt (conflicting information nor unverifiable resource) and/or we lack evidence for at least one of the following parameters: island, end year (if applicable), invasive animal type, eradication status, or primary eradication method
Unknown	The data quality has not yet been assessed for this event

areas (however, note we retain events where fences are used as a tool to achieve eradication at an island scale). We delineate whether the operation required treatment of the entire island, or only part of the island (restricted range), to achieve eradication of the invasive species population at the island scale. We also delineate between incursion responses and restricted range, whereby incursions represent operations to remove a recently arrived population prior to their spread across the island. If an incursion response fails, it is assumed a new eradication operation would be necessary. Although some incursion responses are recorded in the database, there is not a deliberate attempt to record every incursion response for each island because these may reflect a minor or ongoing management activity that may go unrecorded in the sources accessed. A classification of unknown is also used if it is unclear what the eradication type was, and this is also typically used where the cause of the extirpation of the invasive species population is unknown.

The timing of the eradication operation is typically based on the end date for the operation and is reported in years only. We considered eradication end date to be the year that major eradication operations ceased. This typically coincided with the end of hunting / trapping for ungulates and predators or the end of toxicant application (or other methods) for rodent projects. We note that monitoring required to determine if an operation was successful often occurs in years after the operation ending. The primary and secondary method of the eradication is collected, including disease, hunting, trapping, toxicant, other, or unknown. Where toxicant was used we sought to identify the baiting method, including aerial broadcast, bait station or bait piles, hand broadcast, unknown, or other, plus the toxicant compound used.

Eradication status is based on definitions in Table 2. When an eradication event is declared successful, the target invasive was removed from the entire island. We considered failures to be operational failures, i.e. the project did not successfully remove the entire invasive population. We considered reinvasion as separate to operational failure and recorded this separately. Reinvasion was defined as a previously successfully removed population becoming re-established back on the island. In the case of rodent eradications, reinvasion may also represent misdiagnosed failure (Russell, et al., 2010) but can be assessed through techniques such as genetic analyses, distance to potential source populations and the time elapsed between the eradication operation and subsequent rodent detections. When experts or source material indicated uncertainty about whether an invasive rodent population remains due to an operational failure or a reinvasion back onto the island, we assumed operational failure and classified data quality for the event as 'poor'.

DISCUSSION

Collating the location, method, outcome and target animal for invasive vertebrate eradications on islands offers a unique opportunity to contribute to global indicators for conservation. Collating these data over time offers insight into the response of a state-pressure-response model (Niemeijer & de Groot, 2008). The DIISE dataset holds many characteristics identified as necessary for effective threat (i.e. pressure in the state-pressure-response model) databases at a global scale, including: being freely available, spatially explicit, inclusion of a measure of expert validation, and is updated in a reasonable timeframe (Joppa, et al., 2016). The DIISE can contribute towards measuring progress of Aichi Target 9 of the global Convention on Biological Diversity, whereby signatory parties (nations) are committed to controlling or eradicating priority invasive alien species by 2020 (Convention on Biological Diversity, 2011), and is being used for the Biodiversity Indicator Partnership accordingly <https://www.bipindicators.net/indicators/trends-in-invasive-alien-species-vertebrate-eradications>.

The collation of more than 1,000 different eradication events inevitably encounters challenges. Reconciling the area (ha) and location (latitude and longitude) of small islands targeted for invasive species eradications against global data layers, has presented challenges to maintaining accuracy. In general, relying on one dataset (the GID) provides consistency, and seeking to validate those locations with satellite or other imagery should improve rigour. For rodent eradications, there is the risk that some projects classified as successful but reinvaded were in fact misdiagnosed operational failures. The time

Table 2 Eradication status definitions.

Eradication status	Definition
Successful	The operation to eradicate the invasive was successful and confirmed
Failed	The eradication operation was completed (there is an end date) yet it failed to remove the entire invasive population. Operational failure (as opposed to reinvasion). For rodent eradications, if there was uncertainty about why the invasive population remained (failure versus reinvasion), we assumed operational failure and classified data quality as 'poor'
To be confirmed	The eradication effort is complete, but the operation has yet to be "confirmed" as successful or failed. This stage is typical for rodent eradication operations, with confirmation monitoring occurring 1–2 years after the eradication operation has ended
In progress	Eradication operation is currently in progress at time of reporting
Planned	Eradication is being planned for the island at time of reporting. End year will be unknown accordingly
Incomplete	An eradication was started, but not followed through to completion
Trial or research only	The eradication was undertaken for trial or research purposes and the goal was to gain new knowledge, not eradicate invasive species
Unknown	Information does not allow allocation into one of the other mutually exclusive categories and an expert cannot do the same (e.g. unclear if an eradication took place or if the species "died out" naturally). Selection of this category will often be aligned with poor data quality
Unknown pre-status	Eradication was undertaken but the status of the invasive species was unclear beforehand. Typically undertaken for precautionary measures for rodent eradications

elapsed between the operation and invasion, and robust genetic analyses can confirm this classification (Russell, et al., 2010), but these may not be available on all projects, particularly islands that are not visited regularly, or for older projects where genetic tools were not available (Holmes, et al., 2015). In general, data in the DIISE rely on the eradication status provided by the practitioner. Including successful but reinvaded in data summaries may overestimate the success rate, but this can be mitigated by excluding those events. Similarly, outcomes of multiple adjacent islands that may function as a single eradication unit may skew success rates if they are treated as separate events. This can be accommodated for by selecting one representative island in that unit (e.g. see Holmes, et al., 2015).

Opportunities exist to improve and expand the schema and content of the DIISE. The DIISE is currently organised by island unit but currently does not link events based on operation (islands treated concurrently) or eradication unit (Abdelkrim, et al., 2005), whereby an invasive animal population may move freely between adjacent islands based on swimming or flying ability ('natural' reinvasion risk – Harris, et al., 2012). Most (98%) of the target animals in the DIISE are invasive mammals. A handful of bird eradications are recorded although they may require a different spatial organisational unit and consideration, particularly where entire archipelagos are invaded, and birds can move freely between islands. Some areas of the world may be under-represented in the database, including Small Island Developing States (Russell, et al., 2017) where resources to report outcomes may be scarcer, and the known lack of expert contacts in SE Asia, possibly reflecting a language barrier. More deliberate attempts to track these data may expand the dataset.

The DIISE dataset is freely available online, and requests for datasets to answer specific questions are responded to as best possible. There is a genuine resource cost to maintaining this data accessibility and a more significant investment required to undertake a major update. Thus, ensuring financial investment is key to maintaining this service. Despite the best of intentions, errors and omissions may occur in the dataset and, depending on the significance of the end goal users require the data for, additional validation of events in the DIISE may be warranted (e.g. Holmes, et al., 2015). A commonly sought-after use is summary statistics, for which we encourage those to check existing literature as they may already exist from sufficiently recent summaries (e.g. Russell & Holmes, 2015). For those seeking novel statistics not reported elsewhere, using only good or satisfactory data quality events is encouraged, as is being conscious of eradication type (whole island or restricted range). Events generating failure rates for rodents may need to consider that some reinvasion events may be misdiagnosed failures, and for events targeting species that have agricultural or domestic analogues (ungulates, dogs, cats), consideration may need to be given to whether domestic or feral populations are included. Using the data requires agreeing with a terms-of-use and checking with database managers is strongly encouraged to guide appropriate use of data.

Conservation databases provide a key role for informing decision making and assessing trends (e.g. the IUCN Red List) (Joppa, et al., 2016). At a project scale, data from the DIISE regularly features within feasibility assessments, by providing a comparison of proposed activities against past efforts. Data from the DIISE dataset has been used as a baseline to inform other conservation-based studies. Holmes, et al. (2015) and Russell & Holmes (2015) used the data to evaluate trends evident in why rodent eradication failed at higher rates in the tropics although note that predicting failure from operational covariates is not a panacea. Russell, et al. (2017) evaluated trends in where eradications occur, or may be under-reported, amongst different countries of the world. Importantly, recent efforts include Jones, et al. (2016) and Brooke, et al. (2017), who used validated DIISE data to explore biodiversity conservation outcomes, and seabird demographic response to invasive mammal eradications, respectively. Jones, et al. (2016) reported 596 populations of 236 native species on 181 islands benefiting after eradications. These types of studies are immensely valuable for measuring the true 'effect' (Kapos, et al., 2010) of eradication of invasive species on islands as a management action.

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REFERENCES

Abdelkrim, J., Pascal, M., Calmet, C. and Samadi, S. (2005). 'Importance of assessing population genetic structure before eradication of invasive species: Examples from insular Norway rat populations'. *Conservation Biology* 19(5): 1509–1518.

Aguirre-Muñoz, A., Croll, D.A., Donlan, C.J., Henry, R.W., III, Hermosillo, M.A., Howald, G.R., Keitt, B.S., Luna-Mendoza, L., Rodriguez-Malagón, M., Salas-Flores, L.M., Samaniego-Herrera, A., Sanchez-Pacheco, J.A., Sheppard, J., Tershy, B.R., Toro-Benito, J., Wolf, S. and Wood, B. (2008). 'High-impact conservation: Invasive mammal eradications from the islands of western Mexico'. *Ambio* 37(2): 101–107.

Aguirre-Muñoz, A., Samaniego-Herrera, A., Luna-Mendoza, L., Ortiz-Alcaraz, A., Rodriguez-Malagón, M., Méndez-Sánchez, F., Félix-Lizárraga, M., Hernández-Montoya, J.C., González-Gómez, R., Torres-Garcia, F., Barredo-Barberena, J.M. and Loatofski-Robles, M. (2011). 'Island restoration in Mexico: ecological outcomes after systematic eradications of invasive mammals'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 250–258. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Barun, A., Hanson, C.C., Campbell, K.J. and Simberloff, D. (2011). 'A review of small Indian mongoose management and eradications on islands'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 17–25. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Beaver, K. and Mougal, J. (2009). *Review of IAS control and eradication programmes in Seychelles*. With contributions from the Island Conservation Society on control and eradication programmes carried out under the ICS-FFEM Restoration of Island Ecosystems programme – provided by Gerard Rocamora. Prepared by the Plant Conservation Action group. <http://www.pcusey.sc/index.php/component/remository/BS/Project-Technical-Reports/Outcome-3/TechReports3/Review-of-IAS-Control-and-Eradication-Programmes-in-Seychelles-2009/?Itemid=>

Brooke, M.d.L., Bonnaud, E., Dilley, B.J., Flint, E.N., Holmes, N.D., Jones, H.P., Provost, P., Rocamora, G., Ryan, P.G., Surman, C. and Buxton, R.T. (2017). 'Seabird population changes following mammal eradications on islands'. *Animal Conservation* 21: 3–12.

CDFW (California Department of Fish and Wildlife) (2015). *California State Wildlife Action Plan, 2015 Update: A Conservation Legacy for Californians.* A.G. Gonzales and J. Hoshi (eds.). Prepared with assistance from Ascent Environmental, Inc., Sacramento, California. <https://www.wildlife.ca.gov/SWAP>.

Campbell, K. and Donlan, C.J. (2005). 'Feral goat eradications on islands'. *Conservation Biology* 19(5): 1362–1374.

Campbell, K.J., Harper, G., Hansen, C.C., Algar, D.S., Keitt, B.S. and Robinson, S.A. (2011). 'Review of feral cat eradications on islands'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management.* pp. 37–46. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Capizzi, D., Baccetti, N. and Sposimo, P. (2016). 'Fifteen Years of Rat Eradication on Italian Islands'. In: F.M. Angelici (ed.) *Problematic Wildlife: A Cross-Disciplinary Approach.* pp. 205–227. Cham: Springer International Publishing.

Carrion, V., Donlan, C.J., Campbell, K.J., Lavoie, C. and Cruz, F. (2011). 'Archipelago-wide island restoration in the Galápagos Islands: Reducing costs of invasive mammal eradication programs and reinvasion risk'. *PLOS ONE* 6(5): e18835.

Clout, M.N. and Russell, J.C. (2006). 'The Eradication of Mammals from New Zealand Islands '. In: F. Koike, M.N. Clout, M. Kawakichi, M. De Poorter and K. Iwatsuki (eds.) *Assessment and Control of Biological Invasion Risks*, pp. 127–141. Kyoto, Japan and Gland, Switzerland: Shoukadoh Book Sellers and IUCN.

Convention on Biological Diversity (2011). *Conference of the Parties Decision X/2: Strategic Plan for Biodiversity 2011–2020*. <www.cbd.int/decision/cop/?id=12268>.

Depraetere, C. (2007). *Global Islands Database (GID): A Technical Note on a Global Dataset of Islands*. In collaboration with Arthur Dahl, February 2007, Global Islands Network.

Genovesi, P. (2005). 'Eradications of invasive alien species in Europe: a review'. *Biological Invasions* 7(1): 127–133.

Genovesi, P. and Carnevali, L. (2011). 'Invasive alien species on European islands: eradications and priorities for future work'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 56–62. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Glen, A., Atkinson, R., Campbell, K., Hagen, E., Holmes, N., Keitt, B., Parkes, J., Saunders, A., Sawyer, J. and Torres, H. (2013). 'Eradicating multiple invasive species on inhabited islands: the next big step in island restoration?'. *Biological Invasions* 15(12): 2589–2603.

Gregory, S.D., Henderson, W., Smee, E. and Cassey, P. (2014). *Eradications of Vertebrate Pests in Australia: A Review and Guidelines for Future Best Practice*. PestSmart Toolkit publication, Canberra, Australia: Invasive Animals Cooperative Research Centre.

Harper, G. and Carrion, V. (2011). 'Introduced rodents in the Galápagos: colonisation, removal and the future'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 63–66. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Harris, D.B., Gregory, S.D., Bull, L.S. and Courchamp, F. (2012). 'Island prioritization for invasive rodent eradications with an emphasis on reinvasion risk'. *Biological Invasions* 14(6): 1251–1263.

Hess, S.C. and Jacobi, J.D. (2011). 'The history of mammal eradications in Hawai'i and the United States Associated Islands of the Central Pacific'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 67–73. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Hoffmann, B.D., Luque, G.M., Bellard, C., Holmes, N.D. and Donlan, C.J. (2016). 'Improving invasive ant eradication as a conservation tool: A review'. *Biological Conservation* 198: 37–49.

Holmes, N.D., Griffiths, R., Pott, M., Alifano, A., Will, D., Wegmann, A.S. and Russell, J.C. (2015). 'Factors associated with rodent eradication failure'. *Biological Conservation* 185: 8–16.

Howald, G., Donlan, C.J., Galvan, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21(5): 1258–1268.

Jones, H.P., Holmes, N.D., Butchart, S.H.M., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P.E., Cuthbert, R.J., Ebbert, S., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R. and Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences* 113: 4033–4038.

Joppa, L.N., O'Connor, B., Visconti, P., Smith, C., Geldmann, J., Hoffmann, M., Watson, J.E.M., Butchart, S.H.M., Virah-Sawmy, M., Halpern, B.S., Ahmed, S.E., Balmford, A., Sutherland, W.J., Harfoot, M., Hilton-Taylor, C., Foden, W., Minin, E.D., Pagad, S., Genovesi, P., Hutton, J. and Burgess, N.D. (2016). 'Filling in biodiversity threat gaps'. *Science* 352(6284): 416–418.

Kapos, V., Manica, A., Aveling, R., Bubb, P., Carey, P., Entwistle, A., Hopkins, J., Mulliken, T., Safford, R., Stattersfield, A., Walpole, M.J. and Balmford, A. (2010). 'Defining and Measuring Success in Conservation'. In: N. Leader-Williams, W.M. Adams and R.J. Smith (eds.) *Trade-Offs in Conservation.* pp. 73–93. Oxford, UK: Wiley-Blackwell.

Keitt, B., Campbell, K., Saunders, A., Clout, M.N., Wang, Y., Heinz, R., Newton, K. and Tershy, B. (2011). 'The global islands invasive vertebrate eradication database: a tool to improve and facilitate restoration of island ecosystems'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 74–77. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Lorvelec, O. and Pascal, M. (2005). 'French attempts to eradicate non-indigenous mammals and their consequences for native biota'. *Biological Invasions* 7(1): 135–140.

MacKay, J.W.B., Russell, J.C. and Murphy, E.C. (2007). 'Eradicating House Mice from Islands: Successes, Failures and the Way Forward'. In: G.W. Witmer, W.C. Pitt and K.A. Fagerstone (eds.) *Managing Vertebrate Invasive Species. Proceedings of an International Symposium,* pp. 294–304. Fort Collins, Colorado: USDA/APHIS/WS, National Wildlife Research Center.

McChesney, G.J. and Tershy, B.R. (1998). 'History and status of introduced mammals and impacts to breeding seabirds on the California Channel and Northwestern Baja California Islands'. *Colonial Waterbirds* 21: 335–347.

McGeoch, M.A., Butchart, S.H.M., Spear, D., Marais, E., Kleynhans, E.J., Symes, A., Chanson, J. and Hoffmann, M. (2010). 'Global indicators of biological invasion: species numbers, biodiversity impact and policy responses'. *Diversity and Distributions* 16(1): 95–108.

Niemeijer, D. and de Groot, R.S. (2008). 'A conceptual framework for selecting environmental indicator sets'. *Ecological Indicators* 8(1): 14–25.

Nogales, M., Martin, A., Tershy, B.R., Donlan, C.J., Veitch, D., Puerta, N., Wood, B. and Alonso, J. (2004). 'A review of feral cat eradication on islands'. *Conservation Biology* 18(2): 310–319.

Oppel, S., Beaven, B.M., Bolton, M., Vickery, J. and Bodey, T.W. (2011). 'Eradication of invasive mammals on islands inhabited by humans and domestic animals'. *Conservation Biology* 25(2): 232–240.

Phillips, R.B., Wiedenfeld, D.A. and Snell, H.L. (2012). 'Current status of alien vertebrates in the Galápagos Islands: invasion history, distribution, and potential impacts'. *Biological Invasions* 14(2): 461–480.

Russell, J.C., Miller, S.D., Harper, G.A., MacInnes, H.E., Wylie, M.J. and Fewster, R.M. (2010). 'Survivors or reinvaders? Using genetic assignment to identify invasive pests following eradication'. *Biological Invasions* 12(6): 1747–1757.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: Rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Russell, J.C., Cole, N.C., Zuël, N. and Rocamora, G. (2016). 'Introduced mammals on Western Indian Ocean islands'. *Global Ecology and Conservation* 6: 132–144.

Russell, J.C., Meyer, J.-Y., Holmes, N.D. and Pagad, S. (2017). 'Invasive alien species on islands: impacts, distribution, interactions and management'. *Environmental Conservation* 44(4): 359–370.

Tobin, P.C., Kean, J.M., Suckling, D.M., McCullough, D.G., Herms, D.A. and Stringer, L.D. (2014). 'Determinants of successful arthropod eradication programs'. *Biological Invasions* 16(2): 401–414.

Veitch, C.R., Clout, M.N. (eds.) (2002). *Turning the tide: the eradication of invasive species.* Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Veitch, C.R., Clout, M.N. and Towns, D.R. (2011). (eds.) *Island invasives: eradication and management.* Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Witmer, G.W. and Fuller, P.L. (2011). 'Vertebrate species introductions in the United States and its territories'. *Current Zoology* 57(5): 559–567.

Going to scale: reviewing where we've been and where we need to go in invasive vertebrate eradications

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Abstract We are on the edge of the sixth mass extinction on Earth. Islands represent ca. 5% of the earth's land area yet are home to 61% of extinctions in the past 500 years, and currently support 39% of critically endangered species. Invasive species are a leading cause of extinction and endangerment on islands. Invasive vertebrates, particularly mammals, are among some of the most damaging invasive species on islands. Eradicating invasive mammals is an increasingly utilised conservation tool. Nevertheless, conservation intervention needs greatly outstrip the island restoration community's capacity. There are thousands of islands where invasive vertebrates are driving species toward extinction. So, how can the effort be matched to the scale of the problem? One approach is to improve outreach and communications to increase the resources available for projects. There are great stories; but these need to be told compellingly and repeatedly. Increasing social acceptance and support for invasive species eradications will reduce project costs associated with stakeholder engagement. Broadening the funding base can be accomplished by building stronger cost benefit valuations as well as engaging funders of climate change, marine conservation, human wellbeing, and food security. Furthermore, it is important to build upon existing partnerships to create or grow coalitions that can access these resources as part of broader, holistic efforts to address multiple conservation issues.

Keywords: communications, eradication, funding, invasive species, stakeholder engagement

INTRODUCTION

Multiple lines of evidence demonstrate that we are facing a significant global extinction crisis through the loss of biodiversity (Dirzo, 2003; Barnosky, et al., 2011). At a global scale, the response to this crisis includes the 2011–2020 strategic plan for biodiversity, highlighted in 20 targets to reduce pressures on the environment and to curb biodiversity loss (CBD, 2011). Islands are a logical place to focus conservation efforts because they offer a disproportionately higher rate of biodiversity and threatened species per unit area. Islands represent only ca. 5% of the earth's land area yet support ca. 39% of critically endangered species on the IUCN Red List (Tershy, et al., 2015), and an endemic richness of plants and vertebrates that is 8–9 times that on mainlands (Kier, et al., 2009).

Invasive alien species have been implicated as a leading cause of extinctions and endangerment for native plants and animals on islands (Tershy, et al., 2015). In particular, invasive mammals pose a significant risk (Doherty, et al., 2016). The development of tools and techniques to completely remove invasive mammal populations from islands has been a valuable intervention strategy for island managers to overcome this threat (Veitch & Clout, 2002; Veitch, et al., 2011). To date there have been more than 1,200 vertebrate eradication attempts on more than 700 islands with an 85% success rate (DIISE, 2014), and the pace and scale of eradications on islands is increasing (Simberloff, et al., 2018). Following successful eradications, demonstrable biodiversity conservation gains have accrued. A recent literature review found 596 populations of 236 native insular species benefited from 251 invasive mammal eradications on 181 islands (Jones, et al., 2016). Benefits included resident population recovery, recolonisation and unassisted colonisation, plus the enabling of reintroductions and conservation introductions. Similarly, Brooke, et al. (2017) investigated population growth rates in seabirds following invasive mammal eradications on islands and found a median population growth rate of 1.119 based on 181 populations of 69 seabird species.

NOTABLE ADVANCES AND INNOVATIONS

Several key innovations were critical to increasing the rate at which eradications of invasive mammals on islands have occurred. For rodents, New Zealand based programmes that researched the effectiveness of bait station approaches led to a series of successful implementations on small islands (Howald, et al., 2007). The advancement of aerial application techniques, including the use of satellite navigation systems, enabled efforts on larger islands and increased the number of islands treated, including >11,000 ha Campbell Island (Towns & Broome, 2003). These techniques have been exported internationally, with Macquarie Island at >12,000 ha recently declared successful, and implementation units recently treated within the South Georgia eradication reaching almost 30,000 ha. Likewise, for invasive ungulates, the advent of aerial hunting, extensive near real-time data management combined with mapping technology to coordinate large teams and different eradication methods, and the use of Judas goats enabled similar increases in number and size of islands treated (Campbell & Donlan, 2005) whereas aerial application, toxicant development and remote trap monitoring allowed continued increases in island size, efficiency and efficacy to be obtained on cat eradications (Campbell, et al., 2011), including the currently on-going treatment of ca. 65,000 ha Dirk Harthog Island in Australia.

The cumulative impact of numerous existing and on-going innovations is expected to increase the scope and scale of eradications on islands. Models to confirm eradication success (Ramsey, et al., 2009; Ramsey, et al., 2011) provide significant opportunities to reduce costs, particularly for large projects using hunting and/or trapping techniques, by increasing the efficiency of determining when a project is complete. Increased use of these tools, and associated real time, digital data collection and analysis tools, is recommended to increase efficacy, reduce costs and provide more information to enable post project review and analysis for future improvements (Will, et al., 2015).

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Efforts are ongoing to reduce reliance on second generation anti-coagulants for rodent eradications, whose efficacy comes with a trade-off of greater risk to nontarget species (Howald, et al., 2007). These include expanding the use of first generation anticoagulants that pose less risk to non-target species (Poncet, et al., 2011), investigating alternative compounds, such as Norbamide, and investigating new bait recipes that could increase efficacy, such as crab deterrents (Campbell, et al., 2015). Self-resetting traps, a relatively new tool, that have been deployed successfully for eradication on small islands in Puerto Rico and New Zealand, present another alternative on small islands where rodenticide use may not be possible (Carter, et al., 2016). These self-resetting traps present significant potential for biosecurity management and can provide long term protection where reinvasion risk from swimming rodents is high.

New strategies have been developed to overcome the higher failure rate in rodent eradications in the tropics. After a series of high profile rodent eradication failures on tropical islands, a workshop of practitioners, The Tropical Rodent Eradication Review, was convened to evaluate reasons for these failures and develop recommendations to increase success rates in the future. These guidelines were published in 2015 (Keitt, et al., 2015) and several projects implemented since have followed the spirit of these guidelines. It remains to be seen whether efficacy rates will increase as a result, though the second attempt on Desecheo, which followed the guidelines, was declared successful (Will, et al., 2019).

Another promising approach is genetic tools that can lead to eradication of rodent populations (Campbell, et al., 2015, Campbell et al., 2019). Genetically modifying rodents to produce sterile offspring or only males and using gene drives to push for near 100% inheritance of this trait, could lead to eradication at large scales, including on inhabited islands where eradication is not currently feasible. This technology is in the early stages of development for house mice and it is unlikely that it would be available for field trials sooner than a decade from now; longer for commensal rat species. However, there has been significant concern raised about the safety and ethics of pursuing this line of conservation, particularly around the potential for a gene drive to run through an entire species and lead to extinction (National Academies of Sciences, Engineering, and Medicine, 2016). If this technology can be proven safe and gain the appropriate social and political approvals, it could have wide ranging impact on the conservation of large inhabited islands while also providing significant benefit to humans through reduced disease transmission and reduction in agricultural loss.

LOOKING TO THE FUTURE

These efforts have made significant contributions to global progress in protecting biodiversity. However, there remains much to be done, and the conservation need is high. Jones, et al (2016) predicted that 107 highly threatened insular terrestrial vertebrates (229 populations) have benefitted in some way from invasive mammal eradications on islands, however this represents just 12% of all 860 highly threatened terrestrial vertebrates occurring on islands. The picture is brighter for seabirds, where 47% of critically endangered and 74% of endangered species were predicted to have benefitted from invasive mammal eradications to date. Considering the future, McCreless, et al. (2016) found that efforts to control or eradicate relevant invasive species could prevent 41–75% of future predicted extirpations of populations of threatened vertebrate species. Almost half of these extirpations reflect species with a single population (endemic) and thus extirpation is the same as extinction.

The number of islands targeted for eradication are few compared to the number of islands worldwide. Invasive rodents are widespread, with estimates of 80% of the world's island groups being invaded (Atkinson, 1985). Recent estimates suggest there are > 400,000 islands in the world > 10 ha (UNEP-WCMC, 2013) yet only ca. 450 have been the focus of rodent eradications (DIISE, 2014). Thus, the need to increase the scope and scale of efforts to eradicate invasive vertebrates is known (Philips, 2010). A considerable number of these invaded islands are outside the boundaries of what is considered feasible for invasive species eradications today, and innovative approaches will need to be established to realise these opportunities (Campbell, et al., 2015). These include use of some of the innovations mentioned above as well as ones yet to be envisioned. Two additional focal areas for development include the social acceptability of these projects and increased funding to implement projects.

CONSERVING SPECIES ON INHABITED ISLANDS – UNDERSTANDING THE SOCIAL CONTEXT

Due to the overlap of human settlements and biodiversity there has been an increasing interest in eradication projects on inhabited islands (Oppel, et al., 2011). Simberloff, et al. (2018) reported 194 eradication attempts on 94 inhabited islands, and a "sharp uptick" in numbers of attempts on inhabited islands for all species except rodents in 1960 and for rodents in 1990. Notable projects under consideration include Lord Howe Island, Robinson Crusoe, Great Barrier Island and Floreana Island. Glen, et al. (2013) make the case that inhabited islands often support a suite of invasive species and thus restoration efforts can require multi-species eradications that must take into account the ecological impacts of improper sequencing of removals and potential negative consequences of allowing some invaders to remain. Combining this challenge with that of gaining social license to achieve eradication, inhabited islands have been hailed as a next great challenge for conservation (Glen, et al., 2013).

It is likely that most land managers attempting to implement invasive vertebrate eradications on islands would prefer to do so in the relatively accommodating social environment of New Zealand, where signs in tourism shops proudly report on their efforts to control invasive species. Understanding the underlying reasons for social acceptance, or lack thereof, for eradication projects, is an important aspect of planning an appropriate process to achieve stakeholder support and approval for a project. As an example, the New Zealand conservation movement arguably began with efforts to protect its endemic birds, including the national bird, the kiwi (genus *Apteryx*) (Stoltzenberg, 2011). Given that invasive species currently are their greatest threat, it is natural that control and eradication enjoy broad support within the country. Contrast this with the United States, where some suggest the environmental movement can be traced back to Rachel Carson's Silent Spring (1962), which highlighted the imminent extinction of the US national symbol, the bald eagle, (*Haliaeetus leucocephalus*) from pesticide exposure. It is exactly these kinds of underlying human conditions that can impact attitudes about invasive species and the tools to control and eradicate them. Island restoration projects have typically applied significant rigor to the biological science necessary to understand invasive mammal eradication projects. As projects face more complex human dimensions, it will be necessary to apply the same rigor to the social sciences in order to achieve the necessary project support to proceed.

INCREASING FINANCIAL AND STAKEHOLDER SUPPORT

To expand funding opportunities for island restoration projects it is important to expand project justifications beyond biodiversity conservation to include human health and livelihoods, and ecosystem services. This will require new research to document and communicate the non-biodiversity impacts of these projects. For example, the Lord Howe Island rat eradication project underwent a comprehensive Cost Benefit Analysis that demonstrated there would be a benefit cost ratio of 17.0, i.e. 17 dollars in benefits for every dollar spent on the project (Gillespie, 2016). A similar approach was completed for the Cabritos Island donkey eradication in the Dominican Republic (Rijo, 2014). This analysis showed a benefit cost ratio of between 2.0 and 4.2 depending on the methods used to remove all of the donkeys and resulting cost of the work. Additional efforts to highlight the value of vertebrate eradications on islands to humans, including human health (de Wit, et al., 2017), ecosystem services (Peh, et al., 2014), and agriculture will be key to securing the necessary support, both financial and stakeholder, to meet the challenge.

Making a strong link between island restoration and marine conservation is important for maximising available resources. Islands serve an important function in marine ecosystems (Gove, et al., 2016), including providing key breeding habitat for species that are dependent on marine resources. Most seabirds, sea turtles and marine mammals are dependent on islands to reproduce yet are key members of marine ecosystems. Making this case to marine funders and incorporating goals to protect and maintain populations of top level native predators in the management plans of these reserves is a good place to start.

Climate change is projected to have a significant impact on islands and island species and there are significant global financial resources available for addressing climate change impacts. Tershy, et al. (2015) argue that some of the same attributes that make island species vulnerable to invasive species, primarily smaller ranges and population sizes and less genetic diversity, also make them vulnerable to climate change. For many island ecosystems, invasive mammal eradications, in combination with other restoration actions, can increase resilience to projected climate change impacts, and provide refugia for species whose habitat is projected to be lost. However, proposed island restorations on low elevation islands should consider future sea level rise projections (Courchamp, et al., 2014) and include this in the project cost/benefit analysis.

Partnerships are not new to conservation, yet as island restoration projects expand in size and scope, diverse partnerships become more important to their success. Non-governmental organisations and governments working collaboratively together are becoming more commonplace. For example, the United States Fish and Wildlife Service has established a national level Memorandum of Understanding with other government agencies and US based NGOs to facilitate invasive species work and move from a project focus to a more programmatic one. Collaboration between NGOs internationally is also becoming more commonplace in the implementation of eradication projects. An example is the partnership between Island Conservation and Birdlife International on the multi-island, multi-species eradication in the Acteon and Gambier archipelagos, a project led by the local Tahitian NGO, and Birdlife Partner, SOP Manu. There are opportunities to expand these types of governmental and non-governmental partnerships, to enhance the capacity for conservation actions worldwide. Much is written about how to create successful partnerships, and common tenets to these types of partnerships are working to clarify shared values, programme goals, respective responsibilities and definitions of success.

Perhaps one of the greatest opportunities to grow support for island conservation work in this increasingly connected and wired planet is through communication and outreach. Effective communication requires sharing the right information with the right audience at the right time. Story-telling is a communication approach that can make difficult to understand ideas, such as the need to kill non-native species to conserve native ones, more accessible. Having island residents and stakeholders tell their stories or presenting a project from the viewpoint of the native species that will benefit, can resonate far more than statistics and summaries of what has happened somewhere else. The Goodman Center (<www.thegoodmancenter.com>) is a resource that can help train how to develop and tell compelling stories. For island restoration, the audiences are varied – funders, stakeholders, island communities and practitioners. This requires creating story arcs that reflect the values of key decision-makers and involve rigorous and defensible research to create story content. The recovery associated with removing invasive species from islands is often exceptional, providing compelling and dramatic messages that can be shared to generate interest in projects. Investing in the monitoring to document these stories is often under-valued, yet the link to funding future projects is clear. The platform for telling stories and reaching some audiences is evolving quickly alongside technology, thus these social media platforms require constant innovation and novel approaches to reach audiences. Conversely, many island communities communicate the same way they did decades or even a century ago, with shared experiences and face-to-face time as the key medium. Effective and thoughtful planning of communications will continue to evolve as necessary components of island restoration.

CONCLUSION

There are few conservation approaches that can match the return on investment of invasive mammal eradications on islands. As the earth continues to lose biodiversity at a rapid pace, with islands disproportionately affected, it is urgent to increase the rate at which islands are restored. Innovation has played a key role in past increases in eradication efficacy and efficiency (Keitt, et al., 2011) and new innovations are primed to do the same (Campbell, et al., 2015). However, these innovations must expand beyond the technical aspect of how to eradicate invasives and include ways to increase funding and stakeholder engagement and support. With greater buy in for island restoration projects they will become easier to implement.

REFERENCES

Atkinson, I.A.E. (1985). 'The Spread of Commensal Species of *Rattus* to Oceanic Islands and their Effects on Island Avifaunas'. In: P.J. Moors (ed.) *Conservation of Island Birds*, pp. 35–81. ICBP Technical Publication No. 3. Cambridge, UK: ICBP.

Barnosky, A.D., Matzke, N., Tomiya, S., Wogan, G.O.U., Swartz, B., Quental, T.B., Marshall, C., McGuire, J.L., Lindsey, E.L., Maguire, K.C., Mersey, B. and Ferrer, E.A. (2011). 'Has the Earth's sixth mass extinction already arrived?' *Nature* 471: 51–57.

Brooke, M.de L., Bonnaud, E., Dilley, B.J., Flint, B., Holmes, N.D., Jones, H.P., Provost, P., Rocamora, G., Ryan, P.G., Surman, C. and Buxton, R.T. (2017). 'Seabird population changes following mammal eradications on islands'. *Animal Conservation* 10.1111/acv.12344.

Campbell, C. and Donlan, C.J. (2005). 'Feral goat eradications on islands'. *Conservation Biology* 19: 1362–1374.

Campbell, K.J.; Harper, G.; Algar, D.; Hanson, C.; Keitt, B.S. and Robinson, S. 2011. 'Review of feral cat eradications'. In: Veitch, C.R.; Clout, M.N. and Towns, D.R. (eds.) *Island invasives: eradication and management*, pp. 37–46. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Campbell, K.J., Beek, J., Eason, C.T., Glen, A.S., Godwin, J., Gould, F., Holmes, N.D., Howald, G.R., Madden, F.M., Ponder, J.P., Threadgill D.W., Wegmann, A. and Baxter, G.S. (2015). 'The next generation of rodent eradications: Innovative technologies and tools to improve species specificity and increase their feasibility on islands'. *Biological Conservation* 185: 47–58.

Campbell, K.J., Saah, R.J., Brown, P.R., Godwin, J., Gould, F., Howald, G.R., Piaggio, A., Thomas, P., Tompkins, D.M., Threadgill, D., Delborne, J., Kanavy, D.M., Kuikin, T., Leitschuh, C., Packard, H., Serr, M. and Shiels, A. (2019). 'A potential new tool for the toolbox: assessing gene drives for eradicating invasive rodent populations'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 6–14. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Carson, R. (1962). *Silent Spring*. New York: Houghton Mifflin Company.

Carter, A., Barr, S., Bond, C., Paske, G., Peters, D. and van Dam, R. (2016). 'Controlling sympatric pest mammal populations in New Zealand with self-resetting, toxicant-free traps: A promising tool for invasive species management'. *Biological Invasions* 18: 1723–1736.

CBD. (Convention on Biological Diversity) (2011). 'Conference of the Parties Decision X/2: Strategic Plan for Biodiversity 2011–2020'. <www.cbd.int/decision/cop/?id=12268>.

Courchamp, F., Hoffmann, B.D., Russell, J.C., Leclerc, C. and Bellard, C. (2014). 'Climate change, sea-level rise, and conservation: Keeping island biodiversity afloat'. *Trends in Ecology and Evolution* 29: 127–130.

de Wit, L.A., Croll, D.A., Tershy, B., Newton, K M., Spatz D.R., Holmes N.D. and Kilpatrick, A.M. (2017). 'Estimating burdens of neglected tropical zoonotic diseases on islands with introduced mammals'. *The American Journal of Tropical Medicine and Hygiene* 96: 749–757.

Doherty, T.S., Glen, A.S., Nimmo, D.G., Ritchie, E.G. and Dickman, C.R. (2016). 'Invasive predators and global biodiversity loss'. *Proceedings of the National Academy of Sciences* 113: 11261–11265.

DIISE (2014). 'The Database of Island Invasive Species Eradications, Developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand'. <http://diise.islandconservation.org>. Accessed 14 September 2014.

Dirzo, R. and Raven, P.H. (2003). 'Global state of biodiversity and loss'. *Annual Review of Environment and Resources* 28: 137–167.

Gove, J.M., McManus, M.A., Neuheimer, A.B., Polovina, J.P., Drazen , J.C., Smith, C.R., Merrifield, M.A., Friedlander, A.M., Ehses, J.S., Young, C.W., Dillon, A.K. and Williams, G.J. (2016). 'Near-island biological hotspots in barren ocean basins'. *Nature Communications* 7: 10581.

Glen, A.S., Atkinson, R., Campbell, K.J., Hagen, E., Holmes, N.D., Keitt, B.S., Parkes, J.P., Saunders, A., Sawyer, J. and Torres, H. (2013). 'Eradicating multiple invasive species on inhabited islands: The next big step in island restoration'. *Biological Invasions* 15: 2589–2603.

Gillespie, R. (2016). 'Economic Evaluation of the Lord Howe Island Rodent Eradication Project'. Gillespie Economics, unpublished. Report to the Lord Howe Island Board.

Howald, G., Donlan, C.J. Galvan, J.P. Russell, J.C. Parkes, J. Samaniego, A. Wang Y., Veitch, D. Genovesi, P. Pascal, M. Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Jones, H.P., Holmes, N.D., Butchart, S.H., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P.E., Cuthbert, R.J.1., Ebbert, S., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R. and Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences* 113: 4033–4038.

Keitt, B., Campbell, K., Saunders, A., Clout, M., Wang, Y., Heinz, R., Newton, K. and Tershy, B. (2011). 'The global islands invasive vertebrate eradication database: a tool to improve and facilitate restoration of island ecosystems'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 74–77. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millet, J., Pitt, W. and Samaniego-Herrera, A. (2015) 'Best practice guidelines for rat eradications on tropical islands'. *Biological Conservation* 185: 17–26.

Kier, G., Kreft, H., Lee, T.M., Jetz, W., Ibisch, P.L., Nowicki, C., Mutke, J. and Barthlott, W. (2009). 'A global assessment of endemism and species richness across island and mainland regions'. *Proceedings of the National Academy of Sciences* 106: 9322–9327.

McCreless, E.E., Huff, D.D., Croll, D.A., Tershy, B.R., Spatz, D.R., Holmes, N.D., Butchart, S.H. M. and Wilcox, C. (2016). 'Past and estimated future impact of invasive alien mammals on insular threatened vertebrate populations'. *Nature Communications* 7: 12488.

National Academies of Sciences, Engineering, and Medicine (2016). *Gene Drives on the Horizon: Advancing Science, Navigating Uncertainty, and Aligning Research with Public Values*. Washington, DC: The National Academies Press. <https://doi.org/10.17226/23405>.

Oppel, S., Beaven, B.M., Bolton, M., Vickery, J. and Bodey, T.W. (2011). 'Eradication of invasive mammals on islands inhabited by humans and domestic animals'. *Conservation Biology* 25: 232–240.

Peh, K.H., Balmford, A., Birch, J., Brown, C., Butchart, S.M., Daley, J., Dawson, J., Gray, G., Hughes, F.M.R., Mendes, S., Millett, J., Stattersfield, A.J., Thomas, D.H.L., Walpole, M. and Bradbury, R. (2015). 'Potential impact of invasive alien species on ecosystem services provided by a tropical forested ecosystem: A case study from Montserrat'. *Biological Invasions* 17: 461–475.

Phillips, R.A. (2010). 'Eradications of invasive mammals from islands: Why, where, how and what next?' *Emu* 110: 1–7.

Poncet, S., Poncet, L., Poncet, D., Christie, D., Dockrill, C. and Brown, D. (2011). 'Introduced mammal eradications in the Falkland Islands and South Georgia'. In: C.R. Veitch, M.N. Clout and D. Towns (eds.) *Island invasives: eradication and management*, pp. 332–336. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Ramsey, D.S.L., Parkes, J.P. and Morrison, S.A. (2009). 'Quantifying eradication success: The removal of feral pigs from Santa Cruz Island, California'. *Conservation Biology* 23: 449–459.

Ramsey, D.S.L., Parkes, J.P., Will, D.J., Hanson, C.C. and Campbell, K.J. (2011). 'Quantifying the success of feral cat eradication, San Nicolas Island, California'. *New Zealand Journal of Ecology* 35: 163–173.

Rijo, C. (2014). 'Benefits to Biodiversity Outweigh Cost of Vertebrate Eradication on Cabritos Island: Costs Benefits Analysis of Donkeys Eradication'. In: A. Daigneault, P. Brown and N. Ramnanan (eds.) *Economic Impact of IAS in The Caribbean: Case Studies*, pp. 30–37. Port of Spain, Trinidad and Tobago: CABI.

Simberloff, D., Keitt, B., Will, D., Holmes, N., Pickett, Genovesi, P. (2018). 'Yes we can! Exciting progress and prospects for controlling invasives on islands and beyond'. *Western North American Naturalist* 78: 942-958.

Stoltzenberg, W. (2011). *Rat Island*. New York: Bloomsbury USA.

Tershy, B.R., Shen, K.W., Newton, K.M., Holmes, N.D. and Croll D.A. (2015). 'The importance of islands for the protection of biological and linguistic diversity'. *BioScience* 65: 592–597.

Towns, D.R. and Broome, KG. (2003) 'From small Maria to massive Campbell: Forty years of rat eradications from New Zealand islands'. *New Zealand Journal of Zoology* 30: 377–398.

[UNEP-WCMC] 'United Nations Environment Programme World Conservation Monitoring Centre (2013). Global Islands Database *UNEP-WCMC* 30 January, 2013'. <www.unep-wcmc.org>.

Veitch, C.R. and Clout, M.N. (2002). 'Turning the tide of biological invasion: the potential for eradicating invasive species' In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species*, pp. 1–3. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Veitch, C.R., Clout, M.N. and Towns, D.R. (2011). 'Invasive alien species: losses, gains, and engaging the public'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, p. XI. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Will, D.J., Campbell, K.C. and Holmes, N.D. (2015). 'Using digital data collection tools to improve overall cost-efficiency and provide timely analysis for decision making during invasive species eradication campaigns'. *Wildlife Research* 41: 499–509.

Will, D.J., Swinnerton, K., Silander, S., Keitt, B., Griffiths, R., Howald, G.R., Figuerola-Hernandez, C.E., and Herrera-Giraldo, J.L. (2019). 'Applying lessons learnt from tropical rodent eradications: a second attempt to remove invasive rats from Desecheo National Wildlife Refuge, Puerto Rico'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 154–161. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Tackling invasive non-native species in the UK Overseas Territories

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Abstract The 16 UK Overseas Territories (OTs) together account for 94% of the UK's unique biodiversity and make a significant contribution to global biodiversity. Being predominantly islands, the OTs are very vulnerable to the introduction of potentially harmful invasive non-native species, and pressures are increasing with the continual growth of international trade and impact of climate change. Biosecurity is acknowledged as the most cost-effective means of addressing invasive species threats for small islands, and yet the OTs face many challenges in the implementation of biosecurity controls. In 2016 a UK Government funded project "Tackling Invasive Non-Native Species in the UK Overseas Territories" was initiated to improve the biosecurity of the OTs against non-native species in order to improve their environmental resilience and food security through technical assistance and capacity building. A gap analysis carried out in early 2017 assessed the strengths and weaknesses for all 16 OTs along the biosecurity continuum in three areas: (1) prevention (2) early warning and rapid response, and (3) long-term management. Overall, capacity is weakest in the area of prevention and greatest in that of long-term management. Border activities, where implemented, are primarily linked to agricultural production and animal health. Few OTs have carried out horizon scanning or comprehensive pathway analysis or have the capacity to carry out pest risk analysis. Greatest capacity is seen in the relatively well resourced Antarctic and sub-Antarctic territories, and in St Helena Island which was the subject of a 4-year project in anticipation of air access. Legislation is generally weak, and few OTs have developed territorial biosecurity policies or strategies. Officers responsible for biosecurity often have a range of functions in addition to their biosecurity roles, lack access to specialist expertise and diagnostic facilities, and may also lack access to appropriate training. This compromises their ability to deliver effective biosecurity. This situation is common to many small island states.

Keywords: biosecurity, capacity, gap analysis, horizon-scanning, pathway-analysis, prevention

INTRODUCTION

The 16 UK Overseas Territories (OTs) together account for 94% of the UK's unique biodiversity and as such make a significant contribution to global biodiversity (Churchyard, et al., 2014). Despite this, involvement of the UK government in the OTs with regards provision of financial and other resources is minimal, with the OTs receiving only project funding from the UK (e.g. Vaas, et al., 2017). Being predominantly islands, the OTs are very vulnerable to the introduction of potentially harmful invasive non-native species, recognised as the biggest threat to island biodiversity, as well as to food security and sustainable development (Copsey, et al., 2018). Pressures are increasing with the continual growth of international trade, the main driver of the spread of invasive species, resulting in higher numbers of individuals of more species being moved around the world, both deliberately and accidentally. The chances of a new potentially harmful species arriving and establishing in a new area are therefore greater. The implementation of biosecurity measures is aimed at minimising this risk (Copsey, et al, 2018), and contributes towards achievement of Strategic Goal B of the Convention on Biological Diversity, *Reduce the direct pressures on biodiversity and promote sustainable use*, and specifically Aichi Target 9 (UNEP, 2011).

Biosecurity, defined as measures to reduce the risk of introducing or spreading invasive non-native species (and other harmful organisms such as diseases) in the wild, has long been acknowledged as the most cost-effective means of addressing invasive species threats for small islands (for example Tye, 2009). To be effective, actions need to be implemented across the biosecurity continuum, with pre-border controls at the country of origin, inspections and interceptions at the border, and post-border surveillance and interventions in the wider environment, all applied to both deliberate (legal and illegal) and accidental introductions. Once implemented, biosecurity actions must be maintained as part of normal government practice.

The IUCN announced the Honolulu Challenge at the World Conservation Congress 2016, calling for greater action to tackle the issue of invasive non-native species across the globe, with particular attention to preventative action and the development of effective biosecurity policies (IUCN, 2017).

As part of the UK Government's response the 3-year project *Tackling Invasive Non-Native Species in the UK Overseas Territories* was initiated. Its objective is "to improve the biosecurity of the OTs against invasive non-native species to improve their environmental resilience and food security; achieved through reducing the risk and impact of invasion and natural hazards via technical assistance and capacity building".

In order to plan the appropriate capacity building activities, a gap analysis was carried out in January 2017 on biosecurity practices and capacity in all 16 UK OTs (Fig. 1) (information from McPherson, 2016):

Anguilla: one main and a number of smaller islands in the Caribbean region with a total area of 90 km^2 and population of 13,572.

Ascension Island: a single main island in the South Atlantic, with an area of 87 km^2 and population of 1,000.

Bermuda: eight connected islands and over 190 smaller islands in the wider Caribbean with a total area of 53.7 km^2 and population of 65,038.

British Antarctic Territory (BAT): the Antarctic Peninsula and two groups of nearby islands, with a total area of 1,709,400 km^2 and no permanent population.

British Indian Ocean Territory (BIOT); archipelago of over 50 small low-lying islands in the Indian Ocean, with a total area of 50 km^2 and no permanent population, but a large permanent military presence.

In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) (2019). Island invasives: scaling up to meet the challenge, pp. 637–642. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

British Virgin Islands (BVI): Four main islands and over 50 small islets and cays in the Caribbean, with a total area of 151 km^2 and population of 28,882.

Cayman Islands: three islands in the Caribbean, with a total area of 264 km^2 and population of 54,397.

Cyprus Sovereign Base Areas (CSBA): two separate areas, Akrotiri-Episkopi (the Western SBA) and Dhekelia (the Eastern SBA), on the island of Cyprus in the Mediterranean, with a total area of 254 km^2 and population of 15,700.

Falkland Islands: two main islands and over 770 smaller islands in the South Atlantic, with a total area of 12,173 km^2 and population of 2,841.

Gibraltar: a peninsula at the southern coast of Spain, with an area of 6.8 km^2 and population of 31,465.

Montserrat: a single island in the Eastern Caribbean, with an area of 102 km^2 and population of 4,922.

Pitcairn Islands: four islands in the South Pacific, with a total area of 48.7 km^2 and population of 47, all resident on the main island.

St Helena Island: a single main island in the South Atlantic, with an area of 121 km^2 and population of 4,534.

South Georgia and South Sandwich Islands (SGSSI): one main island and several small ones in the South Georgia group and a group of 11 small islands in the South Sandwich Islands Group, all in the sub-Antarctic. Total area is 3,903 km^2 with no permanent population.

Tristan da Cunha: four islands in the South Atlantic, with a total area of 207 km^2 and population of 268.

Turks and Caicos Islands: two island groups of over 120 small islands in the Caribbean, with a total area of 417 km^2 and population of 49,000.

METHODS

A questionnaire was designed, identifying the components required for an effective biosecurity programme along the biosecurity continuum. Emphasis was given to the pre-border and post-border activities targeted by the project, grouped in three areas: 1) Prevention; 2) Early Warning and Rapid Response (EWRR); and (3) Management, Prioritisation and Frameworks (MPF) in the components defined as follows:

Prevention

Pest Risk Analysis (PRA): system established and in use to evaluate the likelihood of the entry, establishment, or spread of a pest or disease, and the associated potential biological and economic consequences. Both phytosanitary and zoosanitary risks covered.

Non-Native Species Risk Analysis (NNRA): comprehensive risk assessment frameworks exist to assess the risk of non-native species (plant and animal) becoming invasive.

Pathway Analysis: prioritised pathways of entry identified, and results used as the basis for procedures.

Horizon Scanning: horizon scanning exercise carried out to identify invasive species most likely to invade via identified pathways.

Contingency Planning: formalised generic contingency plan or plans in place to deal with priority invasive species that are likely to arrive. This is divided into (i) Plants, including both plants and plant health risks (non-native plant pests and diseases); (ii) Animals, including both vertebrates and animal health risks (non-native vertebrates, animal diseases and parasites); and (iii) Other risks (invertebrates other than plant pests, and marine species).

Border Operations: in-place and operational, considering staffing, provision of dedicated facilities, procedures and protocols in place, public awareness, and levels of compliance. Both phytosanitary and zoosanitary risks covered.

Early warning and rapid response

Alert System: clear system in place for reporting incursions or new species, for both plant and animal (vertebrate and invertebrate) risks.

Surveillance: generic and/or incursion specific programmes in place for surveillance of priority invasive

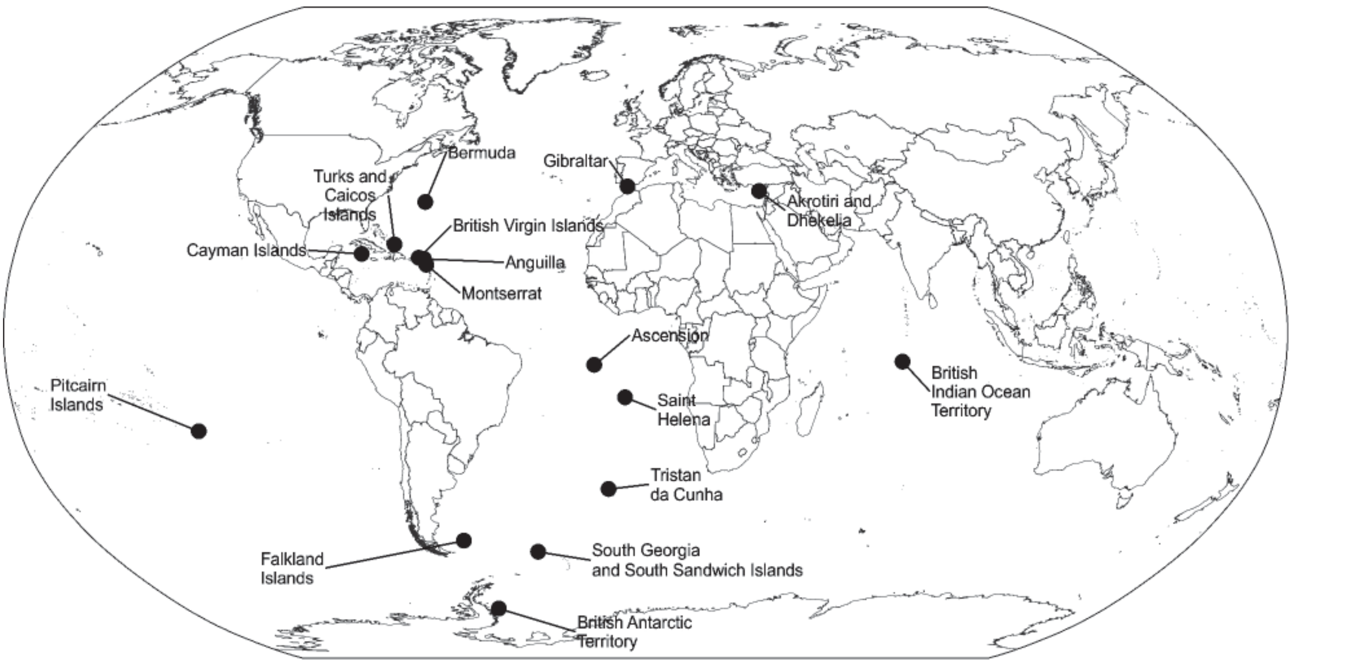

Fig. 1 The 16 UK Overseas Territories.

species. This is divided into (i) Plants, including both plants and plant health risks (non-native plant pests and diseases); (ii) Animals, including both vertebrates and animal health risks (non-native vertebrates, animal diseases and parasites); and (iii) Other risks (invertebrates other than plant pests, and marine species).

Monitoring: generic and/or incursion specific programmes in place for monitoring established priority invasive species.

Rapid Response Capacity: capacity (capability and resources) to provide rapid response to incursions. This is divided into (i) Plants, including both plants and plant health risks (non-native plant pests and diseases); (ii) Animals, including both vertebrates and animal health risks (non-native vertebrates, animal diseases and parasites); and (iii) Other risks (invertebrates other than plant pests, and marine species).

Management, prioritisation and frameworks

Prioritisation: prioritised established invasive species for control/eradication based on global risk management best practice, such as the *Guidelines for invasive species management in the Pacific* (Tye, 2009).

Baseline Data: baseline inventories available for plants (native and non-native), animals (terrestrial vertebrates and invertebrates), and other (marine species).

Territorial Framework: biosecurity legislation in place and enforced; biosecurity strategy or policy in place or endorsed, and being implemented.

Contacts were established in each OT for both agriculture and environment sectors, and territory capacity assessed though a combination of email, telephone interviews and face to face interviews. At least two people were involved in each territory, with the exception of BAT where there was only one. Capacity for each component was rated and scored as follows:

None - No action taken / Nothing in place. Score of 0

Basic - Some actions taken / Basic framework or actions in place / Actions planned in near future and expected to take place. Score of 1.

Some - Some substantial advances while other actions remain to be done / Actions being actively implemented along a planned timeframe. Score of 2.

Good - Substantive actions taken / Substantial framework or actions in place / Action being implemented / Action achieved. Score of 3.

Scores were summed across the components and territories to provide a simple index for comparison purposes. The text and ratings assigned to the components were in all cases agreed and approved by the contacts in-country for each territory. The resulting scores were then cross-checked by Dr Niall Moore of the GB Non-Native Species Secretariat to ensure that the ratings matched the comments; any adjustments were then discussed and agreed by the relevant contacts.

Final scores were checked by visitors from the RSPB, IUCN, Animal and Plant Health Agency (APHA), UK, and South Georgia Heritage Trust with recent experience of the relevant territory. Again, any discrepancies were then discussed and agreed by the relevant contacts before the scores were finalised.

RESULTS

Responses were obtained from all 16 OTs. Overall, respondents welcomed the project and expressed frustration where they identified gaps in their territory.

Table 1 Overall scores in the areas of Prevention, Early Warning and Rapid Response (EWRR) and Management, Prioritisation and Frameworks (MPF) and total scores for each of the 16 OTs and the UK in ascending order, out of a maximum overall score of 66. Maximum possible scores per area are 24 (Prevention and EWRR) and 18 (MPF). Overall mean score excludes that for the UK.

Territory	Prevention	EWRR	MPF	Overall score
Turks and Caicos	4	8	7	19
BIOT	3	5	12	20
CSBA	3	7	11	21
Montserrat	5	8	9	22
Ascension	5	8	10	23
Anguilla	8	4	12	24
Bermuda	5	9	12	26
Tristan da Cunha	7	7	12	26
Pitcairn	9	10	7	26
Falkland Islands	11	10	10	31
Cayman	11	9	13	33
BVI	10	14	10	34
Gibraltar	3	17	17	37
BAT	17	11	17	45
St Helena	14	18	13	45
SGSSI	14	19	18	51
UK	21	20	17	58
Overall mean score for the OTs	8.1	10.3	11.9	

Differences between territories

Scores for each territory in the three categories of Prevention, EWRR and MPF are shown in Table 1, with the territories listed from the lowest overall score (weakest capacity) to the highest (most capacity). The estimated score for the UK is given for comparison.

The three highest scoring territories are the two Antarctic and sub-Antarctic territories, and St Helena. BAT and SGSSI (with total scores of 45 and 51 respectively) benefit from their unique environmental status and considerable research input. St Helena (with a total score of 45) has been the subject of a 4-year project to strengthen biosecurity in anticipation of air access. The total score for SGSSI (51) is closest to that estimated for the UK (58).

A group of four territories have total scores between 31 and 36, comprising in ascending order: Falkland Islands, Cayman Islands, BVI and Gibraltar; Gibraltar has a score accounting for less than 20% of their overall score in the area of Prevention, but scores highly in the other areas.

A group of nine territories have the lowest totals, with scores between 19 and 26 and only one or two points between each, comprising in ascending order: Turks and Caicos Islands, BIOT, CSBA, Montserrat, Ascension Island, Anguilla, Bermuda, Tristan da Cunha and Pitcairn. Three territories are particularly weak in the area of Prevention, with scores accounting for less than 20% of their overall score: BIOT, CBSA and Bermuda. All four have ratings of Basic or None for all components in this area with only two exceptions: Bermuda with a rating of Some for border operations, and CBSA with a rating of Some for contingency planning for animals and animal health risks. Anguilla has a total score accounting for less than 20% of overall in EWRR, with all ratings in this area of Basic or None.

Components of biosecurity

The overall capacity is weakest in the area of Prevention, with an average score of 8.1, and strongest in the area of Management, Prioritisation and Frameworks, with an average score of 11.9 (Table 1). Table 2 shows total scores by component out of a maximum possible score of 48.

The highest scoring components are the group encompassing baseline inventories. This is generally good and especially for plants, with a total score of 43. Baseline knowledge for animals (terrestrial vertebrates and invertebrates) and other (marine species) both had a total score of 35.

The next highest scoring component is a group of four with scores of 28 to 30: alert system, prioritisation, legal framework and border operations.

The greatest capacity gaps are those of horizon scanning and contingency planning for other risks, both with total scores of 8. The second greatest gap is a group of three components: rapid response for other risks, surveillance of other risks and non-native risk analysis, all with scores of 12. Only five OTs have carried out horizon scanning, rated as Good only for BAT which has benefitted from considerable research input. The other OTs did not

Table 2 Total scores for each component: the maximum possible score is 48.

Component			Total score
Prevention			
	Risk Analysis (PRA)		16
	Risk Analysis (NNRA)		12
	Pathway Analysis		17
	Horizon scanning		8
	Contingency Planning	Plants and plant health risks	15
		Animals and animal health risks	23
		Other risks	8
	Border operations		30
Early Warning and Rapid Response			
	Alert System in Place		30
	Surveillance	Plants and plant health risks	23
		Animals and animal health risks	17
		Other risks	12
	Monitoring		23
	Rapid response Capacity	Plants and plant health risks	23
		Animals and animal health risks	24
		Other risks	12
Long-term management			
	Prioritisation		29
	Baseline	Plants	43
		Animals	35
		Other	35
	Framework	Legal	28
		Territorial policy or strategy	20

understand what horizon scanning was. "Other risks" comprises non-crop pest invertebrates and marine species, for which capacity is clearly weaker than for crop pests or plants; even for the UK where surveillance for other risks was the only component which was rated Basic, all the other components being rated as Some or Good.

DISCUSSION

The relatively small population size of the OTs means that biosecurity officers often have a range of functions and responsibilities in addition to their biosecurity roles, lack access to specialist expertise and diagnostic facilities, and may also lack access to appropriate training. This compromises their ability to deliver effective biosecurity. There is a dependence on community support, itself dependant on good levels of awareness and understanding. Officers carrying out biosecurity functions work closely with customs, and this is clearly an important partnership.

Biosecurity practices tend to be based on historic legislation inherited from their colonial pasts and not updated, with procedures aimed at protecting agriculture and production, focusing primarily on managing deliberate introductions to reduce the introduction of crop pests and livestock diseases, with a few exceptions (e.g. BAT and SGSSI). Legislation is weak and scattered across a number of regulations relating to customs, plant health and animal health. The broader threat posed by non-native invasive species to the environment is not being recognised, and extension of biosecurity approaches to species, which are not crop pests or livestock diseases, is generally poor or non-existent.

For many OTs, actions such as border operations and post-border surveillance are focused on easily-identifiable species such as Pacific lionfish *Pterois volitans*, brown tree snake *Boiga irregularis* and Tephritid fruit flies (Diptera: Tephritidae). While this is a good starting point for biosecurity teams, actions need to go further, and target more cryptic species identified as priority, as well as taking a generic approach to detect the unexpected. Biosecurity actions across the continuum are particularly weak for non-crop pest invertebrates, except where there has been a historic incident of note, such as the jacaranda bug (*Orthesia insignis*) outbreak on endangered endemic gumwood trees (*Commidendrum robustum*) in St Helena in the mid-1990s, which raised attention within the Territory to the issue of invasive non-native species.

BAT is distinct in being one of the few OTs which is not an island but one of 29 national Antarctic programmes. As such, BAT has no control over what is done on other stations, or what the tourism industry does with regard to biosecurity unless they come to BAT stations, rendering it vulnerable to intra-Antarctic transfer of non-native species. This issue is recognised as a concern in the Antarctic and included by the Antarctic Treaty Committee for Environmental Protection (CEP) in the 2016 CEP Non-native Species Manual (Anon., 2016a).

CSBA and Gibraltar are also not islands and consist of enclaves adjacent to EU countries (Spain and Cyprus). CSBA has relatively few resources dedicated to biosecurity, and with relatively long leaky land borders with the Republic of Cyprus this is to be expected. Gibraltar puts most attention into actions in the areas of Early Warning and Rapid Response, and Management, Prioritisation and Frameworks, with comprehensive monitoring programmes for existing invasive species, and surveillance programmes and rapid response capability in the event of an incursion. Actions are detailed in the Biodiversity Action Plan (Perez, 2006).

Where OTs have rated capacity as Basic or above in these components it is primarily due to the outcome of a specific research project, usually UK-funded by a competitive research grant such as a Darwin Plus award, or builds on a topical invasive species issue such as the Pacific lionfish (*Pterois volitans*) and pink hibiscus mealybug (*Maconellicoccus hirsutus*) invasions in the wider Caribbean (Morris & Whitfield, 2009; <http://www.cabi.org/isc/datasheet/40171>).

Risk Analysis (PRA and NNRA) comes quite low, with scores of 16 and 12 for PRA and NNRA respectively. Risk analysis, when done correctly, is a time-consuming and complex procedure which requires access to taxonomic and other expertise and, in most cases, funding to bring experts together. The small, resource-limited OTs are challenged to achieve this, and most carry out simplified forms of risk analysis as well as they can, on an ad-hoc basis, with heavy reliance on published databases such as the CABI Invasive Species Compendium and Global Invasive Species Database, and on assessments carried out for Florida, Hawaii and the Pacific Islands for plant species (<http://www.hear.org/pier/wra.htm>). While these make a good match for Pitcairn, their suitability to the other OTs is less certain. Comprehensive, published assessments specifically for the island groups in the Caribbean and South Atlantic would be very helpful.

The introduction of new exotic species as pets is of concern, particularly to the Caribbean territories, due to the risk of escapes or deliberate dumping of potentially invasive species in the wild. In the Caribbean, at least some introductions are linked to hurricanes: in Anguilla it is known that at least two monkeys escaped from an individual, who had them as pets, after a hurricane in 1999, and the green iguana was first introduced on logs of wood during a hurricane in 1995 (R. Connor, Government of Anguilla, pers. comm.). Escapes of exotic fish are not considered a big problem, probably due to the lack of large bodies of fresh water inland in the OTs. Escapes of exotic birds are also not considered a big issue. Currently, one of the commonest domestic species of concern is the cat (*Felis catus*) (R. Connor. pers. comm.). Unwanted kittens are frequently dumped in the wild and form feral populations, threatening wildlife such as the native Anguilla racer snake (*Alsophis rijgersmae*), endemic Antillean iguana (*Iguana delicatissima*), or endemic St Helena wirebird (*Charadrius sanctaehelenae*) (Varnham, 2006).

With the exception of CSBA, all the OTs carry out biosecurity border operations to a greater or lesser extent, and 12 out of the 16 rated this as "Some" or "Good". Focusing limited resources on border inspections and interceptions is cost-effective for islands where the border is clearly defined and defendable. However, in a continental context with leaky borders which cannot be readily defended, an alternative strategic approach is to identify the priority species or pathways of concern and work more widely across the biosecurity continuum, particularly post-border. Tactics adopted are based on the results of pathway analysis and horizon scanning. In this context, high scores across the board for all components aren't necessarily appropriate, instead a package of activities is adopted designed to minimise the identified risks. CSBA and Gibraltar are not island territories and have different priorities. In CSBA, the focus is on the zoosanitary risks of new animal disease outbreaks and public health issues, routine monitoring is of aerial insect vectors, specifically mosquitoes, and rapid response capacity exists to respond in the event of human or animal health outbreak. Gibraltar benefits from strong post-border monitoring, surveillance and prioritisation actions to protect its unique biodiversity, as laid out in the Gibraltar Biodiversity Action Plan and Reserve Management Plan (Anon., 2016b; Perez, 2006).

Ascension Island and BIOT also rated border operations as "Basic". Both territories have limited or no agricultural production and consequently little political incentive in the past to invest in biosecurity border controls. The limited resources available to biosecurity are targeted at post-border actions directed towards the highest risk species, namely mosquitoes of human health concern and fire ants in Ascension Island, and brown tree snake in BIOT. This approach emphasises the importance of horizon scanning, pathway analysis and accurate assessment of risks in the first place, and the need to build capacity in these areas to provide information on where to target resources.

PRIORITIES AND RECOMMENDATIONS

Aiming to build capacity for all OTs so that they have high scores across the board is neither realistic nor suitable. Whereas for many OTs an appropriate strategy would be to devote a substantial proportion of available resources to border operations, for others, such as CSBA or Ascension Island, a more cost-effective strategy instead would be to establish post-border surveillance programmes targeted at identified priority species or pathways. In all the OTs resources are limited, and officers must be very focused in their activities. To do this effectively, each OT needs basic information on the range of potential invasive species (horizon scanning), how they might arrive (pathway analysis) and how to assess risk (PRA and NNRA). Capacity in these fundamentals was found to be lowest in this gap analysis, and initial activities will concentrate in this area:

Building fundamentals:

- Horizon scanning linked with pathway analysis: to determine what potential invasive species are out there and the different ways they can arrive. The information is used to design an appropriate package of responses which guides how the available resources should be best divided up between preventative actions, such as pathway or border operations, and reactive actions, such as surveillance and rapid response.
- Risk analysis: the process of assessing biosecurity risks. OTs need access to support for risk analysis, and a harmonised approach across the OTs to guide practices on-island for:
- Assessment of plant or animal species for potentially invasive characteristics;
- Assessment of the risks of a plant or animal species carrying potentially harmful pests, parasites or diseases.

Establishing the framework:

- Territorial policy or strategy: agreed actions to achieve the appropriate package of response, including a communications strategy for awareness to improve compliance and internal advocacy to promote government support.
- Legislation: regulate across the biosecurity continuum, including actions to contain, control and eradicate established invasive species. Provision of model legislation would allow a harmonised approach across OTs; assistance with drafting to apply it at the territory level is also required.

Delivery:

- Training: on all aspects of biosecurity, with specific needs varying with the Territory. This provides essential underpinning to deliver the fundamentals and framework outlined above.

Adding value:

- Regional coordination: use regional coordination bodies where they exist and are active, linking among the UKOTs and also to appropriate independent countries and other territories.
- Build networks, either strengthening existing or developing new ones, to promote sharing and exchanges, and promote the confidence and inspiration which result from peer-learning networks.

Building capacity in the activities outlined above will equip the officers responsible for biosecurity in the OTs with the capacity to develop other actions such as contingency and rapid response planning.

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REFERENCES

Anon. (2016a). *Biosecurity Handbook, 4th Edition*. BAS Environment Office. <https://www.bas.ac.uk/wp-content/uploads/2016/12/BAS-Biosecurity-Handbook-2016-FINAL.pdf>.

Anon. (2016b). *Gibraltar Nature Reserve Management Plan 2016.* Consultative Draft. <https://www.gibraltar.gov.gi/new/sites/default/files/HMGoG_Documents/2016-GNR_Management_Plan_FINALv2.pdf>.

Churchyard, T., Eaton, M., Hall, J., Millett, J., Farr, A., Cuthbert, R. and Stringer, C. (2014). *The UK's Wildlife Overseas: A Stocktake of Nature in our Overseas Territories*. Sandy, UK: RSPB

Copsey, J.A., Black, S.A., Groombridge, J.G. and Jones, C.G. (2018). *Species Conservation: Lessons from Islands.* Cambridge: Cambridge University Press.

IUCN (2017). *International Union for Conservation of Nature Annual Report 2016*. Gland, Switzerland and Cambridge, UK: IUCN.

McPherson, S. (2016). *Britain's Treasure Islands. A Journey to the UK Overseas Territories*. Poole, UK: Redfern Natural History Productions.

Morris, J.A., Jr., and Whitfield, P.E. (2009). *Biology, Ecology, Control and Management of the Invasive Indo-Pacific Lionfish: An Updated Integrated Assessment*. NOAA Technical Memorandum NOS NCCOS 99.

Perez, C.E. (2006). *Biodiversity Action Plan, Gibraltar: Planning for Nature*. Gibraltar: The Gibraltar Ornithological & Natural History Society.

Tye, A. (2009). *Guidelines for Invasive Species Management in the Pacific: A Pacific Strategy for Managing Pests, Weeds and Other Invasive Species*. Apia, Samoa: SPREP.

UNEP (2011). *The Strategic Plan for Biodiversity 2011–2020 and the Aichi Biodiversity Targets*. UNEP/CBD/COP/DEC/X/2, 29 October 2010, Nagoya, Japan. COP CBD Tenth Meeting.

Vaas, J., Driessen, P.P.J., Giezen, M., van Laerhoven, F. and Wassen, M.J. (2017). 'Who's in charge here anyway? Polycentric governance configurations and the development of policy on invasive alien species in the semisovereign Caribbean'. *Ecology and Society* 22(4): 1.

Varnham, K. (2006). *Non-native Species in UK Overseas Territories: A Review*. JNCC Report No. 372.

A little goes a long way when controlling invasive plants for biodiversity conservation

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Abstract Invasive species, particularly animals, are being eradicated from islands at ever more ambitious scales. In order to protect island biodiversity and the essential ecosystem functions that it provides, however, plant invasions should be given more management attention. While many advances have been made, plant eradication is inherently more difficult than animal eradication due to persistent seed banks, and eradication may not be possible for more extensive populations. While maintenance control has been successful, critics question the sustainability and priority of these efforts, and targets vary widely. Developing consistent and informed targets requires an understanding of how biodiversity varies with invader cover, yet little is known about this topic. Our research suggests that limited control efforts may be highly beneficial. We conducted a meta-analysis of 54 studies to investigate the effects of plant invasions on invertebrate diversity, incorporating invader cover and residence time as potential causal mechanisms. We also contrasted restored plots with otherwise native plots. We found that invertebrate species richness was 31% lower in exotic plots than in native plots, and that there is a threshold at around 70% invader cover after which the negative effects are significant across all studies. Furthermore, these negative effects tended to decrease with time, and invertebrate richness was even greater in restored plots. The implication is that by removing 30% or less of invasive plant cover and restoring natives, we can achieve many of our conservation goals. We argue that by maintaining invasive patches at or below 70% exotic cover at a site in the near term, we can buy time for both the islands' insect herbivores to adapt to use the invader, and for managers to continue improving plant eradication technologies. By retaining native diversity in this way, we can help to increase the resistance and resilience of these systems to global change and other stressors.

Keywords: invader management, invasive plants, invertebrates, island biodiversity, meta-analysis, threshold

INTRODUCTION

Islands support many organisms found nowhere else in the world, and contribute disproportionately to global biodiversity (Kier, et al., 2009). They also provide critical habitat for 45% of the IUCN-listed species (Keitt, et al., 2011). To protect this extraordinary biological diversity, invasive species are being eradicated from islands at ever more ambitious scales (Clout & Veitch, 2002; Burbidge, 2011), and eradication is increasingly promoted as an important direction for island conservation. This is promoting recovery of many rare and endangered species, and of biodiversity as a whole (e.g. Klinger, et al., 2002; Rauzon, et al., 2002). This retained biodiversity can increase the stability of a system (Hautier, et al., 2015), its resistance and resilience to global change (Mori, et al., 2013; Isbell, et al., 2015) and its resistance to further invasion (Tilman, 1999).

Island eradication and control efforts overwhelmingly target invasive vertebrates, as an analysis of previous Island Invasives conference proceedings reveals (Veitch & Clout, 2002; Veitch, et al. 2011; 87% and 97% of the papers, respectively). Yet plant invaders are also key factors in native biodiversity decline (Wilcove, et al., 1998; Gaertner, et al., 2009), with their impacts to disturbance regimes, nutrient cycling, and fluxes of materials and energy altering ecosystem structure and function (Mack & D'Antonio, 1998; Liao, et al., 2008; Ehrenfeld, 2010). Furthermore, invasive animal removals often result in the ecological release of invasive plants (e.g., Klinger, et al., 2002; Zavaleta, et al., 2001). In order to protect island biodiversity and the essential ecosystem functions that it provides, plant invasions should be given more management attention. Yet eradication, the widely preferred alternative to control (Clout & Veitch, 2002; Burbidge, 2011), is often problematic for invasive plants.

Plant eradication is inherently more difficult, and generally more expensive, than animal eradication due to persistent seed banks, although many advances have been made. Under the right conditions, seeds can persist for several hundred years or more (Jha, 2005). On the Pacific Islands of French Polynesia, Hawaii, and New Caledonia, eradication of the invasive alien tree *Miconia calvescens* has not yet been completed despite more than 15 years of intensive control, due to a prolific and persistent seed bank (Meyer, et al., 2011). Similar issues have plagued an eradication programme for *Sagina procumbens* on Gough Island in the South Atlantic, despite an impressive array of innovative control techniques (Cooper, et al., 2011). Invasive plant eradication can be achieved, but it typically involves small populations, treated early in the invasion process, with a swift and strong response (Mack & Lonsdale, 2002; Rejmanek & Pitcairn, 2002).

Where eradication is not feasible, maintenance control may be implemented. Maintenance control is the "coordinated and consistent management of invasive plants in order to maintain the plant population at low levels" (University of Florida, 2018). This approach has been successful, but typically requires a large labour force, and critics question the sustainability and priority of these efforts (Simberloff, 2009). Furthermore, targets for native cover vary widely and invasive cover targets are typically highly stringent. For example, a survey of 21 California habitat restoration plans containing specified thresholds (gathered via a Google search) reveals native cover targets ranging from 15% to 90%, with an average target of 62% (n=20). Exotic cover targets, on the other hand, were never greater than 10% (n=7). It is also unclear how these targets were derived. Developing consistent and informed targets requires an understanding of how biodiversity varies with invader cover, however little is known about this topic.

An important link between plant communities and the greater food web is the invertebrate fauna. Invertebrates are a key component of biodiversity, comprising 97% of all animal species (Spelman, 2012) and playing key roles in nutrient recycling, pollination, seed dispersal, energy flow,

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and structuring plant and animal communities (Gullan & Cranston, 2005). They also respond quickly, sensitively, and locally to environmental changes (Kremen, et al., 1993), and are thus excellent indicators of the consequences of plant invasions and other disturbances. Analysis of invertebrate responses to plant invasions can help delineate the drivers of biodiversity and community patterns, thus guiding the conservation and restoration of diverse native ecosystems (Lodge, 1993; McMahon, et al., 2006).

We conducted a meta-analysis to investigate the effects of plant invasions on invertebrate diversity (as a whole, including both native and non-native species), incorporating invader cover and residence time in the system as potential explanatory variables. We also contrasted the type of sites (restored or intact) used as the native comparison. A meta-analysis approach can be used to combine multiple studies and detect overall trends in biotic responses to environmental factors. Our research suggests that in control efforts, a little may go a long way.

METHODS

We compiled studies through both database queries and subsequent surveys of the references cited in compiled papers. We searched ISI Web of Science in November 2012, using the search string "Topic = (invasive OR exotic AND plant) AND Topic = (arthropod* OR insect* OR invertebrate*). From these searches, we assembled 106 published studies which compared insect, arthropod, or other invertebrate diversity in invaded versus native habitats. These studies were from both island and mainland environments and included dissertations. Studies included by richness and other diversity indices, which were analysed separately. We extracted the data directly from tables or from graphs using the programme DigitizeIt v. 1.5 (Island Bormann, Braunschweig, Germany: <http://www.digitizeit.de>).

Fifty-four studies were eligible for testing using a meta-analysis approach (means, variances and sample sizes were reported) and are included in our meta-analysis (Appendix 1). These studies represent a variety of habitat types throughout the world, ranging from grassland to scrub to riparian. Fifty-two of these studies reported invertebrate richness, and fifteen studies reported values for diversity indices incorporating evenness, with 12 reporting results for the Shannon index, two for the Simpson's index, and one for Fisher's alpha. Insects were the focus of 26 studies, while 16 studies reported results for entire arthropod assemblages, and 12 studies described results for other invertebrate groups.

We extracted descriptor variables, where available, from each study, including latitude, time since establishment of the non-native plant at both the local (study site) and/or regional (hundreds of square kilometres) scale, invader cover, and whether or not the native-dominated site was restored habitat. Where time since establishment was not reported for a given study, we obtained this information from other sources where possible. In order to utilise the studies which reported cover classes or ranges rather than exact values (over half of them), we placed invader cover into six cover classes. We used natural breaks in the data to develop the following classes: <10%, 10–30%, 30–50%, 50–70%, 70–90%, and >90%. Cover was thus considered 'absolute' and not relative. Studies reporting that the invasive plant "formed a monoculture", was "dense and continuous," or "completely dominated the landscape" were conservatively classified into the 70–90% group. We found that model results were not changed by reclassifying these into either 50–70% or >90% cover.

We used the response ratio as an estimator of effect size; in this case, the natural log of the ratio (X_{exotic}/X_{native}), where X represents the mean of either invertebrate species richness or diversity index (analysed separately) for a given study in either the 'exotic' or the 'native' locations. We chose the response ratio for several reasons: first, we were interested in the magnitude of the relative difference in invertebrate diversity between exotic and native vegetation; second, use of the logarithm ensures that deviations in these two variables are treated equally (Hedges, et al., 1999). Lastly, it allowed us to assess both the model and residual variation, giving an estimate of the importance of the variables analysed here.

We calculated a single effect size per study by averaging data collected over multiple years or seasons. When we compared invertebrate richness or diversity in one native area to those in multiple invaded areas or vice versa, we calculated separate effect sizes for each comparison. When studies included multiple levels of descriptor variables (e.g. two or more establishment times), we calculated an average effect size to determine the overall effect of invasion (vs. native plant communities) but calculated separate effect sizes for each level of the descriptor variables when analysing the effects of these descriptor variables on invertebrate richness or diversity.

We performed meta-analyses using the *metafor* (Viechtbauer, 2010) package for R 2.15.0, and used random effects models to calculate overall effect sizes for invertebrate richness and diversity (Viechtbauer, 2010; Gurevitch & Hedges, 1999). To estimate the variation in the effect size described by different categorical variables (cover, study scale, and type of control plot), we used mixed-effects models using the Q statistic. This analysis treats the variables as fixed but includes a random variance component to account for variability across the studies. In one case (invader cover), we also report results from a fixed-effects model, which restricts our inferences to the studies examined. For continuous descriptor variables (latitude, invader time since establishment) we used weighted generalised least squares regression to test their relationships with effect size.

After accounting for the variation attributable to descriptor variables, we estimated residual variation (τ^2) using a restricted maximum likelihood estimator (Viechtbauer, 2005). For studies which reported results for all descriptor variable groups (22), we used the Akaike information criterion (AIC) to determine the model that best fit the data.

RESULTS

Invertebrate species richness was 31% lower in exotic plots than in native plots (effect size = -0.37 ± 0.10 on a 0–1 scale; Z = -5.48, $p < 0.01$; Fig. 1). There was a high amount of variation in the studies using richness to indicate diversity, however (Q = 111, $p < 0.001$). Invertebrate diversity indices that incorporate evenness were less strongly affected than richness values, but still 14% lower in exotic plots (effect size = -0.15 ± 0.10; Z = -3.42, $p < 0.01$). Unlike the effect sizes for species richness, there was not much variation among studies using diversity indices (Q = 13, $p > 0.50$). The absolute value of latitude did not explain a significant amount of heterogeneity in effect sizes for species richness (Q = 1.09, $p = 0.30$), nor did study scale (Q = 0.06, p = 0.97).

Using just data from native plots that had not undergone habitat restoration, invaded plots had lower invertebrate richness compared to native plots (-0.35 ± 0.07; Z = -5.02, $p < 0.01$). There was a stronger effect when plots restored to native species were used for comparison (-0.61 ± 0.17; Z = -1.73, p = 0.08), although this was just a statistical trend, likely due to both low sample size (n=11) and high variability. When analysed together, effect sizes were significantly more negative for the comparisons between invaded and restored sites than invaded vs. otherwise native sites (Q = 5.1, p = 0.02; Fig. 1), indicating that invertebrate diversity was even greater in restored plots than in native plots that did not undergo habitat restoration.

At the local scale, the negative effects of invasive plants on invertebrate richness were greatest at the shortest time since establishment and decreased with time, but this pattern relies on a few key data points and was only marginally significant (Q = 3.0, p =0.08, Fig. 2). At the regional scale, time since invader establishment was not related to effect size (Q= 0.40, $p > 0.50$).

The impact of exotic plants on invertebrate species richness was highly variable below 70% invader cover, and only cover classes above 70% had confidence intervals that did not overlap zero (Fig. 1). When the cover classes below 70% were combined into a single category, the difference in effect sizes between exotic plant cover classes was marginal in a mixed-model analysis (Q = 4.7, p = 0.09), while the groups were very different when the data were fitted to a fixed effects model (Q = 176, $p < 0.0001$).

In all models except time since establishment, residual heterogeneity was significant (p < 0.01), indicating substantial amounts of variation in the effects that were not explained by the models. The effects of descriptor variables on effect sizes for diversity indices were not analysed, both because low sample sizes prevented it and because low residual heterogeneity obviated the need for it.

DISCUSSION

Our results showed a clear negative effect of plant invasions on invertebrate richness and diversity. This has important implications for the diversity and function of the system as a whole, since insects and other invertebrates perform so many important roles in an ecosystem – including food provisioning for higher trophic levels such as reptiles and amphibians, birds, and small mammals (Weisser & Siemann, 2004).

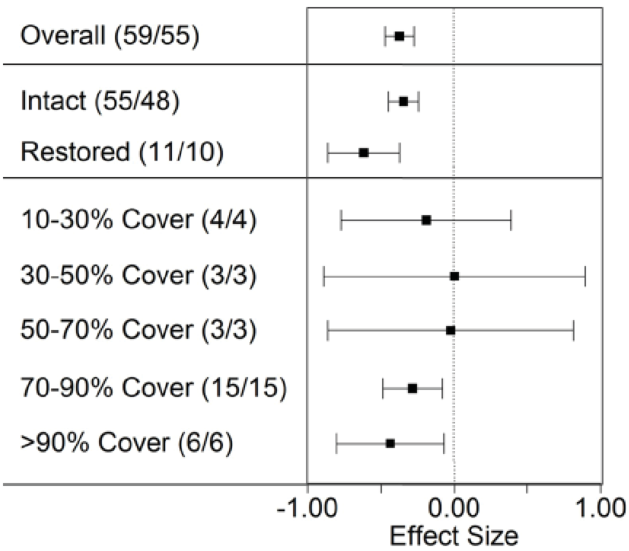

Fig. 1 Mean invertebrate richness effect sizes (± 95% confidence limits) across all studies (top panel), as well as between studies contrasting effect sizes where native plots represented restored or intact habitats (middle panel). The bottom panel shows mean richness effect sizes for exotic plant cover classes. Numbers in parentheses indicate the number of effect sizes and the total number of studies, respectively (some studies had more than one comparison).

Furthermore, the most consistent and significant negative effects of plant invaders on invertebrate richness occur when invasive plants comprise over 70% of cover. One likely reason for this threshold is a decline in the diversity of other plant species when an invader comes to dominate; for instance, Almeida-Neto, et al. (2011) found that only host plant richness explained the unimodal relationship they found between insect herbivore richness and invasive grass cover. Many previous studies have shown that insect and arthropod diversity is positively related to plant species richness, presumably owing to structural and food diversity as well as abiotic variables (e.g., temperature, moisture) (Price, et al., 2011).

The implication of these results is that, in general, with a moderate reduction of invasive plant cover and restoration of native plants to at least 30% cover, we can achieve meaningful progress towards the goal of biodiversity conservation. While some invasive plants will have impacts below this threshold (e.g. Knapp, 2014) this provides a general guideline in the absence of species-specific impact information. If a critical level of plant and invertebrate diversity can be maintained, then so can key ecosystem functions such as nutrient cycling and pollination (Gullan & Cranston, 2005).

Many will be legitimately concerned about indefinite “maintenance management” of plant invaders. Invasive plant management is challenging, and requires a long-term commitment (e.g. Mack & Lonsdale, 2002; Meyer, et al., 2011). However, holding that 70% line by removing invaders and, when needed, restoring at least 30% native plant cover will buy time, both: 1) to allow the islands’ insect herbivores to adjust to using the invader, and 2) for managers to continue improving plant control technologies and eradication strategies. We elaborate on these points below.

A novel plant species may be avoided by insect herbivores because it differs from native plants in characteristics such as nutritional quality, chemical composition, and architecture (Strong, et al., 1984; Kuhnle & Muller, 2009). Even a plant that can technically be eaten may be avoided because it is not recognised as a food source (Lankau, et al., 2004; Dudley, et al., 2012). The number of different herbivores using a novel plant tends to increase with the invader’s time since establishment, however (Kennedy & Southwood, 1984; Brandle, et al., 2008).

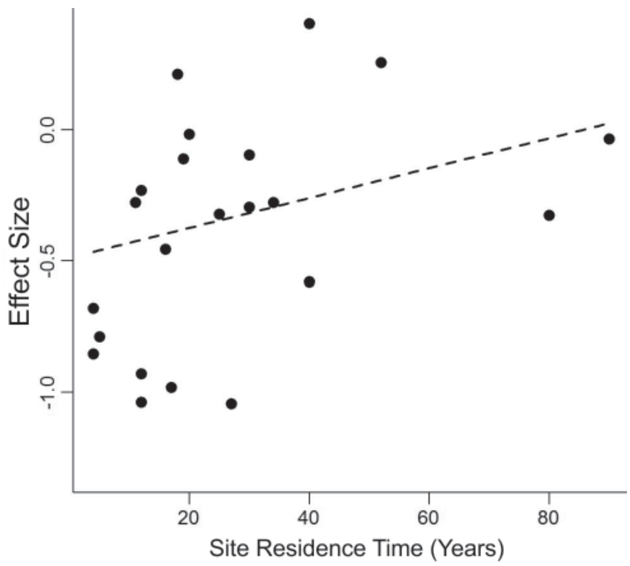

Fig. 2 Relationship between effect size for invertebrate richness and time since invader establishment at a site for the 22 studies for which these data were available. Dashes indicate line of best fit.

In our meta-analysis, where we consider the richness of invertebrates as a whole including multiple feeding guilds in addition to herbivores, we found a trend for invertebrate richness to increase with time since invader establishment (Fig. 2). This effect was only marginally significant – perhaps because it was driven by just a few key points, or perhaps because the effect of residence time is not as strong for invertebrates as a whole as it is for insect herbivores alone.

While these natural enemies are adapting to utilise invasive plant species over time, our control techniques are improving – allowing for both larger and more efficient, effective projects. For instance, a transition from ground to helicopter shooting enabled the eradication of goats on Western Australian islands (Burbidge & Morris, 2002), as did Judas goat technologies (Campbell & Donlan, 2005). Aerial surveys help with plant detection and eradication as well (Coulston, 2002; Knapp, et al., 2011), and treatment techniques have improved to avoid vectoring plant material (Coulston, 2002). Experimentation with techniques from hand-pulling to herbicide to heat and saltwater applications have improved the efficacy of invasive grass control efforts on Laysan Island (Flint & Rehkemper, 2002). Similarly, better herbicides and mapping systems have improved invasive plant control in New Zealand (Wotherspoon & Wotherspoon, 2002). Improvement in baiting technology has enabled the eradication of rats in multiple locations (Thomas & Taylor, 2002; Howald et al., 2007). Lastly, targeting multiple species at one time has proven to be both efficient and effective (Griffiths, 2011; Morrison, 2011).

It is heartening that our results showed restored plots containing even more invertebrate species than other native plots relative to invaded plots (although with greater variability). Flower visitors can be more diverse at restoration than reference sites, even after ≤ one year (Waltz & Covington, 2004; Lomov, et al., 2010). This may be because early-colonising butterflies can be attracted to more open, sunny restored areas disturbed by earth moving, invasive plant removal, and outplanting (Magoba & Samways, 2010; Hanula & Horn, 2011a). Conversely, butterfly richness can decrease as percent plant cover rises (Florens, et al., 2010). Higher invertebrate richness in restored areas is likely also related to greater plant richness and cover (Hanula & Horn, 2011), perhaps due to elements of both early- and later-successional communities being present. In this case, richness would also decrease with time as succession occurs.

CONCLUSION

The theme of this conference is "Scaling Up to Meet the Challenge." Invasive species eradication successes are being achieved at ever-increasing scales, but more attention should be paid to the significant threat of plant invasions. Although invasive plant control is challenging, our research suggests that reducing invader density to just 70% cover can have significant benefits for invertebrate biodiversity and thus ecosystem function. Furthermore, habitat restoration can give that diversity an extra boost. While the existence of seed banks dictates that this is a long-term proposition, we argue that, over time, insect herbivores will adapt to using the invader, while land managers develop ever-better control technologies. The biodiversity that is thus conserved will increase the resistance and resilience of these systems to further invasion and other stressors such as global climate change (Millennium Ecosystem Assessment, 2003; Haddad, et al., 2011), and allow us to truly achieve island conservation.

REFERENCES

Almeida-Neto, M., Prado, P.I. and Lewinsohn, T.M. (2011). 'Phytophagous insect fauna tracks host plant responses to exotic grass invasion'. *Oecologia* 165: 1051–1062.

Ando, Y., Utsumi, S., and Ohgushi, T. (2010). 'Community structure of insect herbivores on introduced and native *Solidago* plants in Japan'. *Entomologia Experimentalis et Applicata* 136: 174–183.

Bailey, J.K., Schweitzer, J.A. and Whitham, T.G. (2001). 'Salt cedar negatively affects biodiversity of aquatic macroinvertebrates'. *Wetlands* 21: 442–447.

Bartomeus, I., Vila, M., and Santamaria, L. (2008). 'Contrasting effects of invasive plants in plant-pollinator networks'. *Oecologia* 155: 761–770.

Bassett, I.E., Paynter, Q. and Beggs, J.R. (2012). 'Invertebrate community composition differs between invasive herb alligator weed and native sedges'. *Acta Oecologica–International Journal of Ecology* 41: 65–73.

Bickel, T.O. and Closs, G.P. (2009). 'Impact of partial removal of the invasive macrophyte *Lagarosiphon major* (Hydrocharitaceae) on invertebrates and fish'. *River Research and Applications* 25: 734–744.

Bock, C.E., Bock, J.H., Jepson, K.L. and Ortega, J.C. (1986). 'Ecological effects of planting African love-grasses in Arizona'. *National Geographic Research* 2: 456–463.

Brandle, M., Kuhn, I., Klotz, S., Belle, C. and Brandl, R. (2008). 'Species richness of herbivores on exotic host plants increases with time since introduction of the host'. *Diversity and Distributions* 14: 905–912.

Burbidge, A.A. and Morris, K.D. (2002). 'Introduced mammal eradications for nature conservation on Western Australian Islands: A review'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 64–70. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Burbidge, A.A. (2011). '2001 to 2010 and beyond: trends and future directions in the eradication of invasive species on islands'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 515–519. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Burghardt, K.T., Tallamy, D.W. and Shriver, W.G. (2009). 'Impact of native plants on bird and butterfly biodiversity in suburban landscapes'. *Conservation Biology* 23: 219–224.

Cameron, G.N. and Spencer, S.R. (2010). 'Entomofauna of the introduced Chinese tallow tree'. *Southwestern Naturalist* 55: 179–192.

Campbell, K. and Donlan, C.J. (2005). 'Feral goat eradications on islands.' *Conservation Biology* 19: 1362–1374.

Chey, V.K., Holloway, J.D., Hambler, C. and Speight, M.R. (1998). 'Canopy knockdown of arthropods in exotic plantations and natural forest in Sabah, north-east Borneo, using insecticidal mist-blowing'. *Bulletin of Entomological Research* 88: 15–24.

Christopher, C.C. and G.N. Cameron (2012). 'Effects of invasive Amur honeysuckle (*Lonicera maackii*) and white-tailed deer (*Odocoileus virginianus*) on litter-dwelling arthropod communities'. *American Midland Naturalist* 167: 256–272.

Clout, M.N. and Veitch, C.R. (2002). 'Turning the tide of biological invasion: the potential for eradicating invasive species'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 1–3. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Cooper, J., Cuthbert, R.J., Gremmen, N.J.M., Ryan, P.G. and Shaw, J.D. (2011). 'Earth, fire and water: applying novel techniques to eradicate the invasive plant, procumbent pearlwort *Sagina procumbens*, on Gough Island, a World Heritage Site in the South Atlantic'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 162–165. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Cord, E.E. (2011). 'Changes in Arthropod Abundance and Diversity with Invasive Grasses'. MSc thesis. Kingsville, TX: Texas A&M.

Coulston, G.J. (2002). 'Control of invasive plants on the Poor Knights Islands, New Zealand'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 79–84. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

de Groot, M., Kleijn, D. and Jogan, N. (2007). 'Species groups occupying different trophic levels respond differently to the invasion of semi-natural vegetation by *Solidago canadensis'*. *Biological Conservation* 136: 612–617.

Dudley, T.L., Bean, D.W., Pattison, R.R. and Caires, A. (2012). 'Selectivity of a biological control agent, *Diorhabda carinulata* Desbrochers, 1870 (Coleoptera: Chrysomelidae) for host species within the genus *Tamarix* Linneaus, 1753'. *Pan-Pacific Entomologist* 88: 319–341.

Durst, S.L., Theimer, T.C., Paxton, E.H. and Sogge, M.K. (2008). 'Temporal variation in the arthropod community of desert riparian habitats with varying amounts of saltcedar (*Tamarix ramosissima*)'. *Journal of Arid Environments* 72: 1644–1653.

Ehrenfeld, J.G. (2010). 'Ecosystem consequences of biological invasions'. *Annual Review of Ecology, Evolution, and Systematics* 41: 59–80.

Flint, E. and Rehkemper, C. (2002). 'Control and eradication of the introduced grass, *Cenchrus echinatus*, at Laysan Island, Central Pacific Ocean'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 110–115. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Florens, F.B.V., Mauremootoo, J.H., Fowler, S.V., Winder, L. and Baider, C. (2010). 'Recovery of an indigenous butterfly community following control of invasive alien plants in a tropical island's wet forests'. *Biodiversity and Conservation* 19: 3835–3848.

Gaertner, M., Den Breeyen, A., Hue, C. and Richardson, D.M. (2009). 'Impacts of alien plant invasions on species richness in Mediterranean-type ecosystems: a meta-analysis'. *Progress in Physical Geography* 33: 319–338.

Gerber, E., Krebs, C., Murrell, C., Moretti, M., Rocklin, R. and Schaffner, U. (2008). 'Exotic invasive knotweeds (*Fallopia* spp.) negatively affect native plant and invertebrate assemblages in European riparian habitats'. *Biological Conservation* 141: 646–654.

Gossner, M. and Ammer, U. (2006). 'The effects of Douglas fir on tree-specific arthropod communities in mixed species stands with European beech and Norway spruce'. *European Journal of Forest Research* 125: 221–235.

Gremmen, N.J.M., Chown, S.L. and Marshall, D.J. (1998). 'Impact of the introduced grass *Agrostis stolonifera* on vegetation and soil fauna communities at Marion Island, sub-Antarctic'. *Biological Conservation* 85: 223–231.

Griffiths, R. (2011). 'Targeting multiple species – a more efficient approach to pest eradication'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 172–176. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Gullan, P.J. and Cranston, P.S. (2005). *The Insects: An Outline of Entomology*. Malden, MA, USA: Blackwell Publishing Ltd.

Gurevitch, J. and Hedges, L.V. (1999) 'Statistical issues in ecological meta-analyses'. *Ecology* 80: 1142–1149.

Haddad, N.M., Crutsinger, G.M., Gross, K., Haarstad, J. and Tilman, D. (2011). 'Plant diversity and the stability of foodwebs'. *Ecology Letters* 14: 42–46.

Hagen, E.N., Bakker, J.D. and Gara, R.I. (2010). 'Aerial arthropod communities of native and invaded forests, Robinson Crusoe Island, Chile'. *Environmental Entomology* 39: 1159–1164.

Hanula, J.L. and Horn, S. (2011a). 'Removing an exotic shrub from riparian forests increases butterfly abundance and diversity'. *Forest Ecology and Management* 262: 674–680.

Hanula, J.L. and Horn, S. (2011b). 'Removing an invasive shrub (Chinese privet) increases native bee diversity and abundance in riparian forests of the southeastern United States'. *Insect Conservation and Diversity* 4: 275–283.

Hartley, M.K., Rogers, W.E. and Siemann, E. (2010). 'Comparisons of arthropod assemblages on an invasive and native trees: Abundance, diversity and damage'. *Arthropod–Plant Interactions* 4: 237–245.

Harvey, K.J., Britton, D.R. and Minchinton, T.E. (2010). 'Insect diversity and trophic structure differ on native and non-indigenous congeneric rushes in coastal salt marshes'. *Austral Ecology* 35: 522–534.

Hautier, Y., Tilman, D., Isbell, F., Seabloom, E.W., Borer, E.T. and Reich, P.B. (2015). 'Anthropogenic environmental changes affect ecosystem stability via biodiversity'. *Science* 348: 336–340.

Hedges, L.V., Gurevitch, J. and Curtis, P.S. (1999). 'The meta-analysis of response ratios in experimental ecology'. *Ecology* 80: 1150–1156.

Herrera, A.M. and Dudley, T.L. (2003). 'Reduction of riparian arthropod abundance and diversity as a consequence of giant reed (*Arundo donax*) invasion'. Biological Invasions 5(3): 167–177.

Hills, N., Hose, G.C., Cantlay, A.J. and Murray, B.R. (2008). 'Cave invertebrate assemblages differ between native and exotic leaf litter'. *Austral Ecology* 33: 271–277.

Holmquist, J.G., Schmidt-Gengenbach, J. and Slaton, M.R. (2011). 'Influence of invasive palms on terrestrial arthropod assemblages in desert spring habitat'. *Biological Conservation* 144: 518–525.

Howald, G., Donlan, C.J., Galvan, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P. Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Hugel, S. (2012). 'Impact of native forest restoration on endemic crickets and katydids density in Rodrigues island'. *Journal of Insect Conservation* 16: 473–477.

Isbell, F., Craven, D., Connolly, J., Loreau, M., Schmid, B., Beierkuhnlein, C., Bezemer, T.M., Bonin, C., Bruelheide, H., de Luca, E., Ebeling, A., Griffin, J.N., Guo, Q.F., Hautier, Y., Hector, A., Jentsch, A., Kreyling, J., Lanta, V., Manning, P., Meyer, S.T., Mori, A.S., Naeem, S., Niklaus, P.A., Polley, H.W., Reich, P.B., Roscher, C., Seabloom, E.W., Smith, M.D., Thakur, M.P., Tilman, D., Tracy, B.F., van der Putten, W.H., van Ruiven, J., Weigelt, A., Weisser, W.W., Wilsey, B. and Eisenhauer, N. (2015). 'Biodiversity increases the resistance of ecosystem productivity to climate extremes'. *Nature* 526: 574–577.

Jha, A. (2005). 'How long can a seed stay alive?' *The Guardian* (16 June 2005). <https://www.theguardian.com/science/2005/jun/16/thisweekssciencequestions1>. Accessed May 2018.

Kappes, H., Lay, R. and Topp, W. (2007). 'Changes in different trophic levels of litter-dwelling macrofauna associated with giant knotweed invasion'. *Ecosystems* 10: 734–744.

Keitt, B., Campbell, K., Saunders, A., Clout, M. Wang, Y., Heinz, R., Newton, K. and Tershy, B. (2011). 'The global islands invasive vertebrate eradication database: a tool to improve and facilitate restoration of island ecosystems'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 74–77. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Kennedy, C.E.J. and Southwood, T.R.E. (1984). 'The number of species of insects associated with British trees – a re-analysis'. *Journal of Animal Ecology* 53: 455–478.

Kier, G., Kreft, H., Lee, T.M., Jetz, W., Ibisch, P., Nowicki, C., Mutke, J. and Barthlott, W. (2009). 'A global assessment of endemism and species richness across island and mainland regions'. *Proceedings of the National Academy of Sciences* 106: 9322–9327.

Klinger, R.C., Schuyler, P. and Sterner, J.D. (2002). 'The response of herbaceous vegetation and endemic plant species to the removal of feral sheep from Santa Cruz Island, California'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 141–154. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Knapp, J.J., Schuyler, P.T., Walker, K.N., Macdonald, N.L. and Morrison, S.A. (2011). 'Benefits of supporting plant and animal eradication projects with helicopters'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 188–191. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Knapp, D.A. (2014). 'Effects of an Exotic Plant Invasion on Arthropod Assemblages.' PhD thesis. Santa Barbara, CA, USA: University of California, Santa Barbara.

Kremen, C., Colwell, R.K., Erwin, T.L., Murphy, D.D., Noss, R.F. and Sanjayan, M.A. (1993). 'Terrestrial arthropod assemblages: Their use in conservation planning'. *Conservation Biology* 7: 796–808.

Kuhnle, A. and Muller, C. (2009). 'Differing acceptance of familiar and unfamiliar plant species by an oligophagous beetle'. *Entomologia Experimentalis et Applicata* 131: 189–199.

Lankau, R.A., Rogers, W.W. and Siemann, E. 2004. 'Constraints on the utilisation of the invasive Chinese tallow tree *Sapium sebiferum* by generalist native herbivores in coastal prairies'. *Ecological Entomology* 29: 66–75.

Liao, C., Peng, R., Luo, Y., Zhou, X., Wu, X., Fang, C., Chen, J. and Li, B. (2008). 'Altered ecosystem carbon and nitrogen cycles by plant invasion: a meta-analysis'. *New Phytologist* 177: 706–714.

Lodge, D.M. (1993). 'Biological invasions: Lessons for ecology'. *Trends in Ecology and Evolution* 8: 133–137.

Lomov, B., Keith, D.A. and Hochuli, D.F. (2010). 'Pollination and plant reproductive success in restored urban landscapes dominated by a pervasive exotic pollinator'. *Landscape and Urban Planning* 96: 232–239.

McMahon, S.M., Cadotte, M.W. and Fukami, T. (2006). 'Tracking the Tractable: Using Invasions to Guide the Exploration of Conceptual Ecology'. In: M.W. Cadotte, S.M. McMahon and T. Fukami (eds.) *Conceptual Ecology and Invasion Biology*, pp. 3–14. Dordrecht, the Netherlands: Springer.

Mack, M.C. and D'Antonio, C.M. (1998). 'Impacts of biological invasions on disturbance regimes'. *Trends in Ecology and Evolution* 13: 195–198.

Mack, R.N. and Lonsdale, W.M. (2002). 'Eradicating invasive plants: hard-won lessons for islands'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 164–172. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Magoba, R.N. and Samways, M.J. (2010). 'Recovery of benthic macroinvertebrate and adult dragonfly assemblages in response to large scale removal of riparian invasive alien trees'. *Journal of Insect Conservation* 14: 627–636.

Magoba, R.N. and M.J. Samways (2012). 'Comparative footprint of alien, agricultural and restored vegetation on surface-active arthropods'. *Biological Invasions* 14: 165–177.

Magura, T., Tothmeresz, B. and Bordan, Z. (2000). 'Effects of nature management practice on carabid assemblages (Coleoptera: Carabidae) in a non-native plantation'. *Biological Conservation* 93: 95–102.

Meyer, J-Y, Loope, L. and Goarant, A-C. (2011). 'Strategy to control the invasive alien tree *Miconia calvescens* in Pacific Islands: eradication, containment or something else?'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 91–96. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Millennium Ecosystem Assessment (2003). *Ecosystems and human well-being: Biodiversity synthesis*. Washington, DC, USA: World Resources Institute.

Mori, A.S., Furukawa, T. and Sasaki, T. (2013). 'Response diversity determines the resilience of ecosystems to environmental change'. *Biological Reviews* 88: 349–364.

Moron, D., Lenda, M., Skorka, P., Szentgyorgyi, H., Settele, J. and Woyciechowski, M. (2009). 'Wild pollinator communities are negatively affected by invasion of alien goldenrods in grassland landscapes'. *Biological Conservation* 142: 1322–1332.

Morrison, S.A. (2011). 'Trophic considerations in eradicating multiple pests'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 208–212. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Osunkoya, O.O., Polo, C. and Andersen, A.N. (2011). 'Invasion impacts on biodiversity: Responses of ant communities to infestation by cat's claw creeper vine, *Macfadyena unguis-cati* (Bignoniaceae) in subtropical Australia'. *Biological Invasions* 13: 2289–2302.

Parr, C.L., Ryan, B.J. and Setterfield, S.A. (2010). 'Habitat complexity and invasive species: The impacts of Gamba grass (*Andropogon gayanus*) on invertebrates in an Australian tropical savanna'. *Biotropica* 42: 688–696.

Pinto, C., Sousa, J.P., Graca, M.A.S. and da Gama, M.M. (1997). 'Forest soil Collembola: Do tree introductions make a difference?' *Pedobiologia* 41: 131–138.

Price, P.W., Denno, R.F., Eubanks, M.D., Finke, D.L. and Kaplan, I. (2011). *Insect Ecology: Behavior, Populations and Communities*. Cambridge, MA, USA: Cambridge University Press.

Pryke, J.S. and M.J. Samways (2009). 'Recovery of invertebrate diversity in a rehabilitated city landscape mosaic in the heart of a biodiversity hotspot'. *Landscape and Urban Planning* 93: 54–62.

Rauzon, M.J., Forsell, D.J. and Flint, E.N. (2002). 'Seabird re-colonisation after cat eradication on Equatorial Jarvis, Howland, and Baker islands, USA, Central Pacific'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* p. 411. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Rejmanek, M. and Pitcairn, M.J. (2002). 'When is eradication of exotic pest plants a realistic goal?' In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 249–253. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Robertson, M.P., Harris, K.R., Coetzee, J.A., Foxcroft, L.C., Dippenaar-Schoeman, A.S. and van Rensburg, B.J. (2011). 'Assessing local scale impacts of *Opuntia stricta* (Cactaceae) invasion on beetle and spider diversity in Kruger National Park, South Africa'. *African Zoology* 46: 205–223.

Samways, M.J. and N.J. Sharratt (2010). 'Recovery of endemic dragonflies after removal of invasive alien trees'. *Conservation Biology* 24: 267–277.

Samways, M.J., Sharratt, N.J. and Simaika, J.P. (2011). 'Effect of alien riparian vegetation and its removal on a highly endemic river macroinvertebrate community'. *Biological Invasions* 13: 1305–1324.

Sax, D. (2002). 'Equal diversity in disparate species assemblages: a comparison of native and exotic woodlands in California'. *Global Ecology and Biogeography* 11: 49–57.

Schirmel, J., Timler, L. and Buchholz, S. (2011). 'Impact of the invasive moss *Campylopus introflexus* on carabid beetles (Coleoptera: Carabidae) and spiders (Araneae) in acidic coastal dunes at the southern Baltic Sea'. *Biological Invasions* 13: 605–620.

Schoeman, C.S. (2008). 'Synergistic Impact of Invasive Alien Plants and the Alien Argentine Ant on Local Ant Assemblages in the Western Cape'. MSc thesis. Stellenbosch, South Africa: University of Stellenbosch.

Simao, M.C.M., Flory, S.L. and Rudgers, J.A. (2010). 'Experimental plant invasion reduces arthropod abundance and richness across multiple trophic levels'. *Oikos* 119: 1553–1562.

Simberloff, D. (2009). 'We can eliminate invasions or live with them. Successful management projects'. *Biological Invasions* 11: 149–157.

Spelman, L. 2012. *National Geographic Animal Encyclopedia*. Washington, DC: National Geographic Society.

St John, M.G., Wall, D.H. and Hunt, H.W. (2006). 'Are soil mite assemblages structured by the identity of native and invasive alien grasses'? *Ecology* 87: 1314–1324.

Strong, D.R., Lawton, J.H. and Southwood, R. (1984). *Insects on Plants: Community Patterns and Mechanisms*. Cambridge, MA, USA: Harvard University Press.

Tallamy, D.W. and Shropshire, K.J. (2009). 'Ranking Lepidopteran use of native versus introduced plants'. *Conservation Biology* 23: 941–947.

Talley, T.S., Nguyen, K.C. and Nguyen, A. (2012). 'Testing the effects of an introduced palm on a riparian invertebrate community in southern California'. *PLOS ONE* 7(8): e42460.

Theel, H.J., Dibble, E.D. and Madsen, J.D. (2008). 'Differential influence of a monotypic and diverse native aquatic plant bed on a macroinvertebrate assemblage: An experimental implication of exotic plant induced habitat'. *Hydrobiologia* 600: 77–87.

Thomas, B.W. and Taylor, R.H. (2002). 'A history of ground-based rodent eradication techniques developed in New Zealand, 1959–1993'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 301–310. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Tilman, D. (1999). 'The ecological consequences of changes in biodiversity: A search for general principles'. *Ecology* 80: 1455–1474.

Triet, T., Man, L.C. and Nga, N.P. (2004). 'Impacts of *Mimosa pigra* on Native Plants and Soil Insect Communities in Tram Chim National Park, Vietnam'. In: M. Julien, G. Flanagan, T. Heard, B. Hennecke, Q. Paynter and C. Wilson (eds.) *Research and Management of Mimosa pigra*, pp. 22–27. Canberra, Australia: CSIRO Entomology.

Ulyshen, M.D., Horn, S. and Hanula, J.L. (2010). 'Response of beetles (Coleoptera) at three heights to the experimental removal of an invasive shrub, Chinese privet (*Ligustrum sinense*), from floodplain forests'. *Biological Invasions* 12: 1573–1579.

University of Florida, Center for Aquatic and Invasive Plants. (2018). 'Maintenance Control Strategy'. <https://plants.ifas.ufl.edu/manage/developing-management-plans/maintenance-control-strategy/>. Accessed May 2018.

Veitch, C.R. and Clout, M.N. (eds.) (2002). *Turning the tide: the eradication of invasive species*. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Veitch, C.R., Clout, M.N. and Towns, D.R. (eds.) (2011). *Island invasives: eradication and management*. Proceedings of the International Conference on Island Invasives. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Viechtbauer, W. (2010). 'Conducting meta-analyses in R with the *metafor* package'. *Journal of Statistical Software* 36: 1–48.

Waltz, A.E.M. and Covington, W.W. (2004). 'Ecological restoration treatments increase butterfly richness and abundance: Mechanisms of response'. *Restoration Ecology* 12: 85–96.

Webb, C.E., Oliver, I. and Pik, A.J. (2000). 'Does coastal foredune stabilization with *Ammophila arenaria* restore plant and arthropod communities in southeastern Australia'? *Restoration Ecology* 8: 283–288.

Weisser, W.W. and Siemann, E. (eds.) (2004). *Insects and Ecosystem Function*. Berlin, Germany: Springer-Verlag.

White, E.M., Wilson, J.C. and Clarke, A.R. (2008). 'Diversity and abundance of arthropod floral visitor and herbivore assemblages on exotic and native *Senecio* species'. *Plant Protection Quarterly* 23: 90–98.

Wilcove, D.S., Rothstein, D., Dubow, J., Phillips, A. and Losos, E. (1998). 'Quantifying threats to imperiled species in the United States'. *BioScience* 48: 607–615.

Wotherspoon, S.H. and Wotherspoon, J.A. (2002). 'The evolution and execution of a plan for invasive weed eradication and control, Rangitoto Island, Hauraki Gulf, New Zealand'. In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 381–388. Occasional Paper SSC no. 28. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Wu, Y.-T., Zhang, X.-D., Zhao, B., Jiang, L.-F., Chen, J.-K. and Li, B. (2009). 'Effects of saltmarsh invasion by *Spartina alterniflora* on arthropod community structure and diets'. *Biological Invasions* 11: 635–649.

Zavaleta, E.S., Hobbs, R.J. and Mooney, H.A. (2001). 'Viewing invasive species removal in a whole-ecosystem context'. *Trends in Ecology and Evolution* 16: 454–459.

Zuefle, M.E., Brown, W.P. and Tallamy, D.W. (2008). 'Effects of non-native plants on the native insect community of Delaware'. *Biological Invasions* 10: 1159–1169.

Appendix 1 Studies used in the meta-analysis and their attributes. "Time?" indicates whether or not time since establishment was reported in the reference, and "Cov?" indicates whether or not cover of the invader was reported. "Control" indicates if the native comparison included restored habitat. "#Exotics" indicates the number of different exotic plant species that were included in the study. "Richn." Indicates if the study evaluated invertebrate species richness, while "Div." indicates if the study evaluated invertebrate diversity.

Reference	Location	Latitude	Native Habitat	Time?	Cov?	Control	#Exotics	Richn.	Div.
Ando, et al., 2010	Central Japan	35.07	Experimental forest field	X			1	X	
Bailey, et al., 2001	Arizona, USA	34.67	Riparian woodland	X			1	X	
Bartomeus, et al., 2008	Spain	42.32	Mediterranean shrubland	X	X		2	X	
Bassett, et al., 2012	New Zealand	35.02	Lake margin	X	X		1	X	
Bickel & Closs, 2009	New Zealand	45.03	Littoral	X		restored	1	X	X
Bock, et al., 1986	Arizona, USA	31.65	Semidesert grassland	X	X		1	X	
Brandle, et al., 2008	Germany	51	Multiple				1	X	
Burghardt, et al., 2009	Pennsylvania, USA	40.25	Suburban residences				1	X	
Cameron & Spencer, 2010	Texas, USA	29.53	Coastal prairie	X	X		1	X	X
Chey, et al., 1998	Sabah, Borneo	5.42	Tropical rainforest	X			5	X	
Christopher & Cameron, 2012	Ohio, USA	39.12	Hardwood forest		X		1		X
Cord, 2011	Texas, USA	27.49	Grassland	X	X		1	X	
de Groot, et al., 2007	Slovenia	46.05	Agricultural fields & ruderal areas		X		1	X	X
Durst, et al., 2008	Arizona, USA	33.65	Floodplain	X	X		1	X	X
Florens, et al., 2010	Mauritius	20.4	"Indigenous forest"			restored	1	X	
Gerber, et al., 2008	Switzerland, Germany, & France	47	Grassland, scrub	X			1	X	
Gossner & Ammer, 2006	Germany	48.18	Spruce forest	X	X		1		X
Gremmen, et al., 1998	Marion Island, SubAntarctic	46.83	"Drainage lines"	X	X		1	X	
Hagen, et al., 2010	Robinson Crusoe Island, Chile	33.63	Lower montane forest	X			1	X	
Hanula & Horn, 2011a	Georgia, USA	33.88	Riparian hardwood forest	X	X	restored	1	X	X
Hanula & Horn, 2011b	Georgia, USA	33.88	Riparian hardwood forest	X	X	restored	1	X	X
Harris, et al., 2004	New Zealand	41.2	Kanuka scrub		X		1	X	
Hartley, et al., 2010	Texas, USA	29.53	Tree plantations	X	1		1	X	
Harvey, et al., 2010	Australia	34	Coastal salt marsh		X		1	X	X
Herrera & Dudley, 2003	California, USA	38.23	Riparian forest	X	X		1	X	X
Hills, et al., 2008	Australia	33.82	Cave trees				2	X	
Holmquist, et al., 2011	California, USA	36.45	Desert spring	X			1	X	

Appendix 1 (Cont'd) Studies used in the meta-analysis and their attributes. "Time?" indicates whether or not time since establishment was reported in the reference, and "Cov?" indicates whether or not cover of the invader was reported. "Control" indicates if the native comparison included restored habitat. "#Exotics" indicates the number of different exotic plant species that were included in the study. "Richn." Indicates if the study evaluated invertebrate species richness, while "Div." indicates if the study evaluated invertebrate diversity.

Reference	Location	Latitude	Native Habitat	Time?	Cov?	Control	#Exotics	Richn.	Div.
Hugel, 2012	Rodrigues Island, SW Indian Ocean	19.72	Tropical forest			restored	1	X	
Kappes, et al., 2007	Germany	51.15	Floodplains		X		1	X	
Magoba & Samways, 2010	South Africa	23.02	Riparian		X	restored	1	X	
Magoba & Samways, 2012	South Africa	18.9	Fynbos scrub		X	restored	1	X	
Magura, et al., 2000	Hungary	48.47	Oak-hornbeam forest	X	X		1	X	X
Moron, et al., 2009	Poland	50.05	Wet meadow	X	X		1	X	
Osunkoya, et al., 2011	Australia	27.83	Eucalyptus & subtropical rainforest	X	X		1	X	X
Parr, et al., 2010	Australia	12.72	Mesic eucalyptus savanna	X	X		1	X	
Pinto, et al., 1997	Portugal	40.28	Riparian				2	X	
Pryke & Samways, 2009	South Africa	33.95	Southern afrotemperate forest, fynbos scrub			restored	1	X	
Robertson, et al., 2011	South Africa	25	Savanna, Sabie-crocodile thorn thicket	X	X		1	X	
Samways & Sharrat, 2010	South Africa	33.55	Riparian			restored	1	X	
Samways, et al., 2011	South Africa	33.3	Riparian, fynbos scrub				1	X	
Sax, 2002	California, USA	37.88	Oak woodland	X			1	X	
Schirmel, et al., 2011	Germany	54.53	Coastal dunes within heath	X	X		1	X	
Schoeman, 2008	South Africa	34.05	Fynbos scrub, renosterveld	X			1	X	
Simao, et al., 2010	Indiana, USA	39.22	Experimental	X			1	X	
St John, et al., 2006	Kansas, USA	39.1	Prairie	X			1	X	
Tallamy & Shropshire, 2009	Eastern USA	36.5 to 45	Multiple				1	X	
Talley, et al., 2012	California, USA	32.75	Riparian woodland				1	X	X
Theel, et al., 2008	Mississippi, USA	33	Aquatic				1	X	
Triet, et al., 2004	Vietnam	10.7	Seasonally inundated grassland	X	X		1	X	X
Ulyshen, et al., 2010	Georgia, USA	33.88	Floodplain forest	X		restored	1	X	
Webb, et al., 2000	Australia	35.4	Coastal foredunes	X	X		1	X	
White, et al., 2008	Australia	27.83	Pasture & grassland	X			1	X	
Wu, et al., 2009	China	31.52	Salt marsh	X	X		1	X	X
Zuefle, et al., 2008	Delaware, USA	39.7	Common garden				1	X	

Achieving large scale, long-term invasive American mink control in northern Scotland despite short term funding

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Abstract The American mink *(Neovison vison)* has invaded most of the United Kingdom following escapes from fur-farms over decades. Its escalating impact on riparian and coastal biodiversity, including seabirds and water voles, is well documented. Starting in north-east Scotland in 2004, long-term, multi-institution mink control efforts have harnessed the enthusiasm of volunteer conservationists to push back the mink invasion over a vast area. Rather than the outcome of a single project with secured long-term funding, this achievement resulted from four successive joined up projects each with short-term funding. The beginnings of the project (2004–2006), under the auspices of the north-east Scotland Biodiversity Partnership were small scale (30 km^2) and centred upon a lowland remnant water vole meta-population. Mink control efforts were scaled-up to 6,000 km^2 of mostly marginal mink habitat as part of the Cairngorms Water Vole Conservation Project (2006–2009) centred on the newly established Cairngorms National Park. The project, led by the University of Aberdeen, was funded by a charity, a UK Research council and Scottish Natural Heritage and involved the national park authority, and three local fisheries trusts. The approach was to deploy a "rolling carpet" of mink control based on the use of mink rafts operated by volunteers and that facilitated mink detection and removal. Substantial funding was then secured for a successor project, the Scottish Mink Initiative (2010–2014) involving, all previous partners plus 14 local fisheries trusts coordinated by the Rivers and Fisheries Trusts of Scotland. Mink were pushed back over a vast area (29,000 km^2) and their spread in coastal areas of north-west Scotland was countered. After a period with minimal bridge funding, coordinated mink control efforts resumed, thanks to the newly funded Scottish Invasive Species Initiative (2017–2021) seeking to extend the approach used with mink to other riparian invasives. Mink remain scarce or absent and water voles are recovering spectacularly. Coordinated mink control delivered tangible conservation benefits and improved understanding of the socio-ecological system despite the challenges of short-term funding.

Keywords: adaptive management, American mink, *Arvicola*, *Neovison vison*, participation, Scotland, water vole

INTRODUCTION

While there have been enormous achievements and improvements in the eradication of a small number of invasive mammalian species (brown (*Rattus norvegicus*) black (*R. rattus*) and Pacific (*R. exulans*) rats, house mice (*Mus musculus*), rabbits (*Oryctolagus cuniculus*), feral cats (*Felis catus*)), on islands of increasing size (DIISE, 2015), there has been comparatively little progress with efforts and guidelines on how to durably control invasive species in those areas where eradication is presently an unattainable goal. Yet, prevention has failed in many areas, such that focussing invasive management efforts exclusively on islands where eradication can be achieved leaves much valued biodiversity impacted by invasive species. Thus, when considering whether to expand resources to protect native biodiversity against the impact of invasive species, a key unknown is what, if anything short of eradication, can be achieved cost effectively and what management regimes might be both ecologically effective and sustainable over the long term.

Eradication can only be achieved where immigration can be prevented or managed (Bomford & O'Brien, 1995). Where this condition is not met, as is the case on continental mainland and large island areas, control of invasives must be the management objective. New Zealand's so-called 'mainland islands' are areas where intensive conservation adaptive and integrated pest management regimes are applied and outcomes are closely monitored (Saunders & Norton, 2001). They are adjacent to other areas where invasives are not managed to the same extent, hence subjected to immigration that, if not dealt with, could lead to recolonisation.

A key feature of mainland islands is that conservation management must be designed so as to last in perpetuity to ensure that the biodiversity and socio-community gains are not lost. It is therefore especially crucial that siting considers all features that may make a mainland island defensible. This may include topography (e.g. presence of peninsulas), ecological gradients or socio-economic interest of the local community that may affect their willingness to participate in ongoing management and adopt biosecurity measures and even the erection of conservation fences (Glen, et al., 2013). An unavoidable corollary of planning for the very long term, is the need for long-term funding commitments. This is crucial to negate the risk that ecosystem restoration will one day be undone should a lack of resources preclude a rapid and decisive reaction following incursion by invasives into a mainland island. In this respect, the fact that New Zealand's mainland islands are operated by the Department of Conservation, a government agency, provides a degree of continuity lacking elsewhere.

Owing to a lack of reported successful instances of control of invasive species in mainland areas, and to a few well publicised failures (e.g. Sheail, 2003; Santulli, et al., 2014), managers have little guidance as to the circumstances under which a mainland island approach might prove successful. Of particular interest is how complex institutional and funding environments need to be navigated when planning long term control of invasives. In the UK, for instance, protected areas are largely privately-owned; conservation legislation incentivises rather than mandates conservation management activities; a significant proportion of conservation action is initiated in a bottom up fashion by non-governmental organisation or local communities (often enabled by government agencies); and funding for projects rarely exceeds 3–5 years in duration.

In this paper, we present an account of the development of a mainland island invasive control effort that grew in spatial extent over 15 years from a localised community-led effort to operate on a vast scale (29,000 km^2) in the north of Scotland. It progressed from pilot, to demonstration

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and, eventually, mainstreaming stages without secured long-term funding but as an enduring partnership between academic researchers and practitioners under an adaptive management framework.

Invasive American mink threatening Ratty the water vole, a British cultural icon

The initial motivation for the project was the protection of the water vole (*Arvicola amphibius*), riparian rodents that used to be very abundant in Britain but that experienced a cumulative mean loss of occupied sites of 98.7% across all regions of England, Scotland, and Wales by 1998 from the 1939 baseline (Moorhouse, et al., 2015). Thus, the water vole was included amongst Species Action Plans and devolved Local Biodiversity Action Plans (LBAPs) when the UK Government launched those plans for the recovery of threatened species and habitats as part of the UK Biodiversity Action Plan in response to the Convention on Biological Diversity in 1994 (UK Biodiversity Partnership, 1995; JNCC, 2006). One of several suggested causes for the catastrophic decline of the water vole was the American mink *(Neovison vison)* that had invaded all but the north-westernmost corner of the UK following historical escapes from fur farms (Fraser, et al., 2015). Its overriding influence became clearer over time (Aars, et al., 2001; Moorhouse, et al., 2015). Accordingly, LBAPs included controlling mink, but little guidance or prescriptions on how this should be implemented were included.

SCALING UP MINK CONTROL: FOUR PHASES OF SPATIAL EXPANSION

Water voles in the catchment of the River Ythan (1995–2007): 100–644 km²

Research into metapopulation processes by ecologists from the University of Aberdeen funded by the Natural Environment Research Council (NERC) (1995) identified a handful of highly fragmented remnant water vole populations in a 100 km² portion of an intensely farmed lowland area north of Aberdeen in NE Scotland (Telfer, et al., 2001) (Fig. 1). Structured surveys revealed that water voles had become regionally scarce or absent where they were once common (Lambin, et al., 1996; Lambin, et al., 1998; Lambin, et al., 2002). The intensively studied metapopulation network was gradually shrinking under the influence of American mink predation, causing the extinction of multiple adjacent colonies (Lambin, et al., 1996; Telfer, et al., 2001) (Table 1).

Fig. 1 Map of northern Scotland showing the five stages of expansion of successive mink control projects from a sub-catchment of the River Ythan (thick black line, numbered 1), the entire catchment of the River Ythan (Grey dashed lined, numbered 2), the Cairngorms National Park (thin black line, numbered 3), the area of the expanded Cairngorms project (dashed black line, numbered 4) and the area where the Scottish Mink Initiative operated (Continuous thick black line, numbered 5).

With funding secured by north-east Scotland's Local Biodiversity Action Plan group from Scottish Natural Heritage (SNH), the government agency tasked with promoting, caring for and improving Scotland's natural heritage (£145,000 over eight years, Fig. 2), the first stage of the northern Scotland control mink project was initiated in 2002. Its modest objective was to safeguard the remnant lowland water vole metapopulations by preventing further encroachment by mink. Initially, a member of staff from the local Ythan District Fishery Board, a statutory body empowered to protect, enhance and conserve Atlantic salmon and sea trout within the Ythan catchment, was employed on a part time basis to control mink (2002–2003). Subsequently, a full-time member of staff, employed by the University of Aberdeen (UoA), was appointed over five consecutive one-year contracts (2003–2007) as mink control activities were extended to the entire 644 km² area of the catchment of the river Ythan as evidence accumulated that it was possible to protect remnant water vole colonies from encroachment by mink (Fig. 1).

This early step was arguably an instance of last ditch conservation, focussed on safeguarding a fast-shrinking isolated remnant water vole metapopulation. It was nevertheless influential in shaping ways of working that became crucial as the project area was expanded 45-fold over the next 10 years.

Key features were:

i) Close links between research on water vole and mink population dynamics and conservation delivery;

ii) Systematic deployment of mink rafts that make it possible to detect the presence of mink and to target cage trapping to those sections of waterways where current mink presence is confirmed (Reynolds, et al., 2004);

iii) Involvement of local residents who were encouraged to volunteer to monitor and report the appearance of signs of mink on mink rafts in their neighbourhoods, allowing a single project officer to effectively control mink of an entire catchment through targeted trapping.

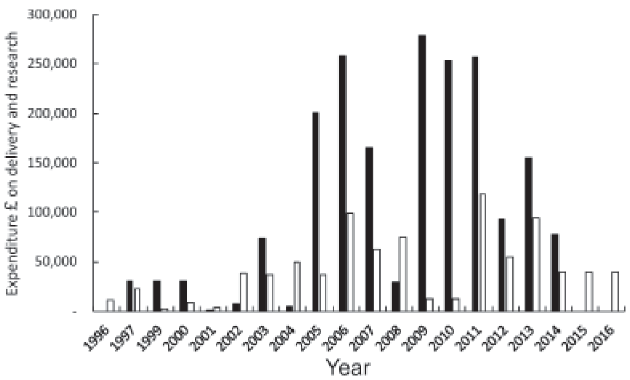

Fig. 2 Annualised expenditure of all projects relevant to water vole conservation and mink control broken down as funding for enabling or evaluating research (white bars) or conservation delivery (black bars).

iv) Partnership with organisations tasked with the management, conservation and enhancement of native freshwater fish and their environments in Scotland and increasingly involved in invasive species management.

In 2009, the river trust in the adjoining catchment of the River Deveron, emulated the project and obtained funding from SNH for an integrated package of invasives control, including American mink. The likely disappearance of the Ythan water vole population was averted, and this population is now thriving and extends across the entire lowland NE Aberdeenshire plain (W Morgan, E McHenry, X Lambin unpublished data).

The Cairngorms Water Vole Conservation Project (2007–2009): 5,500–10,570 km^2

Further surveys of water voles in the uplands of NE Scotland commissioned by SNH and research into metapopulation genetics processes by UoA (1998–2000) uncovered large water vole metapopulation networks in the area that was to become the Cairngorms National Park (CNP) in 2003 (Aars, et al., 2001; Lambin, et al., 1998; WildCRU, 2004) (Table 1). These populations, while in slow decline, had not yet been affected by the American mink invasion to the same extent as lowland populations, owing to the low density of alternative prey for mink in the uplands (Oliver, et al., 2009). They presented the opportunity to preserve functioning metapopulations and the associated ecosystem functions arising from the ecosystem engineering activities of water voles on upland riparian vegetation (Bryce, et al., 2013) as opposed to the more desperate task of rescuing critically endangered survivors.

The CNP encompasses a mountain massif, dominated by heather moorland where shooting of red deer (*Cervus elaphus*), red grouse (*Lagopus lagopus*) and fly fishing of salmon (*Salmo salar*) provide much needed income to the rural economy. In order to make these leisure activities possible, a large number of game keepers and fishing ghillies are employed to intensively manage heather moorland through rotational burning, killing predators of grouse and accompanying anglers. These individuals were recognised as a potential trained workforce that already culled ~ 60–70 mink annually in CNP, hence had the expertise and a professional interest in the issue. Their willingness to step up and coordinate hitherto patchy mink control was ascertained through consultation funded by the newly established CNP in 2004. Thus, we reasoned that the CNP was a potential defensible mainland island stronghold for water voles where mink control could be sustained in perpetuity.

Funding bids to the Tubney Charitable Trust, a charitable organisation for projects that conserve the natural environment in the UK, and to the NERC outlining the ambition to implement a formal active adaptive management approach to control American mink on the large scale of CNP (encompassing 5,500km^2, Fig. 1) were prepared. SNH had again committed match funding should either bid succeed. The Cairngorms National Park Authority (CNPA) and three river trusts managing important salmon rivers flowing from the Cairngorms (River Dee Trust, Spey Foundation and Deveron, Bogie & Isla Rivers Charitable Trust) were also formal partners committing in-kind staff time. Both bids were funded and substantial funding was in place for three years (2006–2009), facilitating the employment of three project officers and one postdoctoral research fellow by UoA.

A detailed account of the project's approach and achievements is given in Bryce, et al. (2011) and Oliver, et al. (2016) and a brief summary only is given here. The approach was to deploy mink rafts with an approximate spacing of 2 km in a 'rolling carpet' fashion to first remove mink from upland areas and subsequently expand coverage

Table 1 Sequence and main findings of research at the University of Aberdeen that enabled the next step of mink control efforts by characterising the system to be managed, that evaluated the achievements of mink control efforts or that provided a strategic evaluation of different ways or working.

Research Project	Years	Scope	Main finding	Funder
S. Telfer PhD	1996–1999	Enabling	Water voles metapopulation processes are disrupted by mink causing spatially correlated colony extinction	UK Research Council
J. Luque Larena Postdoc fellowship	2003–2004	Enabling	Cairngorms Mountains are invaded by mink owing to presence of rabbits in abandoned hill farms	European Union
A. Zalewski Postdoc fellowship	2005–2006	Enabling	Cairngorms Mountains are a partial obstacle to mink dispersal but mink circumvent hills and nevertheless spread	European Union
M.K. Oliver R. Bryce Postdoc fellowships	2006–2009	Evaluation	Strong lowland–highland source–sink dynamics and high mobility between catchments influencing capture rates	UK Research Council
E. Fraser PhD	2010–2013	Enabling	Mink spread in sparsely populated coastal areas is heavily constrained by topography and boat-based ecotourism operators are potential volunteers	SNH
M.K. Oliver Postdoc fellowship	2010	Evaluation	Mink control reduces captures to almost zero in three years. Mink dispersal large-scale (31 km for females), male biased, and links adjacent river catchments	UK Research Council
Y. Melero Postdoc fellowship	2011–2014	Evaluation	No evidence of mating failure at low density causing Allee effect but instead compensatory increase in fecundity at low density	European Union
E. McHenry PhD	2014–2018	Strategic	Doing more with less: optimising investment in detection and control	UK Charity
W. Morgan PhD	2014–2018	Evaluation	Patterns of recovery in water voles	UoA

downstream to remove mink from an increasingly large area, hence protecting the uplands with increasing depth; a version of the 'remove and protect model' with depth (Bell, et al., 2019). The systematic use of mink rafts was made possible by the work of 208 volunteers. We sought volunteers willing to adopt a mink raft and report to a project officer or trapper in their local community whether a mink was present. Only when fresh mink signs were detected was a cage live trap set, hence minimising the time wasted checking empty traps at least once every 24 hours as mandated by law. If a mink was caught (as occurred following 10–22% of detections according to season), it was humanely killed before the raft was returned to monitoring mode. The project officer played a crucial role in coordinating the efforts of volunteers, not all of whom were equipped or qualified to humanely despatch a mink. On detection of the presence of a mink by a volunteer, the full capabilities of the larger volunteer force could be called upon to effectively trap and despatch the mink.

Project officers sought permission to access the land and deploy a mink raft and then recruited local volunteers to operate the raft. Game-keepers who are licenced to carry fire-arms were partners of choice to adopt and operate mink rafts, although it proved difficult to dissuade them from their traditional practice of deploying traps irrespective of evidence of the presence of the focal species (Fig. 3). Two of three project officers had prior family or professional associations with the local game keeping community and this undoubtedly facilitated building constructive relationships. The adoption of rafts by local residents was key to allowing project officers to deploy further rafts downstream in the more biologically productive parts of the CNP and where landownership is more fragmented and residents with a wider diversity of professions live. Here, we adopted a functional approach to participation (Pretty, 1995) whereby local people were co-opted to meet the predetermined objectives of achieving coordinated mink control. Thus, recruitment of volunteers to operate rafts was targeted toward individuals with an interest in nature conservation and natural resource management, such as forest or local government rangers, fishing ghillies, bailiffs, nature reserve managers, but also included numerous local residents made aware of opportunities to become involved in the project through community talks and publication in the local press. Where required, project officers would check mink traps or despatch mink themselves but volunteers were always encouraged to step up their involvement from monitoring rafts only, to trapping or becoming a trained despatcher.

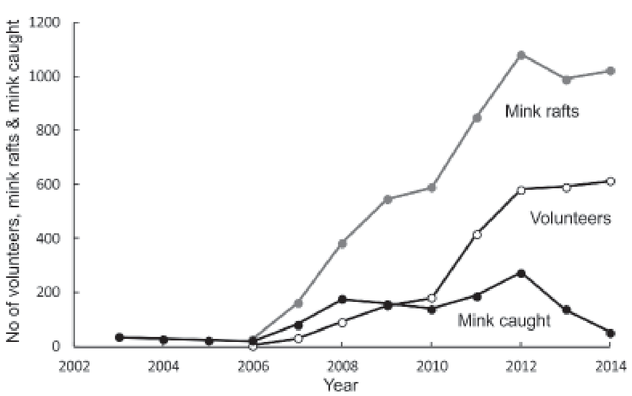

Fig. 3 Temporal dynamics of the number of mink caught per year (black line, black circles), the number of mink rafts deployed (grey line, grey circles) and the number of volunteers contributing to the projects (black line, white circles).

The large project area was subdivided into sub-catchment management units encompassing major tributaries of main catchments (median size: 55 km^2). Analyses of the impact of culling on the population used, as a reference point, the time when mink raft deployment was deemed complete in a sub-catchment by the local project officer. The number of mink captured per km of waterway decreased from an average of 0.16 to 0.06 to 0.01 for sub-catchments in the first, second and third years after inception of control, respectively. This was despite higher fecundity amongst mink that had survived culling (Melero, et al., 2015; Oliver, et al., 2016). Most mink caught in the third year after inception of control were males, reflecting their high propensity to disperse from the natal area. This was also reflected in the high proportion of juvenile males amongst the few mink caught in the higher elevations of the CNP which were cleared of mink by the end of 2007. No mink at all were caught in 3,417 km^2 of montane and moorlands areas of CNP but 376 mink were removed from 5,381 km^2 covering moorland and pastoral areas of lower altitude. There was further evidence of high mink mobility within and between river catchments resulting in compensatory immigration, as mink capture rate in a sub-catchment increased with connectivity to mink still present in other sub-catchments (Bryce, et al., 2011; Oliver, et al., 2016).

The key lessons from the ongoing evaluation of management efforts were:

i) The presence of large-scale lowland-highland source-sink dynamics in mink such that most mink impacting upland biodiversity had dispersed from more productive lowland areas. This motivated a change in the scope of the project when the management group endorsed downstream expansion from 5,500 to 10,570 km^2 at the end of the second year of the project (2007) so as to deplete mink where most were born (Fig. 1).

ii) Deploying a large number of mink rafts and recruiting volunteers is a gradual process and a pool of volunteers must be replenished to make up for volunteer turn-over (Beirne & Lambin, 2013). Different communities and river trusts vary in their ability to embrace conservation volunteering and the resulting asynchrony in the inception of mink control delayed region-wide eradication.

iii) Mink disperse widely and dispersal connects major river catchments, implying an inter-dependence between river catchments and the organisations that manage them. Thus, high mobility of mink dictates that control should be on a very large scale so as to avoid the effects of compensatory immigration.

The Scottish Mink Initiative (2011–2015): 10,570 –29,000 km2

The achievement of the Cairngorms Water Vole Conservation Project elicited much enthusiasm from volunteers who had been part of a rare conservation good news story, as well as from private and public land managers (e.g. CNPA) and Scottish Natural Heritage. As the three-year funding period was coming to an end, there was a real risk that the project would fall from a funding cliff edge such that not only would all biological gains be lost but the volunteer community would become despondent if abandoned. SNH had also been working with the Scottish Wildlife Trust (SWT) and local fisheries' trusts in the north-west Highlands to remove mink in that area, so there was an opportunity to develop a more strategic approach to mink control across the north of Scotland by amalgamating and expanding the various projects into a single, much larger scheme. SNH, along with two other key funders (CNPA and the Tubney Trust) expressed their willingness to renew their funding commitments for a further three years (£478,000; £8,932; £100,000, respectively). However, the partnership research grant scheme run by the UK research council had been discontinued and funding commitments did not include the overheads universities expect from research grants. This made it impossible for UoA to continue as the lead partner of what was increasingly an ambitious conservation delivery project rather than a combination of this and research. Furthermore, it was evident that local organisations managing common natural resources and representing private entities gaining economic benefits from harvesting salmon would be more appropriate long-term custodians of a mainland island project than a university and thereby ensure it had a long-term legacy.

Accordingly, a new partnership was formed involving Rivers and Fisheries Trusts of Scotland (RAFTS) and SWT. RAFTS was a charity with a formal objective comprising "the conservation and enhancement of native freshwater fish and their environments in Scotland". Twenty-six river trusts and foundations were members of RAFTS and it was already actively involved in (mostly riparian plant) invasive management. It had a strong track record in fundraising and project management for its members. It proved to be the ideal body to lead an expanded project and to ensure coordinated action using best practice by its member river trusts at a scale commensurate with the biological challenge posed by mink. Nine river trusts in northern and north-east Scotland were enlisted in a new partnership and they committed in kind resources to removing mink from their river catchments. The renewed funding commitments were critical in allowing an application to the EU-funded LEADER scheme operated by the Scottish Government. The aim of LEADER is to increase support to local rural community and business networks to build knowledge and skills, and encourage innovation and cooperation, in order to tackle local development objectives. A competitive application involving multiple local areas was assembled and further funds (£229,000 from LEADER, and £14,000 from river trusts) were secured, facilitating the appointment of three project officers and a coordinator employed by RAFTS. For the second time, mink control efforts in northern Scotland bounced back from a financial cliff edge.

Owing to the time required for the evaluation of the funding bid and recruiting new project staff, mink volunteers had been left without support or certainty on the future of mink control efforts during the 19-month gap that elapsed between the end of the Cairngorms project in October 2009 and the start of the new Scottish Mink Initiative (SMI) in April 2011. Over that period, a skeleton staff was retained from previous projects to maintain the volunteer and associated mink raft network prior to further expansion (Raynor, et al., 2016). This included one part-time member of staff from the north-west Highlands project. It had adopted a "*cordon sanitaire*" approach, comprising a double line of mink rafts intended to prevent mink from invading northern Scotland, following from recommendations in an unpublished report to SNH (Harrington, et al., 2008). That approach turned out to be flawed owing to mink dispersal abilities, evident in data collected as part of the Cairngorms project but that were unpublished at that time (Oliver, et al., 2016), and to the importance of the coastal environment in driving invasion range expansion (Fraser, et al., 2015).

Four newly appointed SMI staff had to be trained and build new trust relationships with volunteers previously supported by other staff. While some volunteers had continued with their activities in the intervening time and caught a minimum of 139 mink in 2010, many no doubt concluded that the project had come to an abrupt end and ceased their activities. This led to reinvasion of some of the project areas, especially in the vicinity of the crucial catchment of the River Don where inadequate local support had prevented progress with mink control as part of the Cairngorms project (contrast figure 2 in Oliver, et al., (2016) and figure 3 in Melero, et al., (2015)).

Once the full complement of project officers was again embedded in the local community and supported by local river trusts, the approach refined in the previous project was scaled up substantially by SMI resulting in 837 volunteers operating up to 1022 rafts and removing a minimum of 646 mink between 2011 and 2014. This resulted in a vast area encompassing ~29,000 km^2 bounded by seas becoming free of breeding mink as determined by the absence of footprints on mink detection rafts, the metric chosen by the steering group to gauge the effectiveness of the project (Fig. 1), hence increasing our ability to deal with the constant flux of mink moving up from the south. Mink were regularly detected in the southern and western edges of the project area (51 in 2014) especially, reflecting primarily immigration by males during the rut period. A more detailed account of its achievements and of some of the challenges encountered is found in Raynor, et al. (2016).

The Transition to Scotland's Invasive species Initiative (2018-2022): 29,500 km^2

One ultimate objective of SMI was to engender a sense of ownership of the mink management and wider biosecurity, considering the threat posed by aquatic invasives such as the salmon fluke (*Gyrodactylus salaris*) and the giant hogweed (*Heracleum mantegazzianum*), amongst the local fisheries trusts as appropriate to any mainland island project. There was also an aspiration to further build on the partnership by involving more and more trusts, as resources allowed. Thus, during the second half of the funded period (September 2013–August 2015), there was a process of hand-over of all local processes to 10 local rivers and fisheries trusts. This included transfer of responsibility for managing existing networks of volunteers and mink rafts, including all access agreements with land owners, health and safety and standard operating protocols, and all relevant databases. A project coordinator remained employed by RAFTS and each participating river trust received payment to cover costs incurred in undertaking a combination of mink raft checking and maintenance, as well as data collection and support and coordination for the local volunteer network.

Two main limitations to the effectiveness of the handover have been: 1) not all areas of high mink productivity on the lowland coastal plain in the extreme corner of NE Scotland have sufficient salmon resources to maintain functional river trusts. Without additional resources, such areas could again become a source of dispersing mink into adjacent better controlled areas; 2) maintaining mink raft coverage in remote areas of north-west Scotland, where the low human population density; a predominance of red deer over grouse as the primary game species; difficult topography including many coastal islands; and a limited road network all placed significant restrictions on the ability to maintain required coverage for surveillance. The handover arrangements have been severely tested, with mixed results, by the absence of any financial support to any of the trusts between August 2015 and November 2017. During this period, a major reform of freshwater fisheries governance that would have led to river trusts and boards being disbanded was mooted by the Scottish Government and this precluded the submission of grant applications for the successor project by RAFTS.

The proposed reform was ultimately abandoned but led to the demise of RAFTS as an organisation. Scottish Natural Heritage, a key long-term supporter of the project from its very outset, stepped in as lead partner for an application to the Heritage Lottery Fund and an award of £1.59M was announced in August 2017. Thus, after a protracted period without secure funding, a successor to SMI, centred on applying the citizen conservationist approach to a suite of riparian invasives and prepared by RAFTS, will operate from 2018–2022. The new project, the Scottish Invasive Species Initiative, will tackle the challenge of reviving the volunteer network and undoing unavoidable partial reinvasion of the project area for another four years and further increase engagement in invasive management by local communities (Horrill, et al., 2019).

DISCUSSION

Over 15 years, a vast mainland island area has been established in northern Scotland that protects native riparian biodiversity including water voles from the destructive influence of the invasive American mink. The endeavour is the outcome of a succession of research and implementation projects conducted in partnership that optimised mink control effort so they could be scaled-up. Implementation projects progressed from a small-scale pilot phase (in the Ythan), to a two-stage demonstration phase, first evidencing the feasibility of scaling up mink raft deployment and enthusing volunteers to become citizen conservationists (the Cairngorms project) and then, the feasibility of devolving management of such a large scale project to local organisations engaged in natural resource management (the SMI) according to a wider, more strategic framework. The later stage of SMI was the beginning of embedding mink control within the activities of rivers trusts working autonomously but in a coordinated manner. The most recently funded successor project has the ambition to extend the approach refined with mink to a suite of containable riparian plant invasives that are widespread in Scotland.

Long-term invasive species management was achieved despite short-term funding as a result of a succession of fixed-length short-term discrete projects each of three to four years duration, rather than the result of any integrated long-term joined-up endeavour underpinned by secured funding or any strategic decision on the size of any area where mink could be controlled on Scotland's mainland. As the feasibility of controlling mink on a large scale was demonstrated and the endeavour's spatial ambition grew, the very existence of the project was in jeopardy on multiple occasions and some of its achievements were eroded during four gaps between funding cycles. Its future is secured for another four years after the latest two and a half year funding gap since the end of SMI. Although the large spatial reach of the project, its cost effectiveness and hence attractiveness, results from the use of volunteer citizen conservationists, the lack of continuity in funding has been highly detrimental to the trust relationship built between the project and volunteers giving their time freely for conservation. Invasive species control in mainland areas is, by definition, an open-ended commitment and it is paramount the limited resourcing required to maintain what has been achieved should be in place conditionally on evidence of success and sustainability being presented.

The cumulative cost of all components of the project, including the research by EU-funded fellows and four PhDs that enabled the project or contributed to its evaluation under the adaptive management, was £2,800,000. The cost-effectiveness of the project resulted from the use of a workforce of 866 unpaid "citizen-conservationist" volunteers. Based on the assumption that their time contribution amounted to 30 min/2weeks = 13 hours per year per volunteer, the total 2,652 volunteer years is equivalent to 21.6 standard person years, crudely valued at £1,404,00 using the assumptions of Robertson, et al. (2019). Arguably, the value of their contribution is greater still because of the increased awareness of the issues caused by invasives and community cohesion benefits (Evely, et al., 2011).

Although the volunteer approach is relatively cheap, it is not cost-free as volunteers require a degree of support, encouragement, information and re-supplying by project staff. The successive incarnations of the mainland mink control efforts have involved an increasing number of volunteers (peaking at 612 in 2014 Fig. 3) supported by a fixed and small number of project officers. Volunteer retention over time is less than 100 % such that it is constantly necessary to recruit new volunteers. Despite project staff consistently reinforcing the message that "no mink is good news", it remains that the enduring absence of mink on a volunteer's raft contributes to some volunteers dropping out (Beirne & Lambin, 2013). The greatest risk causing volunteer drop-out is the perception that the project has come to an untimely end in the absence of communication from project staff, as arose during the funding gaps, even if efforts to fund-raise for a successor project are underway.

SNH, Scotland's governmental organisation responsible for the management of natural heritage including the threat posed by invasive species, has been an enduring and crucial funder at all stages of mainland mink endeavour ever since 1995. It contributed 45 % of the total £2,803,950 cash cost over 21 years and 62% of the subset (£1,900,000) spent on conservation delivery. SNH is also the main funder of the Hebridean Mink Project (Macleod, et al., 2019), hence is committing substantial resources to managing American mink. However, as with all government agencies, including in New Zealand and the USA, it is constrained by its inability to commit long-term funding for managing established invasive species. Even SNH's Species Action Framework scheme that made sizeable financial contributions to SMI (£710,000 including extensions) was a five-year programme of targeted species management. Furthermore, contributions from SNH were conditional on funding being secured from other funders. Fund-raising by UoA and RAFTS was successful but time-consuming and added complexity to project management and reporting. It is a major concern that given EU funds covered 20 % of total costs and provided for 40% of the research work, the departure of the UK from the EU in 2019 will potentially leave a major hole in funding.

Through all phases of the project, the programmes of research that enabled and evaluated the development of large scale invasive control were always funded by separate funding streams to those used for conservation delivery (such as species recovery or habitat management). This was in response to implicit or explicit indications that while funders of conservation delivery like the sound of adaptive management, they are less keen to pay for it. The modicum of adaptive management achieved resulted largely from universities having access to lots of (predominantly) young, enthusiastic people keen to gain qualifications in conservation through applied research. For adaptive management to be a reality and not just an aspiration, there is a clear need for more integrated (research–management) funding streams delivering vital continuity of support.

Our work demonstrates there is no technical difficulty in expanding working with citizen conservationists for pushing back huge scale invasion. Partnerships and relationships had a critical role in achieving this work (across all project phases). The outcomes have been achieved through those networks and the empowerment of volunteers and interested/affected 'stakeholders', all in spite of repeated uncertainty of funding. There is little doubt that even more could have been achieved had continuous funding been in place. Indeed, short-term funding is a major impediment to efficiency and increases the overall cost of long-term invasive control as lost ground must be recovered. Repeated gaps in funding, associated staff turnover and re-badging of projects are all damaging to the trust relationships built with volunteers. It makes no economic sense to embrace long-term control of invasives without funding it. Scotland, like other countries, needs a long-term stream of funding if it is going to manage invasive species. Thus, the future will tell whether our efforts were bold and trail blazing or overly ambitious and ultimately wasted if the SMI's ambition to become embedded within local management practice in perpetuity is not borne out.

REFERENCES

Aars, J., Lambin, X., Denny, R. and Griffin, A.C. (2001). 'Water vole in the Scottish uplands: distribution patterns of disturbed and pristine populations ahead and behind the American mink invasion front'. *Animal Conservation* 4: 187–194.

Beirne, C. and Lambin, X. (2013). 'Understanding the determinants of volunteer retention through capture-recapture analysis: Answering social science questions using a wildlife ecology toolkit'. *Conservation Letters* 6: 391–401.

Bell, P., Nathan, H. and Mulgan, N. (2019) 'Island eradication within large landscapes: the remove and protect model'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 604–610. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Bomford, M. and O'Brien, P. (1995). 'Eradication or control for vertebrate pests.'. *Wildlife Society Bulletin* 23: 249–255.

Bryce, R., Oliver, M.K., Davies, L., Gray, H., Urquhart, J. and Lambin, X. (2011). 'Turning back the tide of American mink invasion at an unprecedented scale through community participation and adaptive management'. *Biological Conservation* 144: 575–583.

Bryce, R., van der Wal, R., Mitchell, R. and Lambin, X. (2013). 'Metapopulation dynamics of a burrowing herbivore drive spatio-temporal dynamics of riparian plant communities'. *Ecosystems* 16: 1165–1177.

DIISE (2015). The Database of Island Invasive Species Eradications, developed by Island Conservation, Coastal Conservation Action Laboratory UCSC, IUCN SSC Invasive Species Specialist Group, University of Auckland and Landcare Research New Zealand. <http://diise.islandconservation.org>.

Evely, A.C., Pinard, M., Reed, M.S. and Fazey, I. (2011). 'High levels of participation in conservation projects enhance learning'. *Conservation Letters* 4: 116–126.

Fraser, E.J., Lambin, X., Travis, J.M.J., Harrington, L. A., Palmer, S.C.F., Bocedi, G. and Macdonald, D.W. (2015). 'Range expansion of an invasive species through a heterogeneous landscape—the case of American mink in Scotland'. *Diversity and Distribution* 21: 888–900.

Glen, A.S., Pech, R.P. and Byrom, A.E. (2013). 'Connectivity and invasive species management: towards an integrated landscape approach'. *Biological Invasions* 15: 2127–2138.

Harrington, L., Hughes, J. and Macdonald, D. (2008). *Management of American Mink in the Northern Highlands: A Proposed Cordon Sanitaire Approach.* Unpublished report to SNH, the Environment Agency, the People's Trust for Endangered Species and the European Nature Trust. Oxford.

Horrill, C., Oliver, M.K. and Stubbs Partridge, J. (2019) 'Lessons on effectiveness and long-term prevention from broad-scale control of invasive alien species in Scotland's rivers and lochs'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 458–465. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

JNCC (2006). *UK Biodiversity Action Plan*. Peterborough: Department for Environment, Food and Rural Affairs.

Lambin, X., Telfer, S., Cosgrove, P. and Alexander, G. (1996). *Survey of Water Voles and Mink on the Rivers Don and Ythan.* Report to Scottish Natural Heritage. ref: 70/96/F1A/007.

Lambin, X., Fazey, I., Sansom, J., Dallas, J., Stewart, W., Piertney, S.B., Palmer, S.C.F., Bacon, P.J. and Webb, A. (1998). '*Aberdeenshire Water Vole Survey: The Distribution of Isolated Water Vole Populations in the Upper Catchments of the Rivers Dee and Don*.' Report to Scottish Natural Heritage ref: C/LF1/BAT/97/2.

Lambin, X., Graziani, G. and Hon, J. (2002). *Water Vole Survey in the Deveron and Ythan Catchments 2002*. Unpublished report to Aberdeen, north-east Scotland LBAP Partnership and SNH.

Macleod, I.A., Maclennan, D., Withaker, S., Thompson, D.B.A., Raynor, R. and Chaffer, R. (2019) 'Large scale eradication of American mink, *Neovison vison*, from the Outer Hebrides of Scotland'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 261–266. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Melero, Y., Robinson, E. and Lambin, X. (2015). 'Density-and age-dependent reproduction partially compensates culling efforts of invasive non-native American mink'. *Biological Invasions* 17: 1–13.

Moorhouse, T.P., Macdonald, D.W., Strachan, R. and Lambin, X. (2015). 'What Does Conservation Research Do, When Should it Stop, and What Do we Do Then? Questions Answered with Water Voles'. In: D.W. Macdonald and R.E. Feber (eds.) *Wildlife Conservation on Farmland Volume 1. Managing for Nature on Lowland Farms*, pp. 269–290. Oxford: Oxford University Press.

Oliver, M.K., Luque-Larena, J.J., and Lambin, X. (2009). 'Do rabbits eat voles? Apparent competition, habitat heterogeneity and large-scale coexistence under mink predation'. *Ecology Letters* 12: 1201–1209.

Oliver, M.K., Piertney, S.B., Zalewski, A., Melero, Y. and Lambin, X. (2016). 'The compensatory potential of increased immigration following intensive American mink population control is diluted by male-biased dispersal'. *Biological Invasions* 18: 3047–3061.

Pretty, J.N. (1995). 'Participatory learning for sustainable agriculture'. *World Development* 23: 1247–1263.

Raynor, R., Horrill, C., Taylor, M. and Taylor, K. (2016). 'American Mink. Version 1.0'. In: M.J. Gaywood, P.J. Boon, D.B.A. Thompson and I.M. Strachan (eds.) *The Species Action Framework Handbook*, pp. 287–313. Battleby, Perth: Scottish Natural Heritage.

Reynolds, J.C., Short, M.J. and Leigh, R.J. (2004). 'Development of population control strategies for mink *Mustela vison*, using floating rafts as monitors and trap sites'. *Biolological Conservation* 120: 533–543.

Robertson, P., Mill, A., Shirley, M., Roy, S., Adriaens, T., Ward, A., Tatayah, V., and Booy, O. (2019) 'Invasive species removals and scale – contrasting island and mainland experience'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 687–691. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Santulli, G., Palazón, S., Melero, Y., Gosálbez, J. and Lambin, X. (2014). 'Multi-season occupancy analysis reveals large scale competitive exclusion of the critically endangered European mink by the invasive non-native American mink in Spain'. *Biological Conservation* 176: 21–29.

Saunders, A. and Norton, D. (2001). 'Ecological restoration at mainland islands in New Zealand'. *Biolological Conservation* 99: 109–119.

Sheail, J. (2003). 'Government and the management of an alien pest species: A British perspective'. *Landscape Research* 28: 101–111.

Telfer, S., Holt, A., Donaldson, R. and Lambin, X. (2001). 'Metapopulation processes and persistence in remnant water vole populations'. *Oikos* 95: 31–42.

UK Biodiversity Partnership (1995). *Biodiversity: the UK Steering Group Report. Volume 2 : Action Plans*. London: HMSO.

WildCRU, Oxford University (2004). *Water vole surveys in Fife and Central Cairngorms*. Scottish Natural Heritage Commissioned Report No. 058 (ROAME No. F01AC316).

Battling invasive species in the Pacific

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Abstract This paper is a snapshot of how Pacific invasive species battlers are protecting their islands with the assistance of the Global Environment Facility's Pacific Alliance for Sustainability (GEF-PAS) project "Prevention, control and management of invasive alien species in the Pacific". The aims of the project are presented, along with examples of how implementation increased awareness and capacity, and management of invasive plants and animals, throughout the Pacific. Over 100 IAS activities took place across nine countries between 2011 and 2016. The project, one of the largest investments in invasive species management in Pacific history, has raised the benchmark of invasive species management in the Pacific and enhanced regional mechanisms. Hopefully the people within this story inspire and assist other battlers to join the fight and protect our islands from invasive species.

Keywords: Cook Islands, Federated States of Micronesia, GEF-PAS Project, Kiribati, Niue, Palau, regional project, Republic of the Marshall Islands, Samoa, SPREP, Tonga, Vanuatu

INTRODUCTION

The Pacific region is populated by diverse people and spans a third of the earth's surface and encompasses about half of the global sea surface (Fig. 1). There are ca. 2,000 different languages and ca. 30,000 islands. Pacific ecosystems are one of the world's biodiversity hotspots, with a large number of species found only in the Pacific and nowhere else. There are 2,189 single-country endemic species recorded to date. Of these species, 5.8% are already extinct or exist only in captivity. A further 45% are at risk of extinction (SPREP, 2013). The region faces some of the highest extinction rates in the world. The largest cause of extinction of single-country endemic species in the Pacific is the impact of invasive alien species (IAS). Invasive alien species also severely impact economies, ability to trade, sustainable development, health, ecosystem services, and the resilience of ecosystems to respond to natural disasters. Fortunately, we can do something about it. Even in this diverse region, many things are shared in common. The people are self-reliant, rely heavily on their environment to support their livelihoods and share many common IAS issues as they are ultimately connected. Sharing what is learnt regionally benefits the people and their families economically, culturally, and in their daily lives. The 2013 State of Conservation in the Oceania assessment (SPREP, 2013) showed that IAS are the most important driver of species loss in the region and contribute directly to the loss of ecosystem function and loss of resilience, and ability to respond to climate change threats. Invasive alien species also severely impact Pacific economies, ability to trade, sustainable development, health, ecosystem services, and the resilience of ecosystems to respond to natural disasters. The status of the IAS issue in the Pacific is "poor" according to the report on the State of Conservation in Oceania (SPREP, 2013).

Fig. 1 Island nations and groups in the Pacific. Island locations and sizes are not to scale.

In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) (2019). Island invasives: scaling up to meet the challenge, pp. 658–662. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

The "Guidelines for invasive species management in the Pacific: a Pacific strategy for managing pests, weeds and other invasive species" (SPREP, 2009) provide a comprehensive framework for the Pacific Region to respond to IAS at the regional and national levels, endorsed in 2009 by members of both the Secretariat of the Pacific Regional Environmental Programme (SPREP), and the Pacific Community (SPC). This framework is used throughout the Pacific for structuring the National Invasive Species Strategies and Action Plan (NISSAP) and the Territorial Invasive Species Strategies and Action Plan (TISSAP).

The Guidelines were implemented to achieve the objective of reducing the environmental, economic, and human health impacts of IAS in both terrestrial and marine habitats in the Pacific region. The classification of the IAS management themes within the Guidelines allows current and future IAS management activity and success to be measured both nationally and regionally and enables the identification of gaps which need to be addressed.

The project "Prevention, control and management of invasive alien species in the Pacific Islands" (GEF-PAS) was implemented by the United Nations Environment Programme (UNEP) and executed by the SPREP and national partner agencies from 2011 to 2016. The GEF-PAS project goal was to 'conserve ecosystems, species and genetic diversity in the Pacific Region'. The GEF-PAS project structure followed the 'Guidelines for Invasive Species Management in the Pacific' (SPREP, 2009) with three major components: (i) foundations; (ii) problem definition, prioritisation and decision making; and (iii) management action.

This regional approach has supported the establishment of the Pacific Invasive Species Guidelines Reporting Database, a database of national, territorial and regional progress in implementing the "Guidelines", with indicators on priority IAS initiatives. SPREP coordinates two Pacific IAS networks. The Pacific Invasives Partnership (PIP) is an umbrella group of IAS experts from organisations who work on IAS issues in more than one Pacific country. PIP is focused on coordinating IAS assistance in the Pacific region and aims to build cooperation among Pacific experts who provide assistance to Pacific countries and territories. SPREP also coordinates the Pacific Invasives Learning Network (PILN), a peer network of cross-sectoral IAS practitioners in the Pacific. The PILN aims to build cooperation between Pacific countries and territories on IAS issues. There are PILN teams in all but three of the 21 SPREP Pacific island member countries and territories.

A knowledge management system was initiated by the development of the Pacific Invasive Species Battler Series launched in 2016 with nine booklets to date focused on common IAS issues, based on Pacific examples and serving the Pacific Region. They are available from the Pacific Invasive Species Battler Resource Base (<https://piln.srep.org>), a searchable resource base providing the latest information on IAS issues, case studies, and introductory guides on common IAS issues. This resource is designed to increase the capacity of Pacific countries and territories in an effective and efficient manner, The "battler" brand has developed from Pacific IAS practitioners' internal/external communications over the years and serves as an ongoing reminder that if we don't achieve any change on the ground as practitioners we have been ineffective, it reminds us that IAS management is a long-term challenge that most of us will be working on for the rest of our lives and that as a regional collective we are not alone on our individual islands.

OUTCOMES OF THE GEF-PAS PROJECT

Ten Pacific Island countries originally participated in the GEF-PASIAS project: the Cook Islands, Federated States of Micronesia, Kiribati, Marshall Islands, Niue, Palau, Papua New Guinea, Samoa, Tonga and Vanuatu. These were reduced to nine countries with the withdrawal of Papua New Guinea following the mid-term review, due to issues related to the country's readiness to engage with the project. The project was therefore responsible for delivering support to a culturally, geographically and economically diverse set of Small Island Developing States (SIDS) spread across the vast geographical scope of the Pacific Ocean.

In-country subprojects and activities were facilitated by National Project Coordinators and overseen by national Invasive Species Coordinating Committees. The project's goal "to conserve ecosystems, species and genetic diversity in the Pacific region" is broad and aspirational and is backed by the objective "to reduce the environmental, economic, and human health impacts of invasive alien species in both terrestrial and marine habitats in the Pacific region". The project budget was US$7,010,890 consisting of US$3,031,818 in GEF funds and US$3,979,072 in SPREP and country co-financing. The project consists of five components, three of which can be described as core components which relate to the three major areas of work and nine thematic directions.

Component 1 – Foundations: generating support

This component addresses the limited understanding of the threats posed by IAS to the environment, economies, human health and cultural values of decision makers, the private sector and the general public. It aims to raise awareness across all sectors of society of the importance of IAS risks and impacts, and of the benefits of IAS management for biodiversity, the economy and human health. It also aims to actively support IAS management. With raised awareness, it is expected that sufficient resources will become available to enable all national and regional IAS priorities to be addressed and, most importantly, enable capacity building efforts to flourish alongside the development of supportive policy and legislation.

The three thematic directions addressed by Component 1 are:

Generating support— Raising awareness of the impacts of invasive species on biodiversity, the economy, human health and socio-cultural values, and generating support for action to manage and reduce them.

Building capacity— Developing the institutions, skills, infrastructure, technical support, information management, linkages, networks and exchanges required to manage invasive species effectively.

Legislation, policy and protocols— Ensuring that appropriate legislation, protocols, policies and procedures are in place and operating, to underpin the effective management of invasive species.

Component 2 – Problem definition, prioritisation and decision-making: baseline and monitoring

This component aims at addressing the chronic lack of information and data on IAS within the region which impacts on the ability of governments to define priorities, develop national strategies and establish supportive policies and legislation. It aims to ensure that information and data on IAS, their distribution and status is readily available to support informed decision making, strategic planning and effective management. Importantly, the component also aims to address the potential biosecurity and economic impacts of IAS through improved knowledge of trans-boundary movement and regional status of critical IAS.

The three thematic directions addressed by this Component are:

Baseline and monitoring— Establishing a baseline of information on the status and distribution of invasive species and a programme for detecting change, including range changes and emerging impacts.

Prioritisation— Establishing effective systems for assessing risk and prioritising invasive species for management.

Research on priorities— Understanding priority invasives, including species biology and impacts, and developing effective management techniques.

Component 3 – Management action

This component addresses the practical requirements of managing IAS. Until management action is implemented, no progress is made on addressing IAS. The three thematic directions addressed by this component are:

Biosecurity—Preventing the trans-boundary and inter-island movement of IAS in the region by encouraging the establishment of cost-effective biosecurity measures (e.g. rapid response protocols) aimed at reducing the need for costly post-invasion control measures. It aims to assist the establishment of effective systems throughout the Pacific to regulate introductions and to detect and manage unauthorised or accidental introductions across borders or to new islands within countries.

Management of established invasives—Reducing or eliminating the impacts of established invasive species, by eradication, containment, exclusion, or population reduction by physical, chemical or biological control.

Restoration—Restoring native biodiversity or ensuring recovery of other values, after invasive species management.

Each of the above three components dovetails directly with the priority thematic areas of the Guidelines which were developed as a result of an extensive regional stakeholder consultation process in 2007/2008. As such, they reinforce the rationale and justification for the IAS project and its legitimacy in the eyes of the regional IAS stakeholders and their international partners and networks. Together, the three components also address the IAS management weaknesses identified in the Guidelines.

Components 4 and 5 – Project management & monitoring and evaluation

These management-related components establish SPREP as the designated project Executing Agency and support a Project Facilitator and half-time Financial Manager for this purpose. SPREP funding covered the costs of the Project Manager. SPREP also had designated responsibility to ensure an effective monitoring and evaluation framework is established at inception. This role is consistent with SPREP's regional mandate and role to foster national and Pacific-wide strategies consistent with international best practices. SPREP is also able to engage the member organisations of the umbrella coordinating body the Pacific Invasives Partnership to further the goals of the project through provision of advice and the PIP members' own IAS management and capacity building interventions. The project activities strengthened capacity by improving IAS outreach, policies, laws, prevention and management. The project helped participating countries and others in the Pacific region to address existing and future biological invasions.

ACTIONS

More than 100 IAS activities took place across nine countries between 2011 and 2016 (SPREP, 2016). Here the scope and range of the purpose of actions undertaken and examples of those actions are highlighted.

Awareness raising and capacity building

Awareness of the impacts of IAS was increased at the local, governmental and political level. As an example, a royal visit to Toloa Rainforest by His Majesty King Tupo VI and Her Majesty Queen Nanasipau'u raised the profile of IAS management in Tonga. School scholarships were also presented by Her Majesty Queen Nanasipau'u to the top three Tupou College forest restoration team members at a national school prize-giving.

Awareness of IAS is important to create or support actions. For countries to take control of their responses to invasives, the first steps were to develop awareness in communities (local to national, and across a range of social roles), to mainstream IAS issues, to create or access long-term external funding mechanisms, and to generally increase the support for IAS issues. As an example, engaging posters were made by teams in Palau, Vanuatu, and the Cook Islands to communicate which species were invasive, what they affected, and boost the idea that individuals can take action. Outreach is a vital component of battling IAS because an educated, engaged community produces fast, effective action.

Capacity building of institutions, skills, infrastructure, technical support, information management, networks and exchanges required to manage IAS effectively were developed. Particularly given the strong customary land ownership in the Pacific, on-site management requires whole-of-community engagement, and the strong community ties in the Pacific are a strength for IAS early detection and rapid response. Local people with site knowledge and experience were integral to project implementation and benefited from learning new field techniques and scientific approaches, enhancing regional capacity. Direct engagement with field action makes local communities more likely to maintain site management, value their environment, and support or generate future conservation.

As examples, biosecurity training was provided in Kiribati, and a multi-country workshop was held in Samoa to support the prevention of IAS movements between islands. Training to detect and manage little fire ants was conducted in Vanuatu. A workshop on eradicating rodents from small islands was held in 2015, with participants from Kiribati, Republic of the Marshall Islands, Tonga, and Wallis and Futuna practising the eradication techniques on Malinoa and Motutapu islands in Tonga. The removal of the rats (*Rattus* spp.) has already boosted bird populations, such as the fuleheu or wattled honeyeater (*Foulehaio carunuculata*) and misi or Polynesian starlings (*Aplonis tabuensis*). Black-naped terns (*Sterna sumatrana)* were nesting and had eggs on the beaches of both islands.

The Pacific region is under-resourced regarding research capability and IAS, biodiversity, and ecosystem data. The limited resources available for IAS management demand that achievable goals are prioritised based on research and available data and that priorities meet the expectations of all stakeholders. Further, as Parties to international environmental agreements such as the Convention on Biodiversity, the region needs to show progress and success in meeting their obligations under these agreements.

Invasive plant species actions

Weed risk assessments can be costly and time-consuming, and vital information for assessments such as seed viability may not be known. Given the limited resources available to Pacific island countries and territories and the existence of almost 2,000 species with existing weed risk assessments for the Pacific, the most effective first step is to ensure that existing weed risk assessments are being used. This was a focus of the project and resulted in the Battler series publication "Find answers online to common invasive species questions".

Weed risk assessments contribute to prioritisation of target species, areas, and activities in combination with stakeholder consultation and local knowledge during NISSAP formulation. There are ongoing priority weed programmes operating in the Pacific which are showing success towards eradication. Widespread weeds can sometimes be targeted using biological rather than chemical or physical control: 36 natural enemies have established on 19 weed species in the Pacific. Since 1911, 17 countries and territories have deliberately released biological control agents on weeds in the Pacific. There are many opportunities for distributing existing agents further around the Pacific and opportunities to target new species.

Unlike biological control agents that were introduced to target invasive animals but were devastating to those islands to which they were introduced, such as the Indian mongoose for controlling snakes and the rosy wolf snail for controlling the giant African snail (Lissachatina fulica), the use of biological control to manage widespread weeds has been much more successful and much safer following standard international protocols such as host-specificity testing on other possible desirable plants. Internationally, 483 agents have been released with none resulting in unpredictable non-target effects (M. Day & L. Hayes, pers. comm.).

The development of biological control agents for weeds can be initially expensive. However, once agents are researched and located, they can provide an endless service of controlling invasive plants to a degree where their impact is greatly reduced. Further, once the initial agents are confirmed as effective, it is relatively cheap to move them to new countries or locations following any additional location-specific host-testing that may be required.

Vanuatu was fortunate to benefit from two new agents being developed for African tulip tree under a Landcare Research New Zealand project with the Cook Islands. The agents required minimal further host specificity testing and are due to be introduced to Vanuatu in the near future.

Palau has benefited from many years work on *Mikania micrantha*, which has a natural rust enemy already established in many countries. Accordingly, previous host specificity testing has been carried out extensively for many countries, leaving just one plant for which Palau was required to do tests on. The rust was introduced to Palau but was not successful for undetermined reasons. It is planned that the rust will be moved to some states within the Federated States of Micronesia, such as Yap, where *Mikania micrantha* is a serious weed. *Mikania micrantha* is located in 20 Pacific countries and territories; however, the rust agent has only been introduced to six to date.

Invasive animal species actions

There are good examples of sustainable control projects operating in the Pacific. A key action for environmental protection is to prevent the spread of IAS across international or internal borders. The four main stages are pre-export control, pre-border control, at-border control, and post-border rapid response.

Niue created a harmonised Biosecurity Bill which allows environmental concerns to be addressed along with the traditional agricultural and trade concerns. Early detection and rapid response (EDRR) plans have been created for the Cook Islands, Kiribati, and Samoa. The plans detail the staff and funding requirements, identify best practices regarding known target species, and convey decisions made about the country's approach to the known and potential IAS. Simulation exercises were completed to identify gaps before a response becomes necessary.

The Battler series booklet "Catch it early: invasive species early detection and rapid response" outlines the components of effective IAS response systems. The creation of response plans, training of staff, and procurement of equipment needs to be supported by ongoing engagement, regular refresher simulation exercises, and greater public awareness to maintain fast responses to incursions. The accidental introduction of five mongooses to Tonga in 2016 demonstrated the need for rapid, planned response action. Long-term management is often required for IAS that cannot be eradicated due to their value as a cultural or livelihood resource or simply the amount of resources required to do so. Such management requires ongoing resourcing but may be the only option available, so the value that is being protected from the IAS needs to outweigh the cost of management.

Managing pigs is a balance between cultural or food needs and environmental needs. Domestic pigs are kept in pens as an important food source, but pigs that get out of pens cause a lot of damage. Investing in upskilling locals to the level of professional hunters has paid off in Niue. In 12 months, approximately 130 pigs have been hunted by locals, reducing the population by one half from the estimated pre-hunt total. Professional pig-hunting dogs, global positioning systems, and expert mentoring have resulted in a sustainable, low-cost method for managing pigs on Niue. In Samoa and Vanuatu, crown of thorns starfish is the target of on-going control. Although a native species, crown of thorns can become invasive following modifications to the environment by man or natural disasters such as tsunamis and cyclones. In both countries, the local communities are provided with tools, procedures, and support to lower the impact that the outbreaks have on their local marine ecosystem.

Many IAS are already widespread in the Pacific and impacting biodiversity, including in protected natural areas. When this is the case, there are still options to protect these species and ecosystems with a site-led or asset-based approach. Exclusion of IAS is an option if the surrounding environment contains widespread or otherwise unmanageable IAS which may affect high-value areas. It can also be used as a short-term measure until a solution becomes available. In Mount Talau National Park in the Vava'u islands of Tonga, a rare plant *Casearia buelowii*, which is endemic to Mt Talau and only survives through fewer than 20 individuals, was being continually undermined by pigs, exposing the roots to damage and the heat of the sun. A Tongan pig fence was constructed around the site to exclude the pigs until a long-term pig management solution can be found. Widespread IAS may also be contained to restrict their arrival in uninfested areas. On Niue, the primary infestation of taro vine is situated within the villages of Alofi and Alofi South. Isolated infestations are targeted to contain the infestation to the primary infestation site.

There are good examples of site-led or asset-based restoration projects operating in the Pacific. Restoration supports species recovery and the continued provision of ecosystem services. Ideally, restoration involves the community at multiple scales because restoration is a long-term, if not continual, process. The Kingdom of Tonga is restoring two key ecological sites: the Toloa Rainforest and

Mt Talau. The Toloa Rainforest restoration efforts include a Pacific (*R. exulans*) and ship rat (*R. rattus*) control programme, via bait stations, for the whole forest to save native bird and plant species from predation. Replanting of native trees in the rainforest began in 2014 to improve structure and size and to reintroduce plant species that have gone extinct within the forest. Planting will continue until the extension of the forest is complete and the forest sub-canopy and disturbed sites are restored. Weed and rat control will continue into the future. Toloa Rainforest is the last remaining stand of native forest on the main Tongan island of Tongatapu and serves as an educational resource for the schools of Tonga. A key aspect of the project is making information readily available for people who visit the forest, and many of these informational products explain native species and IAS threats. A trail, with rest and wildlife viewing stops throughout, has also been developed.

Hengahenga, or Tongan whistler (*Pachycephala jacquinoti*), are recovering on Mt Talau following rat control. Rodents have been controlled for four years with statistically significant increases in the number of Tongan whistler (endemic to Vava‘u) and other birds such as the Polynesian triller (*Lalage maculosa*) and Polynesian starling. Rats heavily impact the survival and productivity of the Tongan whistler because the birds build an open bowl nest that is easily accessed by rats. The control programme is run by the local community with the assistance of the Vava‘u Environmental Protection Association. It uses a rat bait-take database that captures, stores, and reports on bait take at each bait station during the programme and allows analysis of bait take to inform success at lowering the rat population, identify areas of high rat activity, and allow for more economical use of the bait. Hengahenga are now seen and heard in the surrounding area with many Tongans witnessing this bird for the first time in their lives.

Samoa has also embarked on restoring two important sites, Mount Vaea Reserve and O Le Pupu Pu‘e National Park. On Mt Vaea, the focus has been on controlling widespread weeds, which form 90 percent of all stems within the regular survey sampling plots. Following weed control, each area is re-vegetated with native trees which quickly form a canopy, reducing the ability of the light-demanding weed species to grow. The weed species that are adapted to grow under low light conditions are regularly managed. Six hectares of Mt Vaea have now been restored, with the on-site nursery having provided19,000 trees for volunteers and the local village to plant. The focus at O Le Pupu Pu‘e has been on planting to suppress the *Merremia peltata* infestations that are restricting regeneration following disturbances such as cyclones. Again, the closed canopy of the revegetated areas is supressing this light-demanding species.

FUTURE PACIFIC PROJECTS

The GEF-PAS Project was evaluated by the United Nations Environment Programme (UNEP, 2017). The summary of project criteria “Strategic relevance”, “Achievement of outputs”, “Communication and public awareness”, “Supervision, guidance and technical backstopping” all had overall ratings of Highly Satisfactory; “Socio-political sustainability” had an overall rating of Highly Likely; “Preparation and readiness” had an overall rating of Unsatisfactory.

The increased establishment and support of existing national institutionalised IAS programmes needs to be carried forward, building upon the success of GEF-PAS. Project under-preparedness and -readiness is common to most Pacific projects due to the lack of capacity and increased logistical challenges in such a large geographic area. Future project preparation times for large, complex projects in the Pacific need to be extended for one year to allow for this.

The “Guidelines for Invasive Species Management in the Pacific” provide an effective framework to plan IAS programmes and can be directly used to plan actions within NISSAPs to measure and analyse actions between countries with consistency. Regional support was vital in achieving quality outcomes. The GEF-PAS project allowed support to be increased substantially, and this needs to continue. Empowering country coordinators and stakeholders through coaching, capacity building, and support in-country, is the most important means to sustain IAS management capability. This needs continued support.

The Pacific Invasives Learning Network appears to be the most effective means for Pacific practitioners to work together and learn from each other. PILN creates and supports the regional flow of information regarding IAS management. PILN requires sustained support. Successful projects resulted in increased visibility and support by local communities, other related sectors, and at the political level. This support is indicated by institutionalisation within government agencies of a core IAS role, as has been the case in some countries, and the commitment to progressing IAS management within the following Global Environment Facility replenishment cycles and other funds.

The GEF-PAS project has made significant progress. The implementation of GEF 6 and EDF 11 projects will build substantially on these successes and continue to refine and enhance the management of IAS in the Pacific Region. Climate change is increasing the intensity and urgency of the response to IAS by reducing the capacity and resilience of Pacific ecosystems and societies to adapt to climate change. Invasive alien species management needs to be an accepted tool for Pacific ecosystems and communities to adapt to climate change.

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REFERENCES

SPREP. (2009). *Guidelines for invasive species management in the Pacific: A Pacific strategy for managing pests, weeds and other invasive species*. Compiled by Alan Tye. Apia, Samoa. <https://www.sprep.org/node/9588>.

SPREP. (2013). *State of Conservation in Oceania Regional Report* (2013) Published by the Secretariat of the Pacific Regional Environment Programme. <https://www.sprep.org/attachments/Publications/BEM/state-conservation-oceania-report.pdf>.

SPREP. (2016). *Battling Invasive Species in the Pacific: Outcomes of the Regional GEF-PAS IAS Project; Prevention, control and management of invasive species in the Pacific Islands* Secretariat of the Pacific Regional Environment Programme. <https://www.sprep.org/attachments/Publications/BEM/battling-invasive-species-pacific.pdf>.

UNEP. (2017). *Prevention, control and management of Invasive Alien Species in the Pacific Islands.* Evaluation Office of UN Environment, Terminal Evaluation of UN Environment Project. <https://www.google.com/url?sa=t&rct=j&q=&esrc=s&source=web&cd=1&ved=2ahUKEwjJpt2OhPPcAhXGgLwKHb4KBkcQFjAAegQIABAC&url=https%3A%2F%2Fwww.thegef.org%2Fsites%2Fdefault%2Ffiles%2Fproject_documents%2F3664_2017_TE_UNEP_Regional_BD_FSP_UNEPSPEM_IAS_Pacific_1.pdf&usg=AOvVaw1XY9XpCiwpWr_JBuKLlvR0>.

Maximising conservation impact by prioritising islands for biosecurity

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Abstract Invasive alien species are one of the primary threats to native biodiversity on islands worldwide, and their expansion continues due to global trade and travel. Preventing the arrival and establishment of highly successful invasive species through rigorous biosecurity is known to be more economic than the removal of these species once they have established. However, many islands around the world lack biosecurity regulations or practical measures and establishing biosecurity will require social and financial investments. Guiding these investments towards islands where native biodiversity is at highest risk from potential invasions is of strategic importance to maximise conservation benefit with limited resources. Here we implement an established prioritisation approach, previously used to identify which islands will have the greatest conservation gains from the eradication of invasive species, to identify which islands would benefit the most from establishing or improving biosecurity. We demonstrate this approach for 318 islands in the Caribbean UK Overseas Territories and Bermuda where we considered all threatened native terrestrial vertebrates that are vulnerable to the most harmful invasive vertebrates (black and brown rats, cats, small Indian mongoose, green iguana). The approach calculates the increase in conservation threat score resulting from anticipated negative effects of potential invaders on native biodiversity, and highlighted Sombrero (Anguilla) and Cayman Brac (Cayman Islands) as important islands where threatened reptile species would likely be eliminated if rats, feral cats or mongoose invaded. Feasibility and cost implications should now be investigated more closely on the highlighted islands. The prioritisation presented here can be expanded to more islands and more invasive/native taxa (herbivores, plants and invertebrates), but requires a classification of the severity of potential impacts between invasive and native species for which currently little information exists. Besides highlighting opportunities for biosecurity, this approach also highlights where knowledge gaps about population sizes of and threats to reptiles with restricted ranges exist.

Keywords: Caribbean, feral cat, iguana, invasive mammals, mongoose, rats, reptiles

INTRODUCTION

The majority of the world's archipelagos have been invaded by non-native species, some of which have detrimental effects on native biodiversity (Atkinson, 1985; McCreless, et al., 2016; Turbelin, et al., 2017). Although some islands can be restored by eradicating certain invasive species, such operations can be expensive (Martins, et al., 2006; Holmes, et al., 2015). The limited amount of funding available for island restoration efforts has motivated managers to prioritise the islands where an eradication would yield the greatest biodiversity benefits at global and regional levels (Brooke, et al., 2007; Dawson, et al., 2015; Stanbury, et al., 2017). However, current technologies limit restoration via eradication to 15% of islands that have been invaded (Keitt, et al., 2019), hence eradication is not a universal solution to preserve global island biodiversity.

Preventing harmful species invading those islands which still have globally significant biodiversity values is an important and efficient avenue to prevent loss of biodiversity (Broome, 2007; Russell, et al., 2008; Spatz, et al., 2017). Biosecurity measures also require financial investments, both initially and in perpetuity, to detect and eliminate any potential invaders to islands (Oppel, et al., 2011; Key & Moore, 2019). Because the costs for biosecurity can be considerable, financial constraints can also limit the number of islands that can be protected with effective biosecurity measures (Moore, et al., 2010; Greenslade, et al., 2013). Here we propose to use established prioritisation approaches (Brooke, et al., 2007; Dawson, et al., 2015; Stanbury, et al., 2017) to guide the investment of resources for biosecurity to minimise the risks of invasion of non-native vertebrates to islands where they would cause the greatest loss of biodiversity. We demonstrate this approach for 318 islands that belong to United Kingdom Overseas Territories (UKOTs) in the Caribbean and Bermuda.

The islands in the Caribbean UKOTs feature globally important biodiversity (Forster, et al., 2011; Dawson, et al., 2015; Churchyard, et al., 2016), with a large number of endemic reptiles, birds, and plants. Due to centuries of human habitation and inter-island trade, most islands have been invaded by some non-native species (Hilton & Cuthbert, 2010), but only a few islands contain the complete suite of invasive vertebrate species present in the Caribbean region. In addition, >100 small and uninhabited islands are still free of invasive vertebrate species and function as refugia for some globally threatened species that cannot coexist with harmful invasive vertebrates (Dawson, et al., 2015). Preventing the invasion of non-native vertebrates that have caused significant declines to native species on other islands could secure globally significant populations of threatened vertebrates. Despite the recognised threat of invasive species to endemic biodiversity, biosecurity regulations and implementations are generally insufficient to reduce the risk of further spread of invasive species between islands in the Caribbean region (RSPB, 2017; Key & Moore, 2019).

We conducted a prioritisation that identifies those islands where the invasion of five potentially harmful invasive vertebrates could cause the greatest loss to biodiversity in the Caribbean UKOTs. We recommend immediate investment in feasibility studies and biosecurity on those islands to avoid the invasion of these five species and the subsequent loss of native biodiversity, and we recommend that similar approaches should be used in other regions, or indeed globally, to identify islands where investment in biosecurity is most urgently needed.

METHODS

Study area

We used all 318 islands in the five Caribbean UKOTs (Anguilla, British Virgin Islands, Cayman Islands, Montserrat, and Turks and Caicos Islands) and in Bermuda, which is situated 1,500 km north of the Caribbean but

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is climatically similar (Fig. 1). These islands are mostly tropical and range from small sandy islets of 0.01 ha to islands with mountain ranges and a variety of habitat types > 20,000 ha. Only 14 islands are permanently inhabited by human communities of up to 65,000 people, while the remaining islands are either completely uninhabited, function only as tourist resorts or destinations, or are visited temporarily by fishermen.

Selection of potential invasive species

To assess biodiversity loss that could result from the invasion of harmful animal species, we selected the five most harmful invasive terrestrial vertebrates (McCreless, et al., 2016) that are widespread in the Caribbean region. Green iguanas (*Iguana iguana*) are known to hybridise and compete with native reptiles (Gibbon, et al., 2000; Vuillaume, et al., 2015), small Indian mongoose (*Urva auropunctata*) are versatile predators considered one of the worst invasive species (Hays & Conant, 2007; Barun, et al., 2008), brown (*Rattus norvegicus*) and black rats (*R. rattus*) and feral cats (*Felis catus*) are efficient predators that can have detrimental effects on island biodiversity (Towns, et al., 2006; Jones, et al., 2008; Medina, et al., 2011; Nogales, et al., 2013). These five species are distributed widely across islands in the Caribbean (Kairo, et al., 2003; Dawson, et al., 2015) and are therefore potential invaders of all islands in the region.

Distribution of native and invasive species

For each island we previously collated information on the presence of native and invasive terrestrial vertebrate species for an eradication prioritisation (Dawson, et al., 2015) and a general inventory of biodiversity (Churchyard, et al., 2016), and updated these previous compilations with recent information and threat assessments (IUCN, 2017). We considered all globally threatened terrestrial vertebrate species (including marine turtles) as listed on the International Union for Conservation of Nature Red List of Threatened Species (IUCN, 2017) and all colonial seabird species and restricted range bird species. We also included reptiles of conservation concern that are endemic to a single territory or inhabit fewer than 15 islands across their range (Dawson, et al., 2015). We updated this information with new records shared by local partner organizations since 2013 (Hedges, 2017). We considered the green iguana that exists on Montserrat as a genetically distinct conservation management unit, because it is genetically closely related to the iguana on Saint Lucia, which is treated as a native

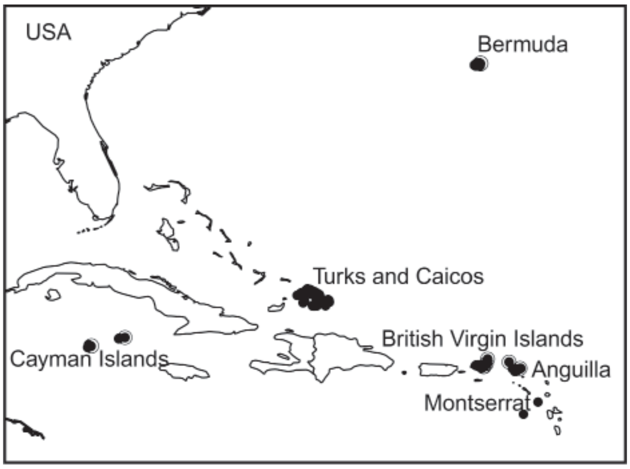

Fig. 1 Location of 318 islands (black dots) in six United Kingdom Overseas Territories where the priority for biosecurity was assessed. Circles around islands indicate the location of the highest priority islands listed in this paper.

species of conservation concern (Powell, 2004; Stephen, et al., 2013; Vuillaume, et al., 2015). Due to the lack of sufficient distribution data and limited existing knowledge of interactions, native and invasive plant or invertebrate species were not considered in this prioritisation.

Calculating the conservation threat score of islands

We followed the approach of Dawson, et al. (2015) to calculate the conservation threat score (termed 'conservation value' in Dawson, et al., 2015) of each island based on the sum of each native species' vulnerability. The vulnerability was calculated as the product of the global threat status, the irreplaceability, which indicates the global significance of an island's population, and the severity of impact of the most harmful invasive vertebrate species already present on an island (i.e. the species with the greatest severity of impact score; Dawson, et al., 2015; Stanbury, et al., 2017). We scored threat and impact categories on both a linear and logarithmic scale to address the arbitrariness of assigning quantitative values to normative categories (Game, et al., 2013; Helmstedt, et al., 2016). The severity of impact was classified in three categories, depending on whether an invasive species had no impact on a native species (0), small to moderate impact that would reduce population size but allow the native species to persist (1), or a severe impact that would eventually lead to the local extinction of the native species (2). We classified unassessed reptiles as 'At Risk', which received a numerical value equivalent to 'Vulnerable' (Dawson, et al., 2015).

Simulating the invasion of islands to calculate increase in conservation threat score

To quantify the magnitude of biodiversity loss that could result from invasion, we first assessed which of the five selected invasive species were already present on an island in 2016, and then simulated the arrival and invasion of those species that were not yet present in 2016. We then re-calculated the conservation threat score of each island as described above, where the vulnerability of each native species was adjusted to reflect the most harmful invasive species on the island, which may be one of the simulated invaders. We assumed that all invasive species not yet present on an island would invade, because biosecurity measurements should, in our opinion, not be tailored for a single species but guard against the arrival of a broad suite of species. However, we emphasise that our prioritisation could also be performed for single species invasions, but assessing the merits of guarding against one or another invasive species would require information about the relative invasion risk of various species.

The calculation of the conservation threat score depends on a classification of the threat posed by each invasive species to each native species, but these threats can be hypothetical for interactions between certain island endemic species and invasive species that have so far not invaded the respective island. Consequently, we drew on taxonomically related or otherwise very similar species to specify the potential threat that would result from invasion. For example, if black rats adversely affect a small *Sphaerodactylus* gecko on one island, we assumed that a similarly sized *Sphaerodactylus* species that is endemic to an island without any rats would suffer similar effects if the island were invaded by rats (Case & Bolger, 1991).

Prioritising islands for biosecurity

Islands that should receive the most immediate investment into biosecurity are those where the native fauna would face the greatest increase in conservation threat score if the five selected vertebrate species invaded. We therefore calculated the difference in conservation value at present and after the simulated invasion of the five vertebrate species, and ranked islands based on the magnitude of this difference. We present the results as a ranking table and include information on island size and human population size for each island. These aspects will affect the complexity and cost of biosecurity measures, as well as the probability of invasive species arrival and establishment, but they did not factor into our prioritisation of islands for biosecurity, which was entirely based on the potential threat to native biodiversity. All calculations were performed in R 3.2.5 (R Development Core Team 2015) based on the code provided by Dawson, et al. (2015).

RESULTS

Of the 318 islands in our assessment, 125 did not have any invasive species on them, and 150 (47%) did not have any of the five focal invasive species. Of the islands with any of the five focal invasive species, 31 (10%) had one invasive, 117 (37%) had two, 12 (4%) had three, 6 (2%) had four, and only two islands (Tortola and Virgin Gorda, British Virgin Islands) had all five of the focal invasive species. On 183 islands (57.5%) the invasion of any of the five focal invasive species would not lead to an increase in the conservation threat score, because the native vertebrates on these islands were not at greater risk of predation from those invasive species that have not yet invaded. Thus, biosecurity measures to prevent the invasion of at least one of the five focal species would be useful on 133 islands in our assessment.

We identified several important islands across the Caribbean UKOTs and Bermuda where biosecurity could help prevent the loss of globally important biodiversity (Table 1). Two islands emerged where an invasion of non-native vertebrates could lead to an increase in the conservation threat score more than five times greater than on any other island included in our study, mostly due to the potential loss of Critically Endangered endemic reptiles (Table 1): Sombrero (Anguilla), and Cayman Brac (Cayman Islands).

Among the most important islands we identified for biosecurity, three were inhabited by >1000 people and have existing populations of rats, feral cats, and green iguanas (Cayman Brac, Grand Cayman, and Montserrat, Table 1). However, the small Indian mongoose is so far absent from those islands and reducing the risk of invasion of this efficient predator on islands that already have other harmful invasive species could help secure globally important biodiversity. Together with Montserrat, Anegada in the British Virgin Islands was among the top priorities for biosecurity to reduce the risk of invasion of black rats and small Indian mongoose, despite both islands also being a high priority for the eradication of already existing invasive species (Dawson, et al., 2015).

DISCUSSION

We show that effective biosecurity on islands in the Caribbean UK Overseas Territories could reduce the risk of further spread of harmful invasive vertebrates to islands where globally threatened reptiles and birds would be at risk. Investing in effective biosecurity procedures and educating the public and policy makers about the risks to their national heritage when no biosecurity is in place should be the immediate next steps of UK and local governments, private island owners, and international funding bodies. Our approach offers the guidance to focus on a limited number of vulnerable islands, as more than half of the islands we evaluated are not at immediate risk of further biodiversity loss from the invasion of the five invasive vertebrate species that we selected.

Similar to other prioritisations identifying islands for eradication of invasive species (e.g., Harris, et al., 2012; Dawson, et al., 2015; Stanbury, et al., 2017), our list is subject to incomplete information about the distribution of both native and invasive species. The distribution of several reptile species is poorly documented across many islands of the Caribbean, and their threat status is also poorly assessed on the IUCN Red List, both of which may affect our assessment of their local importance and therefore introduce bias to our projections of loss in conservation value (Russell, et al., 2017). Further surveys to increase the knowledge of native and invasive species on islands would be beneficial but should not be used as an argument to delay the immediate adoption of effective biosecurity protocols to safeguard the most important islands that we identified.

Besides thorough knowledge about the native and invasive species occurring on an island, our approach also requires a classification of the interactions between native and invasive species. Because these interactions can be hypothetical for single-island endemic native species that have not been exposed to invasive species, due caution is necessary when interpreting the output of our prioritisation. We used the response of taxonomically similar species to the same invasive species to predict biologically plausible consequences of an invasion, but interactions between native and invasive species are often complex and unpredictable (Simberloff & Von Holle, 1999; Simberloff, 2006). We encourage researchers to provide robust and reliable predictions about the potential consequences of invasions to assist with strategic investment decisions for reducing the risk of invasive species becoming established on islands harbouring globally important biodiversity (Moore, et al., 2010).

In summary, we demonstrated that biosecurity is not only important on small uninhabited islands or privately owned tourist resorts where natural habitats remain and endemic and globally threatened species persist. Even on large and populated islands such as Grand Cayman, Cayman Brac, and Montserrat, the invasion of small Indian mongoose could result in a significant deterioration of the conservation status of several globally threatened vertebrates (Hays & Conant, 2007). We therefore urge local governments, private island owners (e.g. Mosquito Island) and communities to carefully inspect all incoming cargo and people and establish ongoing measures to detect and remove any new invasive species. Training of border officials and conservation staff, public education and awareness campaigns targeting the accidental introduction of invasive species onto uninhabited islands by visiting people (e.g. fishermen, tourists) should also be implemented, because international and domestic biosecurity measures are currently weak across all Caribbean UK Overseas Territories (Key, 2017; RSPB, 2017). Laws governing biosecurity measures in the Caribbean UK Overseas Territories and Bermuda are disjointed, not comprehensive and scattered through various environmental, agricultural and customs regulations. Collaboration under existing national legislative mechanisms may improve the situation quickly prior to enacting any new legislation (RSPB, 2017). We would also encourage regional collaboration in developing biosecurity measures, information sharing and learning from any existing biosecurity initiatives.

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Table 1 The top 25 islands in Caribbean UK Overseas Territories and Bermuda where the invasion of five common vertebrate species could potentially cause the greatest increase in threats to native biodiversity. Note that some invasive species are already present on some islands (i.e. those with current threat score > 0), and only the potential new invaders are listed; islands with a current threat score >0 would also benefit from the removal of already existing invasive species. Human population size and island area are provided for information, as they will affect invasion risk and effort required for biosecurity. The current and post-invasion threat scores are calculated as the sum of all impact scores of invasive species on all threatened native species present on an island before and after potential invasion.

Island	UKOT	Human popul'n	Island area (ha)	Current threat	Post-invasion threat	Potential increase in conservation threat score	Potential invaders	Globally threatened species at risk from invasion
Sombrero	Anguilla	0	29.2	0.0	5,729.2	5,729.2	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Sphaerodactylus* sp., *Ameiva corvina*
Cayman Brac	Cayman Islands	2,098	3,889.4	1,364.7	6,879.3	5,514.5	Small Indian mongoose	*Dendrocygna arborea, Crocodylus acutus, Anolis luteosignifer, Typhlops epactius, Tropidophis schwartzi, Anolis maynardii, Celestus maculatus, Cyclura nubila caymanensis*
Mosquito Island	British Virgin Islands	0	49.2	50.6	978.1	927.5	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Spondylurus semitaeniatus, Sphaerodactylus parthenopion, Cyclura pinguis, Amphisbaena fenestrata*
Little Scrub Island	Anguilla	0	4.1	0.0	737.6	737.6	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Ameiva corax*
Grand Cayman	Cayman Islands	53,160	20,159.4	1,249.5	1,986.5	737.0	Small Indian mongoose	*Eretmochelys imbricata, Chelonia mydas, Caretta caretta, Dendrocygna arborea, Crocodylus acutus, Cyclura lewisi, Anolis conspersus, Crocodylus rhombifer, Typhlops caymanensis, Tropidophis caymanensis*
Salt Island	British Virgin Islands	0	78.2	161.7	406.6	244.9	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Eretmochelys imbricata, Spondylurus semitaeniatus, Spondylurus sloanii, Amphisbaena fenestrata*
Carval Rock	British Virgin Islands	0	1.0	0.0	242.7	242.7	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Amphisbaena fenestrata, Sphaerodactylus* sp.
Anegada	British Virgin Islands	200	3,844.4	11,103.9	11,313.9	210.0	Black rat, small Indian mongoose	*Chelonia mydas, Eretmochelys imbricata, Dermochelys coriacea, Cyclura pinguis, Spondylurus anegadae*
Nonsuch Island	Bermuda	0	8.2	106.0	310.0	204.0	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Plestiodon longirostris, Pterodroma cahow*

Table 1 (continued) The top 25 islands in Caribbean UK Overseas Territories and Bermuda where the invasion of five common vertebrate species could potentially cause the greatest increase in threats to native biodiversity. Note that some invasive species are already present on some islands (i.e. those with current threat score > 0), and only the potential new invaders are listed; islands with a current threat score >0 would also benefit from the removal of already existing invasive species. Human population size and island area are provided for information, as they will affect invasion risk and effort required for biosecurity. The current and post-invasion threat scores are calculated as the sum of all impact scores of invasive species on all threatened native species present on an island before and after potential invasion.

Island	UKOT	Human populn	Island area (ha)	Current threat	Post-invasion threat	Potential increase in conservation threat score	Potential invaders	Globally threatened species at risk from invasion
Horn Rock	Bermuda	0	0.3	106.0	310.0	204.0	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Plestiodon longirostris, Pterodroma cahow*
Inner Pear Rock	Bermuda	0	0.9	105.3	308.5	203.3	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Plestiodon longirostris, Pterodroma cahow*
Ginger Island	British Virgin Islands	0	102.2	0.0	201.3	201.3	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Eretmochelys imbricata, Spondylurus semitaeniatus, Amphisbaena fenestrata*
Fallen Jerusalem	British Virgin Islands	0	19.5	0.0	194.4	194.4	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Eretmochelys imbricata, Spondylurus semitaeniatus, Amphisbaena fenestrata*
Fish Cay	Turks and Caicos	0	8.5	0.0	170.4	170.4	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Eretmochelys imbricata, Cyclura carinata*
Bush Cay	Turks and Caicos	0	7.8	0.0	168.4	168.4	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Eretmochelys imbricata, Cyclura carinata*
Round Rock	British Virgin Islands	0	6.4	0.0	140.3	140.3	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Spondylurus semitaeniatus, Amphisbaena fenestrata*
Montserrat	Montserrat	4922	10157.4	12571.3	12708.1	136.8	Green iguana, small Indian mongoose	*Dermochelys coriacea, Eretmochelys imbricata, Chelonia mydas, Caretta caretta, Turdus lherminieri, Icterus oberi, Diploglossus montisserrati, Leptodactylus fallax, Anolis lividus, Mabuya montserratae, Iguana iguana* (Montserrat)
Big Sand Cay	Turks and Caicos	0	54.5	0.0	131.7	131.7	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Chelonia mydas, Cyclura carinata, Leiocephalus psammodromus*

Table 1 (continued) The top 25 islands in Caribbean UK Overseas Territories and Bermuda where the invasion of five common vertebrate species could potentially cause the greatest increase in threats to native biodiversity. Note that some invasive species are already present on some islands (i.e. those with current threat score > 0), and only the potential new invaders are listed; islands with a current threat score >0 would also benefit from the removal of already existing invasive species. Human population size and island area are provided for information, as they will affect invasion risk and effort required for biosecurity. The current and post-invasion threat scores are calculated as the sum of all impact scores of invasive species on all threatened native species present on an island before and after potential invasion.

Island	UKOT	Human populn	Island area (ha)	Current threat	Post-invasion threat	Potential increase in conservation threat score	Potential invaders	Globally threatened species at risk from invasion
Six Hills East Cay	Turks and Caicos	0	3.1	0.0	123.0	123.0	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Cyclura carinata, Aristelliger hechti*
Six Hills West Cay	Turks and Caicos	0	6.2	0.0	123.0	123.0	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Cyclura carinata, Aristelliger hechti*
Middleton Cay	Turks and Caicos	0	4.6	0.0	121.5	121.5	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Cyclura carinata, Tropidophis greenwayi*
Long Cay (Turks)	Turks and Caicos	0	18.4	0.0	117.0	117.0	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Cyclura carinata, Leiocephalus psammodromus*
White Cay	Turks and Caicos	0	2.4	0.0	111.8	111.8	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Cyclura carinata*
Indian Cay	Turks and Caicos	0	2.9	0.0	111.3	111.3	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Cyclura carinata*
Plandon Cay	Turks and Caicos	0	15.5	0.0	110.3	110.3	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Cyclura carinata*
Middle Creek Cay	Turks and Caicos	0	41.6	0.0	110.3	110.3	Brown rat, black rat, feral cat, green iguana, small Indian mongoose	*Cyclura carinata*

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REFERENCES

Atkinson, I.A.E. (1985). 'The spread of commensal species of *Rattus* to oceanic islands and their effects on island avifaunas'. In*:* P.J. Moors (ed.) *Conservation of Island Birds.* pp. 35–81. Cambridge, United Kingdom: International Council for Bird Preservation.

Barun, A., Budinski, I. and Simberloff, D. (2008). 'A ticking time-bomb: The small Indian mongoose in Europe'. *Aliens* 26: 14–16.

Brooke, M.d.L., Hilton, G.M. and Martins, T.L.F. (2007). 'Prioritizing the world's islands for vertebrate-eradication programmes'. *Animal Conservation* 10: 380–390.

Broome, K. (2007). 'Island biosecurity as a pest management tactic in New Zealand'. In*:* G. Witmer, W. Pitt, K. Fagerstone and C. Clark (eds.) *Proceedings of an International Symposium Managing Vertebrate Invasive Species.* pp. 104–107. Lincoln: University of Nebraska.

Case, T.J. and Bolger, D.T. (1991). 'The role of introduced species in shaping the distribution and abundance of island reptiles'. *Evolutionary Ecology* 5: 272–290.

Churchyard, T., Eaton, M.A., Havery, S., Hall, J., Millett, J., Farr, A., Cuthbert, R.J., Stringer, C. and Vickery, J.A. (2016). 'The biodiversity of the United Kingdom's Overseas Territories: A stock take of species occurrence and assessment of key knowledge gaps'. *Biodiversity and Conservation* 25: 1677–1694.

Dawson, J., Oppel, S., Cuthbert, R.J., Holmes, N., Bird, J.P., Butchart, S.H.M., Spatz, D.R. and Tershy, B. (2015). 'Prioritizing islands for the eradication of invasive vertebrates in the United Kingdom overseas territories'. *Conservation Biology* 29: 143–153.

Forster, J., Lake, I.R., Watkinson, A.R. and Gill, J.A. (2011). 'Marine biodiversity in the Caribbean UK overseas territories: Perceived threats and constraints to environmental management'. *Marine Policy* 35: 647–657.

Game, E.T., Kareiva, P. and Possingham, H.P. (2013). 'Six common mistakes in conservation priority setting'. *Conservation Biology* 27: 480–485.

Gibbon, J.W., Scott, D.E., Ryan, T.J., Buhlmann, K.A., Tuberville, T.D., Metts, B.S., Greene, J.L., Mills, T., Leiden, Y. and Poppy, S. (2000). 'The global decline of reptiles, Déjà Vu amphibians'. *BioScience* 50: 653–666.

Greenslade, P., Burbidge, A.A. and Lynch, A.J.J. (2013). 'Reply to: Protecting islands from pest invasion: Optimal allocation of biosecurity resources between quarantine and surveillance'. *Biological Conservation* 157: 434.

Harris, D., Gregory, S., Bull, L. and Courchamp, F. (2012). 'Island prioritization for invasive rodent eradications with an emphasis on reinvasion risk'. *Biological Invasions* 14: 1251–1263.

Hays, W.S.T. and Conant, S. (2007). 'Biology and impacts of Pacific Island invasive species. 1. A worldwide review of effects of the small Indian mongoose, *Herpestes javanicus* (Carnivora : Herpestidae)'. *Pacific Science* 61: 3–16.

Hedges, S.B. (2017). *Caribherp: West Indian Amphibians and Reptiles* (<www.caribherp.org>). Philadelphia, Pennsylvania: Temple University.

Helmstedt, K.J., Shaw, J.D., Bode, M., Terauds, A., Springer, K., Robinson, S.A. and Possingham, H.P. (2016). 'Prioritizing eradication actions on islands: It's not all or nothing'. *Journal of Applied Ecology* 53: 733–741.

Hilton, G.M. and Cuthbert, R. (2010). 'The catastrophic impact of invasive mammalian predators on birds of the UK Overseas Territories: a review and synthesis'. *Ibis* 152: 443–458.

Holmes, N., Campbell, K., Keitt, B., Griffiths, R., Beek, J., Donlan, C. and Broome, K. (2015). 'Reporting costs for invasive vertebrate eradications'. *Biological Invasions* 17: 2913–2925.

IUCN (2017). IUCN Red List of Threatened Species. Gland, Switzerland: IUCN – The World Conservation Union. <http://www.iucnredlist.org/>.

Jones, H.P., Tershy, B.R., Zavaleta, E.S., Croll, D.A., Keitt, B.S., Finkelstein, M.E. and Howald, G.R. (2008). 'Severity of the effects of invasive rats on seabirds: A global review'. *Conservation Biology* 22: 16–26.

Kairo, M., Ali, B., Cheesman, O., Haysom, K. and Murphy, S. (2003). *Invasive Species Threats in the Caribbean Region.* Arlington, VA: The Nature Conservancy.

Keitt, B., Holmes, N., Hagen, E., Howald, G. and Poiani, K. (2019). 'Going to scale: Reviewing where we've been and where we need to go in invasive vertebrate eradications'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 633–636. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Key, J. (2017). *Tackling Invasive Non-native Species in the UK Overseas Territories – Gap Analysis of Biosecurity Capacity.* Sand Hutton, UK: Animal and Plant Health Agency.

Key, G.E. and Moore, N.P. (2019). 'Tackling invasive non-native species in the UK Overseas Territories'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 637–642. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Martins, T.L.F., Brooke, M.d.L., Hilton, G.M., Farnsworth, S., Gould, J. and Pain, D.J. (2006). 'Costing eradications of alien mammals from islands'. *Animal Conservation* 9: 439–444.

McCreless, E.E., Huff, D.D., Croll, D.A., Tershy, B.R., Spatz, D.R., Holmes, N.D., Butchart, S.H.M. and Wilcox, C. (2016). 'Past and estimated future impact of invasive alien mammals on insular threatened vertebrate populations'. *Nature Communications* 7: 12488.

Medina, F.M., Bonnaud, E., Vidal, E., Tershy, B.R., Zavaleta, E.S., Josh Donlan, C., Keitt, B.S., Corre, M., Horwath, S.V. and Nogales, M. (2011). 'A global review of the impacts of invasive cats on island endangered vertebrates'. *Global Change Biology* 17: 3503–3510.

Moore, J.L., Rout, T.M., Hauser, C.E., Moro, D., Jones, M., Wilcox, C. and Possingham, H.P. (2010). 'Protecting islands from pest invasion: optimal allocation of biosecurity resources between quarantine and surveillance'. *Biological Conservation* 143: 1068–1078.

Nogales, M., Vidal, E., Medina, F.M., Bonnaud, E., Tershy, B.R., Campbell, K.J. and Zavaleta, E.S. (2013). 'Feral cats and biodiversity conservation'. *BioScience* 63: 804–810.

Oppel, S., Beaven, B., Bolton, M., Vickery, J.A. and Bodey, T.W. (2011). 'Eradication of invasive mammals on islands inhabited by humans and domestic animals'. *Conservation Biology* 25: 232–240.

Powell, R. (2004). 'Conservation of iguanas (*Iguana delicatissima* and *I. iguana*) in the Lesser Antilles'. *Iguana* 11: 239–246.

RSPB (2017). *Assessment of legislation supporting biosecurity in the Caribbean UK Overseas Territories and Bermuda.* Sandy, Bedfordshire, UK: Royal Society for the Protection of Birds.

Russell, J.C., Towns, D.R. and Clout, M.N. (2008). *Review of Rat invasion Biology: Implications for Island Biosecurity.* Science for Conservation 286. Wellington, New Zealand: Department of Conservation.

Russell, J.C., Meyer, J.-Y., Holmes, N.D. and Pagad, S. (2017). 'Invasive alien species on islands: Impacts, distribution, interactions and management'. *Environmental Conservation* 44: 359–370.

Simberloff, D. and Von Holle, M. (1999). 'Synergistic interactions of nonindigenous species: invasional meltdown'. *Biological Invasions* 1: 21–32.

Simberloff, D. (2006). 'Risk assessments, blacklists, and white lists for introduced species: Are predictions good enough to be useful?'. *Agricultural and Resource Economics Review* 35: 1–10.

Spatz, D.R., Holmes, N.D., Reguero, B.G., Butchart, S.H.M., Tershy, B.R. and Croll, D.A. (2017). 'Managing invasive mammals to conserve globally threatened seabirds in a changing climate'. *Conservation Letters* 10: 736–747.

Stanbury, A., Thomas, S., Aegerter, J., Brown, A., Bullock, D., Eaton, M., Lock, L., Luxmoore, R., Roy, S., Whitaker, S. and Oppel, S. (2017). 'Prioritising islands in the United Kingdom and crown dependencies for the eradication of invasive alien vertebrates and rodent biosecurity'. *European Journal of Wildlife Research* 63: 31.

Stephen, C.L., Reynoso, V.H., Collett, W.S., Hasbun, C.R. and Breinholt, J.W. (2013). 'Geographical structure and cryptic lineages within common green iguanas, *Iguana iguana*'. *Journal of Biogeography* 40: 50–62.

Towns, D., Atkinson, I. and Daugherty, C. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?'. *Biological Invasions* 8: 863–891.

Turbelin, A.J., Malamud, B.D. and Francis, R.A. (2017). 'Mapping the global state of invasive alien species: patterns of invasion and policy responses'. *Global Ecology and Biogeography* 26: 78–92.

Vuillaume, B., Valette, V., Lepais, O., Grandjean, F. and Breuil, M. (2015). 'Genetic evidence of hybridization between the endangered native species *Iguana delicatissima* and the invasive *Iguana iguana* (Reptilia, Iguanidae) in the Lesser Antilles: management implications'. *PLOS ONE* 10: e0127575.

Working with the local community to eradicate rats on an inhabited island: securing the seabird heritage of the Isles of Scilly

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Abstract The inhabited Isles of Scilly, 45 km off the south-western tip of the UK, are home to 13 seabird species including European storm petrel (*Hydrobates pelagicus*) and Manx shearwater (*Puffinus puffinus*), for which the UK has a global responsibility. Between 1983 and 2006, the overall seabird population in Scilly declined by c.25%. This decline triggered the establishment of the Isles of Scilly Seabird Recovery Project, a partnership with the aims to reverse seabird decline and engage the local community and visitors in conserving Scilly's seabird heritage. The eradication of brown rats (*Rattus norvegicus*) from St Agnes and Gugh represented the result of over a decade of preparatory work, involving raising awareness and gaining 100% support from the community. The two islands are home to 85 people. Therefore additional, and somewhat unusual, preparations were required (including clearing sheds, communicating with school children and taking precautions to ensure the safety of pets) during the ground-based baiting operation. In 2016 St Agnes and Gugh were officially declared 'rat-free', meaning worldwide this is one of the largest community-based eradications to have been successful. Biosecurity on inhabited islands is complex, so to ensure the project's sustainability, efforts have been community-led. The community has taken ownership of protecting its seabirds, with 100% saying rat removal and the subsequent increase in seabirds has had, or will have, a positive effect on ecotourism, a key source of income for the islands. No less than 68% of the community said their businesses have directly benefited. This project represents a case study for other community-based projects, showcasing how eradications can gain community support and benefit both wildlife and human populations.

Keywords: biosecurity, brown rat, eradication, Gugh, inhabited, public support, St Agnes

INTRODUCTION

The eradication of invasive species from islands has become one of the most important tools for biodiversity conservation but it can also improve local socio-economics, human health and ecotourism. Rodents have been successfully eradicated from islands throughout the world, including a number of UK islands (Bell, et al., 2000; Zonfrillo, 2001; Towns & Broome, 2003; Bell, 2004; Howald, et al., 2007; Bell, et al., 2011; Thomas, et al., 2017; Bell, 2019). Most of these islands have been uninhabited and many consider that islands with significant human populations, an unreceptive local community or occurrence of livestock and domestic animals are unlikely to be feasible for eradication (Campbell, et al., 2015). Given that an increasing number of eradications are being investigated on inhabited islands, the importance of the engagement and inclusion of local communities has been highlighted in a number of recent eradication and research projects (Oppel, et al., 2010; Bryce, et al., 2011; Eason, et al., 2011, Walsh, et al., 2019). The opinions and safety of the local community need to be a priority in any eradication planned for inhabited islands (Stanbury, et al., 2017). Without compliance of the full community, access to properties may be denied which may result in the failure of eradicating every rodent or following the eradication, community members may compromise ongoing biosecurity measures.

Human activities can affect the success of an eradication campaign, particularly waste management, food storage, buildings harbouring rat nesting materials, and limited access to certain areas of the island. On the inhabited UK islands where previous eradications have been completed, they have been staffed by personnel working for the owners of the island, for example Lundy, UK (Bell, 2004) and Isle of Canna, UK (Bell, et al., 2011) whereby the parties involved are working within the confines of employment contracts. This is not the case with community members. Other wildlife control projects may have seen decision-makers 'persuade the community' to accept their decision, e.g. the delayed rodent eradication programme for Lord Howe Island (Australia) whereby many inhabitants felt excluded from initial planning (Crowley, et al., 2017b).

The purpose of this paper is to set out the community involvement through the various stages of the Isle of Scilly Seabird Recovery Project, how the views of the local community were collected and used in the design and delivery of the project to establish and maintain community support and evaluate how successful the project was in achieving this.

Background to 'Isles of Scilly Seabird Recovery Project'

The Isles of Scilly are 45 km off the southwest tip of the UK (Fig. 1). As an island group, they are made up of five inhabited islands (St Mary's, St Martin's, Tresco, Bryher and St Agnes and Gugh) and up to 190 uninhabited islets and stacks (1,641 ha, Parslow, 2007). The Isles of Scilly are nationally important for many species of seabirds, supporting 20,000 birds of 13 native species including the burrow-nesting species Manx shearwater (*Puffinus puffinus*) and European storm petrel (*Hydrobates pelagicus*) (Lock, et al., 2006). Declines of 25% had raised significant conservation concerns about the future of the seabirds on the islands. The Isles of Scilly 'Seabird Liaison Group' (SLG) is a partnership between Royal Society for the Protection of Birds (RSPB), Natural England, Isles

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of Scilly Wildlife Trust (IOSWT), Area of Outstanding Natural Beauty (AONB) and Isles of Scilly Bird Group, working within the 'Isles of Scilly Seabird Conservation Strategy' since 2006 (Lock, et al., 2006; Lock, et al., 2009; St Pierre, et al., 2014). This strategy describes the status and context of the seabird populations on the Isles of Scilly and identifies priority actions including current and future measures to improve the available habitat for seabirds (Lock, et al. 2006; Lock, et al. 2009; St Pierre, et al., 2014). The eradication of brown rats (*Rattus norvegicus*) from St Agnes and Gugh was identified as a priority action to remove the threat of mammalian reinvasion on the neighbouring uninhabited island of Annet and provide the opportunity for Manx shearwaters and storm petrels to breed successfully once St Agnes and Gugh were cleared of rats.

St Agnes (105 ha) and Gugh (37 ha) are two islands connected by a rock and sand bar at low tide and are separated from the island of St Mary's by a deep,1 km wide channel. The main habitats are farmland, ponds, maritime heathland and grassland, rocky shores and sandy beaches (Parslow, 2007). Non-native *Pittosporum crassifolium* and *Coprosma repens* were introduced as part of the flower farming industry as shelter hedges in the late 1800s. There is a pub, a Post Office and shop, two cafes, a campsite and two community halls. Brown rats were accidentally introduced to the Isles of Scilly from shipwrecks in the 1700s and were widespread and abundant across both islands (McCann, 2005). The 'community' of St Agnes is defined as the 85 residents who live full time on St Agnes, plus two part-time residents who live on Gugh for six months of the year in holiday homes.

Prior to and during the period of the 'Isles of Scilly Seabird Conservation Strategy 2006–2013', seabird conservation awareness activities were delivered on the islands through community engagement by RSPB, IOSWT and AONB. These activities were delivered through press releases, articles and presentations updating residents on the outcomes of annual seabird monitoring surveys, seabird youth education, advocacy at the island fetes and beach cleans. These activities represented a decade of preparatory work enabling the community of St Agnes and Gugh to learn about and take pride in protecting its seabird heritage.

Fig. 1 Map of the Isles of Scilly, 45 km south-west off the tip of the UK.

In 2010, the SLG held a workshop on St Mary's, to initially obtain the views of residents on all inhabited islands regarding options for control, eradication and the importance of seabirds. This workshop provided the mandate for the SLG to commission a detailed assessment into the feasibility of eradicating brown rats from St Agnes and Gugh (Bell, 2011a). Due to eradication projects failing on other inhabited island elsewhere in the world (Oppel, et al., 2010), SLG required the feasibility assessment to include social and economic evaluation. It was not known how the community would feel about eradicating rats; whether they would feel the proposed action necessary, or how they would evaluate social, economic and health benefits of such an operation. If a person's values and sensitivities are dismissed, then they will not engage with operational processes which can jeopardise the whole project. The assessment had to focus on obtaining the opinions of all community members. The feasibility assessment was conducted by Wildlife Management International Ltd (WMIL; Bell, 2011a; 2011b)

The 'vision' for the project primarily focused on protecting Manx shearwaters and storm petrels, because rat eradication was considered the only land-based option that would feasibly increase the abundance of these species.

MATERIALS AND METHODS

Feasibility phase

The community firstly needed to understand that *control* of rats was not an option and that eradication was only a viable goal if all parties worked together. The feasibility study therefore set out to ascertain each resident's opinion on whether they would support the eradication of rats, what benefits they would expect for themselves and the community, and what would motivate them to keep the islands rat-free. Face-to-face interviews using a standard questionnaire were conducted with all adults on St Agnes and Gugh. Controversial topics are often better received if personnel are open to discuss less positive outcomes, acknowledging inherent risks and ethical challenges as it allows questions to be voiced and addressed from the outset (Crowley, et al., 2017a).The risks stated were (a) inconvenience (e.g. temporary or long-term changes in waste disposal, pet and livestock treatment), (b) time away from other activities due to volunteer participation during eradication and long term biosecurity and monitoring, (c) adjustment to new regulations (e.g. undergoing rodenticide training), (d) that economic benefits may take time and only apply to some community members, and (e) funding for eradication may come from grant funding, which communities may feel reduces the availability of financial resources for alternative projects.

In order to make their own decisions, community members each needed to have a full understanding of the technical aspects of the rat removal operation, and what their personal role in the project could potentially be. The feasibility assessment incorporated two general community workshops; a combined meeting with the six farmers to discuss the eradication in detail, covering aspects that were specifically relevant to stock, crops and farms as well as face-to-face meetings on each farm; visits to St Agnes School; and face-to-face meetings with representatives from each household. Every resident was asked to provide full details of their willingness to support a potential eradication and any stipulations they had. Achieving complete rat eradication was only part of the process, the legacy of the project was to keep the islands rat-free in perpetuity. The feasibility study therefore also set out to ascertain the willingness of each community member to carry out biosecurity measures in the long term.

Interim phase

While RSPB and AONB continued to deliver education work to invest in on-the-ground community relationships between June 2011 and January 2013, a Project Steering Group and Communications Group were formed.

Start of the project; preparation for 'rat-removal ready' phase

When funding was confirmed, the five-year 'Isles of Scilly Seabird Recovery Project' (IOSSRP) was launched. Two staff members were employed by RSPB, providing continuity for the community at each phase. Through a competitive tender process, WMIL were the successful contractors for rat eradication. Community conservation actions at this stage were named 'rat-removal ready' actions and were focused on reducing potential rat food and harbourage to a minimum, so rats could be easily detected and take bait when the eradication phase commenced.

The IOSSRP recognised the importance of monitoring the response of other species on St Agnes and Gugh following the eradication of brown rats and implemented a monitoring programme for birds, mammals (shrew and rabbits), invertebrates and vegetation. This work was completed under contract by Spalding Associates. Most species benefit from rat eradications on islands, but there have also been unforeseen and negative impacts recorded in several projects around the world (Courchamp, et al., 2003; Towns, et al., 2006; Bell, et al., 2011).

Eradication and short-term monitoring phase

The eradication delivered by WMIL, was a ground-based bait station operation using rodenticide over winter when natural food was minimal (Bell, 2019). Monitoring tools were used to detect any rats not taking bait or avoiding bait stations. Community members were required to assist WMIL with specific eradication activities such as checking bait stations in their own homes and reporting rat sightings. During the eradication phase, WMIL and IOSSRP personnel built good relationships with community members to create the best foundation for well-coordinated actions for on-going biosecurity and potential incursions in the future.

Post eradication monitoring and final check phase

WMIL produced a Biosecurity Plan and returned for a six-week 'final check' phase in winter 2016. IOSSRP personnel trained community members to assist with the monthly checks of the permanent biosecurity stations and surveillance after a 'rat on a rat' (ROAR) call (a 24-hour hotline based at IOSWT where anyone can report rat sign or a suspected rat sighting). A ROAR required a monitoring grid extending 300 m in all directions from the sighting spot with daily checks for a month (and was removed when no evidence of a rat was detected). During the 'final check', questionnaires as part of semi-structured interviews were carried out. The community questionnaire consisted of 22 socio-economic evaluation questions, 14 delivery questions and eight biodiversity questions. Semi-structured interviews represented feedback from the full population of St Agnes and Gugh. Qualitative analysis was deemed the best fit as interviews allow each person to express themselves, including personal narrative, and common themes can emerge (Crowley, et al., 2017a). It is known that successful eradication of rodents has turned some islands into attractions for visitors, facilitating the establishment of local tourism businesses (Oppel, et al., 2010). Therefore, specific questions were asked to ascertain whether tourism or other businesses had benefited on St Agnes and Gugh following the eradication of rats. During these interviews, community members who were able to commit to long-term biosecurity actions were registered with RSPB as Seabird Heritage Volunteers (SHVs) and were provided with additional training and support to complete these actions.

Long-term monitoring phase

The SHVs took ownership of their biosecurity roles to continue to keep the islands rat-free after the formal end of the IOSSRP project. SHV Coordinators were recruited in the community to coordinate these community volunteers and record data from each biosecurity action. An updated Biosecurity Plan for St Agnes and Gugh was prepared by IOSSRP with contributions from SHVs. A Maintenance Plan was written by the partners and the community aspects were 'sense-checked' by the SHVs.

RESULTS

Feasibility phase

All community members valued seabirds and supported the eradication of rats for the protection of seabirds. The collective support for the project was not solely for seabirds but for the added benefits to people (Bell, 2011a; Bell, 2011b). Rats were having an impact on the livelihood, health, enjoyment and lifestyle of the local community as well as the biodiversity of the island (Bell, 2011a; Bell, 2011b). Farmers reported rats were damaging crops and taking or damaging stock food, fishermen reported rat damage to lobster pots and nets and the campsite suffered damage to tents and customer's food and belongings. Over ¾ of residents reported rats entering their houses. It was estimated that rats were costing the St Agnes and Gugh community approximately £15,000 per year (between £10 and £1,000 per household per year), due to purchasing bait and damage to property and goods (Bell, 2011a; Bell, 2011b).

While explaining that *'the decision to carry out the project is yours'*, the eradication methodology and actions were discussed with the community to ensure that they had all the information needed to make the decision of whether to proceed with the project or not. This gave the community an opportunity to air concerns such as finding adequate funding (86% of residents), incorrect waste management causing eradication failure (80% of residents) and community involvement and support (77% of residents). These concerns were addressed or actions to mitigate these concerns were outlined including information on possible funding streams; bespoke waste training at each property, eatery and farm; provision of rat-proof garbage bins and composters; revised process for waste collection and removal to St Mary's; and the communication strategy (including a 24-hour call line).

Interim phase

A number of activities were completed during the interim phase including putting 'Frequently Asked Questions' on project partner websites, addressing community and wider community questions; delivering two press releases; education and outreach activities on how to detect rats and shrews and providing funding updates to the community. Funding applications were completed and included fully-costed mitigation options for identified issues collected during the feasibility assessment.

Preparation 'rat-removal ready' phase

A five-year 'activity programme' was developed for the community and visitors. A full audit of St Agnes and Gugh was carried out in June 2013 to prepare the islands and provide final 'rat- removal ready' instructions to all residents as requested by the community during the feasibility assessment.

Eradication and short-term monitoring phase

All community members allowed daily access to property for WMIL personnel to carry out the ground-based bait station eradication using rodenticide (either Contrac®, containing the anticoagulant bromadiolone at 0.005% w/w or Roban Excel®, containing the anticoagulant difenacoum at 0.005% w/w) in more than 1,000 stations between October 2013 to March 2014 (Bell, et al., 2019). There was no rat-sign after three and a half weeks (Bell, et al., 2019).

There were no instances of non-target species being affected by the bait (Bell, et al., 2019). Nine rats were picked up above the surface; six of them were discovered by community members, and eradication personnel responded immediately by collecting the carcasses (Bell, et al., 2019). WMIL trained IOSSRP personnel to gain expertise in eradication techniques, which enabled them to further support the community for the later phases. WMIL and IOSSRP personnel delivered the activity programme which included two community update talks, weekly update newsletters and school education sessions.

The eradication methods were reviewed throughout by the Project Steering Group and adaptations were made when necessary. On farms a number of baiting tunnels were dislodged by stock (no bait was consumed) and a number of monitoring stations (i.e. non-toxic flavoured wax) were eaten by cows, so farmers and WMIL liaised to organise a rotation of paddocks where cows would graze, allowing tunnels and monitoring tools to be moved in and out of these areas at certain times and remain intact (Bell, et al., 2019).

Post eradication monitoring and final check phase

Monitoring of the key species showed breeding success for the first time in living memory post-eradication. There were eight Manx shearwater chicks recorded in 2015 and 32 in 2015. Storm petrels returned to breed in 2016 with nine breeding pairs recorded. IOSSRP personnel trained 12 community members to assist checking the permanent monitoring stations and surveillance from 'rat on a rat' (ROAR) calls (Fig. 2).

Fig. 2 IOSSRP personnel train the SHV Coordinators in biosecurity methods. Credit Nick Tomalin.

There were 28 ROAR reports during this post-eradication monitoring phase. Community members assisted the IOSSRP team establish and maintain the ROAR surveillance grid. After the final check was completed, it was deemed appropriate to adapt a ROAR response to the community checking the permanent biosecurity stations only instead of establishing and maintaining a 300 m wide monitoring grid (unless additional evidence of a rat was identified).

The questionnaire responses showed that the entire community felt the eradication had a positive effect on the island and the community (Tables 1–7).When asked what they liked about the project, 31% of the community enjoyed having the eradication team on the islands, 15% liked having no rats on the islands any longer, 10% liked the eradication team and community working together towards the successful completion of the eradication and 5% liked how the project worked closely with the St Agnes School (Table 1). The community thought the project gathered the island together and allowed everyone to work together towards a common goal (Table 2). Half the residents felt that this project had made a positive change to the history for the island including raising cultural awareness of the seabirds and their importance to St Agnes and Gugh and the Isles of Scilly (Table 3). All of the community felt that the project had benefited the economy of the island, with several businesses on the island directly benefitting during and after the eradication (Tables 4 and 5).

Table 1 Response 'themes' from St Agnes and Gugh community members (shown as number of people and percent of the community) to the question 'What did you like about the project?'

Theme of reply	No.	%	Social (S) Biodiversity (B) or Delivery (D) theme
The team being on the islands – nice people to have around	18	31%	S
No rats	9	15%	B
Team and community working together for a common goal	6	10%	S
Team were unobtrusive and respectful which made the experience enjoyable	6	10%	S
The project worked with the school	5	8%	S
Manx shearwaters and storm petrels breeding success	4	7%	B
Team helped me learn about wider island biodiversity	4	7%	B
Learnt about rats and their ecology	2	3%	S
The eradication was professionally delivered	2	3%	D
Like to see the bait-take in real time and the speed of operation in daily updates from the team and in newsletters	2	3%	D
I was sceptical at the start but was proved wrong, complete eradication is possible	1	2%	D

Table 2 Response 'themes' from St Agnes and Gugh community members (shown as number of people and percent of the community) to the question 'Do you think there have been any positive or negative impacts to community by the removal of rats from St Agnes and Gugh?'.

Theme of reply	No.	%
Negative or no impact	0	0%
Positive (no further comment)	16	28%
Positive, community no longer needs to worry about damage caused by rats	14	24%
Positive impacts for farms and visitor accommodation	12	21%
Positive, the project generated interest in the community	7	12%
Positive as it was nice for the community to have the team on the islands in winter	4	7%
Positive, the community was united and not divided in any way, it was a community project	3	5%
Positive, due to the school and children being involved throughout	1	2%
Positive, apart from the increase in rabbits which is negative for farmers	1	2%

Table 3 Response 'themes' from St Agnes and Gugh community members (shown as number of people and percent of the community) to the question 'Do you think there have been any positive or negative impacts to culture/history by the removal of rats from St Agnes and Gugh?'

Theme of reply	No.	%
No impact	29	50%
Positive, as we are making history here on St Agnes	10	17%
Positive impact (no further comment)	7	12%
Positive, culturally we have all worked together as a community	3	5%
Positive, raised cultural awareness of where birds are in our history, memory, collective consciousness, part birds played in our community. Better for historical buildings	2	3%
Positive, as part of our history that we bought rats over and now we are putting our mistake right	2	3%
Positive, we have better waste management and awareness of how to think carefully about staying rat-free	2	3%
Positive, as the project will reinstate historical bird lovers	2	3%
Positive, we can look back and feel proud. I have kept all articles about the project for a community scrapbook to help us remember details correctly.	1	2%

Table 4 Response 'themes' from St Agnes and Gugh community members (shown as number of people and percent of the community) to the question 'Do you think there have been any positive or negative impacts to economy by the removal of rats from St Agnes and Gugh?'

Theme of reply	No.	%
Positive (no further comment)	17	29%
Positive in respect to what other community members have told them, but not personally to them	12	21%
Farmers and/or fishermen will not lose profits from rat damage	8	14%
The project itself brought extra business to the islands (using accommodation/ shop/boats)	6	10%
Don't have to spend money on rat control and damage	5	9%
More boating/bird tours	4	7%
More visitors due to not having rats in lets/tents	3	5%
More visitors in the future if we market the islands as 'rat- free'	3	5%

Table 5 Response 'themes' from St Agnes and Gugh community members (shown as number of people and percent of the community) to the question 'Has your business benefited from this project?

Theme of reply	No.	%
Not applicable	19	32%
No longer have to worry about rat damage to any goods	12	20%
More tourists in holiday lets and accommodation as a result of media exposure	4	7%
Profit in the shop, accommodation	3	5%
Business is now more hygienic and safe for visitors without rats	3	5%
The project team used the boats more, visitors on wildlife trips have increased by 200%, there has been more publicity through the project	3	5%
Yes (no further comment)	3	5%
Composter and bins provided by the project have benefited business	2	3%
Tourists have a more positive experience on the islands	2	3%
Yes, more visitors camping and buying ice-cream as they know the campsite is rat-free	2	3%
The WMIL team using holiday lets	2	3%
Possible knock-on effect as more visitors	1	2%
Guests are actively interested in the project, improving their stay	1	2%
Team bought eggs	1	2%
'Lifelong learning' has benefited from walks and talks	1	2%

Table 6 Response 'themes' from St Agnes and Gugh community members (shown as number of people and percent of the community) to the question 'Do you think there have been any positive or negative impacts to tourism by the removal of rats from St Agnes and Gugh?

Theme of reply	No	%
Positive (no further comment)	14	24%
Positive in respect to what other community members have told them, but not personally to them	4	7%
Positive, visitors' experience of the islands could be negative due to rats in tents/lets/on beaches	21	36%
Positive, more birdwatcher and tourists will visit to see more seabirds	12	21%
Positive if we market being rat-free more to visitors	4	7%
Positive, already heard good feedback from visitors	2	3%
Positive, the project has already promoted the islands as a travel destination, tourists told me they were here as they saw the project/islands on BBC Countryfile	1	2%
Positive, seabird boat tours have had far more visitors onboard due to the project, my business has a 10% increase in turnover due to the project	1	2%

Long-term monitoring phase

Legacy workshops held in 2016 confirmed the role of the SHVs in the on-going biosecurity of St Agnes and Gugh. Quarterly biosecurity monitoring completed by the community SHVs to date has not detected any rats (J. Peacock, St Agnes, pers. comm.).

DISCUSSION

Feasibility phase

Community 'stipulations' or requirements to address concerns were developed following the questionnaire and face-to-face interviews.

The community requested updates on funding opportunities, waste training and provision of bins and composters, a bespoke audit of actions to get the islands 'rat-removal ready' and clear communication lines between the eradication team and the community through community talks, face-to-face dialogue, newsletters and school education visits.

As most residents had concerns over the health and safety of the children, a 'school education day' was delivered whereby school children saw snap traps, bait, tube and lockable bait stations and received training on how to stay safe (Fig. 3). Concerns about personnel whom residents didn't recognise being on their land were resolved by WMIL suggesting that all personnel wear an identifiable uniform (i.e. blaze orange hats with the project logo). Concerns over where the money for travel and subsistence for the eradication team would be spent were answered by WMIL assuring residents that much of it would be spent on St Agnes and Gugh using local providers (i.e. purchasing milk and eggs from the local farmers and supplies from the St Agnes Store). Concerns were also expressed over the potential poisoning of non-target species, particularly pet cats (24 were present during operation) and dogs (four were present). The safety of pets is always a concern to owners, so the mitigation information was provided sensitively, including explanation of the unlikelihood of accidental poisoning due to the design of the bait station and unlikely access to the rodenticide. Pet owners were given information that the antidote to the anticoagulant rodenticide (vitamin K injections and tablets) would be stored on St Agnes, with WMIL personnel being contactable 24 hours a day throughout the operational phase to administer the antidote if necessary. Residents were asked to alert eradication personnel of any dead rats found above the surface so the carcasses could be retrieved immediately.

Table 7 Response 'themes' from St Agnes and Gugh community members (shown as percent of the community) to the question 'What support will you offer the project?' asked in 2010 (during feasibility phase questionnaires) and 2017 (long-term monitoring phase).

Theme of reply	% of community members changed from 'No' in 2010 to 'Yes' in 2017 (descending order)
In-kind logistical support	59%
Other (mainly in-kind support such as lifts in vehicles)	58%
Volunteering time	55%
Training in rodent detection and identification	41%
Long-term monitoring for rodents	37%
Training in interview and site inspection procedures and methods	30%
Assisting with any contingency operation	22%
Check for rodent damage to your own cargo	19%
Written support to decision makers (e.g. funders, councillors, MPs).	19%
Listed as a reporting location (where any rat sighting is reported to you for action)	17%
Transporting food to and between islands in rodent-proof containers	17%
Installing and maintaining a bait station on your vessel and/or property	13%
Partner to the project	No change
Financial support	No change

Several residents raised the issue regarding the possible impacts of rat eradication on the wider ecology of the islands; in particular in regards to the endemic 'Scilly shrew' consuming bait; rabbits consuming bait during the operation as well as potentially increasing rapidly after the eradication; birds eating the bait; cats prey-switching from rats to other species such as birds. WMIL explained the long-term monitoring and mitigation options for these species such as providing diet information for Scilly shrews (insectivorous diet as opposed to cereal-based diet); mitigation methods for rabbits including additional wires on either side of the bait stations to reduce access, and rabbit control by the community as necessary after the eradication; mitigation methods for birds including bait station design preventing access; and mitigation methods for all non-target species including daily careful monitoring of bait blocks for signs of non-target species consumption, re-sighting of bait stations and the use of 'crow-clips' (which further prevent entry by birds such as gulls and corvids). It was recommended that no new cats come to the island if previous cats were originally kept as ‘ratters’, and collars and bells should be used for all pet cats. Two residents also struggled with the ethical dilemma of eradicating a species but decided that the threat to seabirds was of larger concern, and the complete eradication of rats was the only viable solution to remove the threat to seabirds on the islands.

The feasibility report (Bell, 2011a; Bell, 2011b) detailed the ‘technical conservation actions’ required and confirmed that the entire community on St Agnes and Gugh were supportive and willing to carry out general and bespoke actions.

Start of the project; preparation for ‘rat-removal ready’ phase

Before the eradication phase, the community helped complete a number of required actions including the cessation of any baiting for 12 months prior (snap traps were supplied for local control). Livestock food and bedding on the six farms was reduced to minimum levels and rat-proof feed storage systems were implemented. To ensure there were no areas without bait, livestock pens, paddocks fences, windbreaks and stone walls were mapped using GIS to ensure complete bait station coverage. Where possible, farmers carried out these necessary actions, but any work not completed was carried out by WMIL and IOSSRP personnel the month before the eradication.

Residents’ waste management practices were improved by the provision of new bins and composters as part of ‘Bin Friendly Days’. ‘Shed clearance days’, ‘beach clean days’ and ‘wood collection and bonfire night’ reduced rat food and harbourage around the islands. The St Agnes School held an ‘Apple Day’ to remove wind-fallen apples from the ground. Rats were trapped for resistance testing to confirm final bait choice for the eradication. Index trapping results estimated the rat population on St Agnes and Gugh to be between 3,000 and 3,500 rats. Any restrictions or sensitivities in regard to accessing peoples land and properties was obtained.

Fig. 3 Bait awareness workshop with St Agnes School children.

As entrance to St Agnes and Gugh via boats is not regulated by any authority, this presented the highest risk pathway for biosecurity. Talks to all community members and the Harbour Users Group (for all boat users on Scilly) regarding biosecurity requirements and vigilance were held throughout the project.

Eradication and short-term monitoring phase

The contractors, team members and community members worked well together to ensure complete eradication of rats which would be confirmed after a further two-year check.

Post eradication monitoring and final check phase

Various themes emerged from the post-eradication interviews which are summarised in Tables 1–7, including:

Social: The entire community felt the project had positively affected their day-to-day life. A strong theme was they no longer needed to worry about rats *“They used to be on my mind, worrying about where they are and what they do”.* Most of the community (86%) felt the removal of rats had improved health due to the reduction of diseases spread by rats. When asked ‘what did you like most about the project?’ eleven themes developed with social-themed responses being most popular (Table 1). When asked ‘what did you dislike most about the project?’ the answer ‘nothing’ was overwhelmingly the most popular answer with three other themes (increase in other nuisance species, ethical dilemma and concern about accidental pet poisoning) being mentioned, however they felt that each concern had been mitigated against (Table 1). When asked if the project had any positive or negative impacts on the community, 100% answered ‘positive’ (Table 2). One theme that stood out was that ‘*the community was united and not divided*

Fig. 4 WMIL training community member to store bait box safely. Credit Alastair Wilson.

in any way, it was a community project'. When asked if there had been any impacts to culture and history (Table 3), one person said, "*It has raised cultural awareness of where birds are in our history, memory, collective consciousness and the part birds played in our community"*.

Economy: Again, the entire population felt the project had benefited the local economy (Table 4), with most of this benefit to certain sectors; agricultural, fishing and particularly tourism and that the benefits had potential to increase. Over two-thirds of the community (68%) felt that their businesses had benefited from the project (Table 5). A section of the community (17%) had developed new products; e.g. one farmer explained that '*Apple day had been the catalyst to a new apple juice product and cider products he developed'*. Another community member explained that *'visitors on his 'boating wildlife trips' had increased by 200%, as there has been high publicity of the project, combined with interpretation resources, so he could offer improved tours"*. Publicity was an added benefit to the project, which was not originally anticipated by the IOSSRP 'activity programme'. Shows such as BBC 'Countryfile', BBC 'One Show', BBC 'Springwatch' and a German wildlife show, were viewed by approximately 20 million viewers in total (pers. comms.) and directly led to increased tourism with one community member saying '*A tourist told me they had visited due to seeing the project on BBC 'Countryfile'.* Tourism generates the largest income on the island (Blue Sail, 2011), and 100% of the population felt the project had a positive impact on tourism (Table 6).

Interestingly, once rats had been eradicated, more residents (94% in 2016 compared to 76% in 2010) recognised that they had been having a greater issue with rats than first thought, regarding damage, and on reflection the cost rats had caused them was revised as being higher (Table 7).

Biodiversity: Compared to the 2012 questionnaire, the number of residents being sympathetic to seabirds had increased by 47% (Table 7). Regarding the wider species present on St Agnes and Gugh, none of the community felt that the eradication of rats had any negative impact on any non-target species.

Project procedures and delivery: All of the community were happy with the project procedures and methods (Table 1). When asked if it was helpful having WMIL team members assisting 'rat-removal ready' action 'shed clearance' one person said: *'it generated goodwill in the community and got everyone on board with the project'*. When asked whether the different communication methods were correct, the entire community said yes. Common themes were, '*clear explanation of what we needed to do and when*', '*involved everyone and engagement with all children at the school*', '*the team was passionate about the cause*', '*we felt listened to, as things were altered if we asked for them to be*'.

The final questions asked what support the community could give to future biosecurity to keep the islands rat-free. More residents were willing to offer support compared to 2010 (Table 7). An additional 20 community members said they would volunteer to assist biosecurity monitoring, due to being proud of the project and wanting to play their part to keep the island rat-free. A total of 32 community members have registered with RSPB as 'Seabird Heritage Volunteers' (SHVs).

Long-term monitoring phase

The role of the SHVs was confirmed as covering five tasks; (1) checking permanent monitoring stations once a month; (2) sustaining biosecurity on boats and freight; (3) carrying out surveillance for potential incursions (within 24 hours of a 'ROAR' call'); (4) assisting with incursion response baiting; and (5) assisting with the ecological monitoring of the key species.

Each SHV received LANTRA rodenticide training as well as bespoke training for incursion response protocols; a social media (Facebook) group was set up as a mechanism to send monthly check information to the SHV coordinator; biosecurity protocols were reaffirmed; and incursion response methodology was revised (i.e. check all biosecurity stations within 24 hours of a 'ROAR' especially those with the stations nearest to the report location and report back to the SHV coordinator) and tested by a 'mock incursion response' exercise.

If rat-sign is found at any time in the future, the SHV coordinator will inform IOSWT on St Mary's and the RSPB Conservation Officer in Penzance. The SHVs will swap monitoring wax for rodenticide in their biosecurity stations within 24 hours and report any new rat sign. An RSPB-coordinated incursion team will arrive to assist incursion response baiting for one month, with the SHVs assisting where possible.

In addition to biosecurity monitoring, SHVs assist IOSSRP personnel and IOSWT contractors to survey Manx shearwater and storm petrel breeding sites (using play-back at burrows) and 'evening chick-check walks'.

The IOSWT has committed to fund the work outlined in the 'St Agnes and Gugh Maintenance Plan', including, but not limited to, ongoing biosecurity training for the community, seabird surveys and resources required to keep the biosecurity shed functional.

CONCLUSION

The success of this project was due to three factors; the vision for the sustainability and legacy of the project from concept; robust preparation; and being 'community based'. Community members joined decision-making processes from the offset, and in advance of this, a decade of preparation activities meant relationships had started to be built and methods on how to protect seabird heritage had started to be shared. These relationships then sustained trust through the 'rat-removal ready actions' and eradication phase, enlisting an excellent contractor and team whom worked with the community addressing all stipulations, and having available, adaptive project staff to accommodate community concerns when required. Community members therefore felt listened to and valued.

The IOSSRP experience shows that, to ensure that an island restoration project on an inhabited island runs successfully, the support and agreement from the community must be secured. It is vital that access to all properties is obtained to effectively carry out an eradication. The community must share the project's vision and feel that they are one of the beneficiaries. To do this, they will need to be included in the decision-making process and management of the project. In this way the legacy of the project will be much stronger. The larger the community, the longer, potentially, the project managers will need to ensure that the residents are all at the same position of understanding through the various stages of the project. Archipelagos or groups of islands bring additional stakeholders and interested parties that need to be engaged compared to single islands. Ten years is not an unreasonable timescale depending upon the starting point, the value placed upon seabirds by the community, and the strength of the project partnership.

It is important to recognise the social science requirements for eradications planned on inhabited islands. The views and concerns of each and every resident and stakeholder group are important. Community engagement and consultation should be completed during every stage of an operation. Most importantly, all aspects of the eradication should be debated with the community in the early stages of the proposal. Unlike eradication operators, most members of the public do not have any knowledge of the principles and techniques of an eradication, particularly in regard to rodenticide choice and operational procedures. It is important that each community member understands these aspects and how they will be affected by the day-to-day operational requirements. A lack of public awareness about invasive species impacts and misunderstanding of eradication techniques from island communities are thought to have been responsible for the opposition of proposed eradications on inhabited islands around the world and investing in greater education and consultation effort can ensure a suitable environment for eradication projects to proceed (Bryce, et al., 2011).

The additional, and more unusual, preparations which were required on St Agnes and Gugh (e.g. clearing sheds, communicating with school children and taking precautions to ensure the safety of the community's pets) were essential and would have contributed to what was effectively a three and half week eradication period. Maybe even more importantly, was that these activities were a possible turning point for the community, where they recognised what was involved for the whole project to be successful. The methods used in this project ensured the community knew that staff and community were part of a team striving for the same goal, which would be challenging, but rewarding for birds and people.

The defining factors underpinning the success of the IOSSRP were the professional management of the eradication, dedicated and passionate volunteer team involvement, efficient and systematic monitoring, adapting to local conditions and ensuring a community-inclusive approach. The trust and knowledge the community gained during the preparation and eradication phase paired with the positive impacts the eradication of rats had on the seabirds and socio-economics for the community turned into 'pride and ownership' of *their* project.

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REFERENCES

Bell, M.D., Bullock, I. and Humpridge, R. (2000). *The Eradication of Rats from Ramsey Island, Wales*. Unpublished technical report to Royal Society for the Protection of Birds, South-west Regional Office, Devon, UK. Blenheim, New Zealand: Wildlife Management International Ltd.

Bell, E. (2004). *Lundy Seabird Recovery Project 2002–2004: The Eradication of Black and Brown Rats from Lundy Island*. Unpublished client report to the Royal Society for the Protection of Birds, South-West Regional Office, Devon, UK. Blenheim, New Zealand: Wildlife Management International Ltd.

Bell, E.A. (2011a). *Isles of Scilly Seabird Recovery Project: Summary Report: Improving Rodent Control on Uninhabited Islands, Assessment of the Feasibility of Rat Removal across the Isles of Scilly Archipelago and Feasibility of Rat Removal from St Agnes and Gugh*. Unpublished client report to Isles of Scilly Seabird Recovery Project Partnership. Blenheim, New Zealand: Wildlife Management International Ltd.

Bell, E.A. (2011b). *Isles of Scilly Seabird Recovery Project: Part 3: Assessing the Feasibility of Removing Brown Rats from St Agnes and Gugh, Isles of Scilly*. Unpublished client report to Isles of Scilly Seabird Recovery Project Partnership. Blenheim, New Zealand: Wildlife Management International Ltd.

Bell, E.A., Boyle, D., Floyd, K., Garner-Richards, P., Swann, B., Luxmoore, R., Patterson, A. and Thomas, R. (2011). 'The ground-based eradication of Norway rats (*Rattus norvegicus*) from the Isle of Canna, Inner Hebrides, Scotland'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 269–274. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Bell, E.A. (2019). 'It's Not All Up in the Air: the development and use of ground-based rat eradication techniques in the UK'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 79–87. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Bell, E., Floyd, K., Boyle, D., Pearson, J., St Pierre, P., Lock, L., Mason, S., McCarthy, R. and Garratt, W. (2019). 'The Isles of Scilly seabird restoration project: the eradication of brown rats *Rattus norvegicus* on the inhabited islands of St Agnes and Gugh, Isles of Scilly'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 88–94. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Blue Sail. (2011). *Future of Tourism on Scilly*. UK: Blue Sail.

Bryce, R., Oliver, M.K., Davies, L., Gray, H., Urquhart, J. and Lambin, X. (2011). 'Turning back the tide of American mink invasion at an unprecedented scale through community participation and adaptive management.' *Biological Conservation* 144: 575–583.

Campbell, K.J. Beek, J., Eason, C.T., Glen, A.S., Godwin, G., Gould, F., Holmes, N.D. Howald, G.R., Madden, F.M., Ponder, J.B., Threadgill, D.W., Wegmann, A. and Baxter, G.S. (2015). 'The next generation of rodent eradications: Innovative technologies and tools to improve species specificity and increase their feasibility on islands.' *Biological Conservation* 185: 47–58.

Courchamp, F. Chapius, J-L. and Pascal, M. (2003). 'Mammal invaders on islands: impacts, control and control impact.' *Biological Review* 78: 347–383.

Crowley. S., Hinchliffe. S. and McDonald. R. (2017a). 'Invasive species management will benefit from social impact assessment'. *Journal of Applied Ecology* 54(2): 351–357.

Crowley. S., Hinchliffe. S. and McDonald. R. (2017b). 'Conflict in invasive species management'. *Frontiers in Ecology and the Environment* 15(3): 133–141.

Eason. C., Miller. A., Ogilvie. S. and Fairweather. A. (2011). 'An updated review of the toxicology and ecotoxicology of sodium fluoroacetate (1080) in relation to its use as a pest control tool in New Zealand'. *New Zealand Journal of Ecology* 35(1): 1–20.

Howald, G., Donlan, C.J., Galvan, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21: 1258–1268.

Lock, L., Brown, A., Clitherow, J., Mawer, D. and St. Pierre, P. (2006*).* 'Isles of Scilly Seabird Conservation Strategy 2006–2009'. Paper delivered at the *Isles of Scilly Seabird Liaison Group, Isles of Scilly, UK, 20* October 2006.

Lock, L., Brown, A., Webber, J., Mawer, D. and St. Pierre, P. (2009*).* 'Isles of Scilly Seabird Conservation Strategy 2009–2013'. Paper delivered at the *Isles of Scilly Seabird Liaison Group, Isles of Scilly, UK, 25* October 2009.

McCann, J. (2005). 'The introduction of the brown rat (*Rattus norvegicus*)'. *Somerset Archaeology and Natural History Journal* 2005: 139–141.

Oppel, S., Beaven, B., Bolton, M., Vickery, J., and Bodey, T. (2010). 'Eradication of invasive mammals on islands inhabited by humans and domestic animals.' *Conservation Biology* 25: 232–240.

Parslow, R. (2007). *The Isles of Scilly*. Yorkshire, UK: Collins New Naturalist.

Stanbury, A., Thomas, S., Aegerter, J., Brown, A., Bullock, D., Easton, M., Lock, L., Luxmoore, R., Roy, S., Whitaker, S. and Oppel, S. (2017). 'Prioritising islands in the United Kingdom and crown dependencies for the eradication of invasive alien vertebrates and rodent biosecurity.' *European Journal of Wildlife Research* 63: 31.

St Pierre, P., Mason, S., Sugar, K., Buckley, P., Heaney, V., and Pearson, J. (2014.) 'The Isles of Scilly Seabird Conservation Strategy 2014–2018'. Paper delivered at the *Isles of Scilly Seabird Liaison Group, Isles of Scilly, UK.*

Thomas, S., Brown, A., Bullock, D., Lock, L., Luxmoore, R., Roy, S., Stanbury, A. and Varnham, K. (2017). 'Island restoration in the UK – past present and future'. *British Wildlife* 28: 231–242.

Towns, D. R. and Broome, K. G. (2003). 'From small Maria to massive Campbell: Forty years of rat eradications from New Zealand islands'. *New Zealand Journal of Zoology* 30: 377–398.

Towns, R.T., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?' *Biological Invasions* 8: 863–891.

Walsh, A., Wilson, A., Bower, H., McClelland, P. and Pearson, J. (2019). 'Winning the hearts and minds – proceeding to implementation of the Lord Howe Island rodent eradication project: a case study'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge*, pp. 522–530. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Zonfrillo, B. (2001). 'Wildlife Conservation on Ailsa Craig'. *The Thomas Duncan Memorial Lectures. Friends of the McKechnie Institute* 22: 1.

A plan for the eradication of invasive alien species from Arctic islands

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Abstract Invasive alien species represent one of the most significant threats to Arctic ecosystems and their inhabitants. Rapidly changing environmental conditions and a growing interest in resource extraction, settlement and tourism make the Arctic region particularly vulnerable to biological invasion. For this reason, invasive alien species are of substantial concern to the Arctic Council, a multi-national body comprised of Canada, the Kingdom of Denmark (including Greenland and the Faroe Islands), Finland, Iceland, Norway, Russia, Sweden, and the United States, as well as six international organisations that represent Arctic indigenous peoples as Permanent Participants. The Arctic Council's *Arctic Invasive Alien Species (ARIAS) Strategy and Action Plan* includes the priority to: "actively facilitate the eradication of invasive alien species from island ecosystems throughout the Arctic, as well as the recovery of native island species and habitats that have been impacted by invasive alien species." A multi-national team of governmental and non-government partners is collaborating in the development of an action plan (hereafter 'islands plan') for the eradication of invasive alien species from Arctic island ecosystems. The intent of the plan is to provide a vision and strategy for a region-wide approach to the eradication of island invasive alien species as a multi-national commitment. The islands plan will set forth a strategy for prioritising island eradications consistent with the growing pressures on ecological and cultural systems. We have a unique opportunity in the Arctic to take decisive action to prevent and mitigate the adverse impacts of invasive alien species that plague much of the rest of the world. The eradication of invasive alien species from islands in other parts of the world provides useful insights into best practices, including approaches to prioritisation and cost-effectiveness.

Keywords: Arctic Council, invasive alien species, non-native species, policy, prioritisation

INTRODUCTION

Throughout the world, invasive alien species have driven the endangerment and extinction of a wide range of plants and animals (Wilcove, et al., 1998; McNeely, et al., 2001; Bellard, et al., 2016), contributed to the degradation of freshwater, marine, and terrestrial ecosystems (Howard, 1999; Rahel & Olden, 2008; Pejchar & Mooney, 2009) and hastened the alteration of ecological cycles (Chapin, et al., 2000; Towns, et al., 2006; Kurle, et al., 2008; Doherty, et al., 2015). Invasive alien species place constraints on a wide range of ecosystem services that underpin human well-being and economic growth, such as pollination, food and fibre production, disease prevention, climate resilience, and recreational opportunities (Mack, et al., 2000; Mooney & Hobbs, 2000; McNeely, 2001; Ehrenfeld, 2010; Simberloff, 2011). Invasive alien species are regarded as a threat to national security; in addition to undermining food, water, and energy security, they may impede military readiness or cultural survival of native peoples (White House, 2016).

Three primary factors make islands particularly vulnerable to the impacts of invasive species: geographic isolation, size and high percentage of global biodiversity per area (Reaser, et al., 2007; Kier, et al., 2009). While relatively few invasive alien species have been documented in the Arctic region (Fig.1) and there is currently no systematic effort to build a comprehensive dataset and thus provide species lists, biological invasion is expected to increase in concert with increasing human activity and climate change (Walther, et al., 2009; Hall, et al., 2010; Bennett, et al., 2015). The threat that invasive alien species pose to Arctic island ecosystems is thus of growing concern (Meltofte, 2013). Fortunately, Arctic governments and their partners still have the opportunity to act decisively to prevent and mitigate the adverse impacts of invasive alien species that plague much of the rest of the world.

ECOLOGICAL CONTEXT

More than 21,000 species of mammals, birds, fish, amphibians, reptiles, invertebrates, plants, and fungi are native to the Arctic. Highly charismatic species include the polar bear (*Ursus maritimus*), narwhal (*Monodon monoceros*), caribou/reindeer (*Rangifer tarandus*), and snowy owl (*Bubo scandiaca*). The Arctic is characterised by extreme seasonality; many species migrate long distances in order to follow resource productivity, some species by the millions. Although Arctic ecosystems are low in species richness, abundance is often high (e.g. sea birds) (Meltofte, 2013; Fernandez, et al., 2014).

Fig. 1 The Arctic Region. There are varying approaches to defining the Arctic according to geophysical, ecological, or political criteria. For the purposes of this paper, the CAFF delineation of the Arctic is used (including 32 million km²).

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Invasion pathways of particular concern in the Arctic include: commercial shipping (i.e. introductions via ballast water, hull biofouling); the introduction of organisms and reproductive material through horticulture and aquaculture activities; large-scale tree planting for aesthetics, fuel, windbreaks, and carbon sequestration; transport of contaminated material and equipment for energy development and mineral exploration; and tourism, including recreational hunting and fishing (e.g. through contaminated boats, equipment, and gear). Examples of other anthropogenic pathways include translocated piers, docks and pilings, marine debris and the release or escape of live animals (e.g. from fur farms or the commensal rodents (*Mus* spp., *Rattus* spp.) inadvertently transported to the islands) (CAFF, 2017). Table 1 provides examples of specific pathways of introduction, the species introduced and the implications for the Arctic. At this time, data are insufficient to develop a comprehensive list of non-native species in the Arctic.

SOCIO-ECONOMIC AND POLITICAL CONTEXT

Numerous people, those who reside in the Arctic and many who do not, benefit from the region's natural resources. Approximately four million people live in the Arctic, including indigenous peoples who depend upon subsistence gathering and harvesting of native species as a major source of their daily food intake and as a vital element of their culture. Each year, commercial fisheries harvest millions of tons of native marine organisms valued in the billions of U.S. dollars (Christiansen & Reist, 2013; Sundet, 2014).

Extractive industries (e.g. oil, gas, and minerals) are already well-established in the region and are expanding their activities as melting ice makes access to natural resources more feasible. The increase in rate and numbers of commercial investments in the Arctic is expected to increase the risk of biological invasion into and throughout the region (Emerson & Lahn, 2012; Miller & Ruiz, 2014; Eguíluz, et al., 2016).

Invasive alien species do not respect jurisdictional boundaries. Effective communication and collaboration with neighbouring countries, stakeholders, and trading partners is of paramount importance in the prevention, eradication, and control of invasive alien species in the Arctic. The Arctic Council—a policy framework that includes Arctic Council member countries (known as States), Permanent Participants (Arctic indigenous communities), and Observers (generally, non-member States)—recognises the connection between economic well-being, social stability, and environmental health. The Council actively promotes cooperation, both within the Arctic and globally, to address the environmental changes facing the region (Arctic Council, 2013), ideally through an ecosystem-based approach which balances conservation and sustainable use of the environment (PAME, 2011).

The *Arctic Biodiversity Assessment's* findings (Meltofte, 2013; Box 1) have served as the impetus for the Arctic Council's programme of work on invasive alien species. In May 2017, the Council adopted the *Arctic Invasive Alien Species (ARIAS) Strategy and Action Plan* (CAFF, 2017). This document is a call to action voiced by Arctic nations;

The Arctic Biodiversity Assessment

The *Arctic Biodiversity Assessment* (Meltofte, 2013) recognises that there are currently few invasive alien species in the Arctic, and underscores that more are expected with climate change and increased human activity. Authors recommended:

"Reducing the threat of invasive alien/non-native species to the Arctic by developing and implementing common measures for early detection and reporting, identifying and blocking pathways of introduction, and sharing best practices and techniques for monitoring, eradication and control. This includes supporting international efforts currently underway, for example those of the International Maritime Organization to effectively treat ballast water to clean and treat ship hulls and drilling rigs. (Recommendation 9)"

Actions for Arctic Biodiversity: Implementing the Actions of the Arctic Biodiversity Assessment 2013–2021 (CAFF, 2015) sets forth two actions to address Arctic invasive alien species:

Action 9.1 (2015-2017): Develop a strategy for the prevention and management of invasive species across the Arctic, including the identification and mitigation of pathways of introduction of invasions. Include involvement of indigenous observing networks, which include invasive and new species reporting, to assist with early detection.

Action 9.2 (2017-2019): Incorporate common protocols for early detection and reporting of non-native invasive species in the Arctic into CAFF's Circumpolar Biodiversity Monitoring Programme (CBMP).

Table 1 Examples of introduction pathways and impacts.

Pathway	Species	Impact(s)
Escape from fur farms	American mink (*Mustela vison*)	High predation on native species in Iceland and Scandinavia (Birnbaum, 2013)
Gardening and land reclamation	Nootka lupine (*Lupinus nootkatensis*)	Successful competition against native plants that has changed the ecological structure and function in Iceland (Magnusson, 2010)
Intentional releases into the natural environment for food	Red king crab (*Paralithodes camtschaticus*)	Effective predation of a wide range of marine species in some Norwegian fjords (Oug, et al., 2011)
Intentional releases into the natural environment for hunting	Raccoon dog (*Nyctereutes procyonoides*)	Effective predation of ground-nesting birds and amphibians, and service as a vector of rabies and other pathogens and parasites in northern Scandinavia (Sutor, et al., 2010; Kowalczyk, 2014; Dahl & Åhlén, 2016)

Table 2 Arctic Invasive Alien Species Strategy and Action Plan priority actions.

Arctic Invasive Alien Species Strategy and Action Plan	
1. Inspire urgent and effective action: Raise awareness of the unique opportunity that the Arctic Council and its partners have to inspire the urgent and effective action necessary to protect the Arctic from invasive alien species.	
1.1	Promote and, as needed, develop targeted communications and outreach initiatives to raise awareness of the urgent need and unique opportunity to protect the Arctic region from the adverse impacts of invasive alien species;
1.2	Encourage Arctic States and non-Arctic States (including Arctic Council Observer States), working collaboratively with Permanent Participants, to implement effective programmes for preventing the introduction and controlling the spread of invasive alien species through domestic actions and/or international agreements and relevant guidelines, such as the International Convention for the Control and Management of Ships' Ballast Water and Sediments, and the IMO *Guidelines for the control and management of ships' biofouling to minimise the transfer of invasive aquatic species* (Biofouling Guidelines);
1.3	Promote and coordinate the Arctic Council's work on invasive alien species with relevant scientific, technical, and policy-making bodies and instruments; and
1.4	Encourage the integration of the outputs of the Arctic Council's work on invasive alien species into international efforts and legal and institutional frameworks, especially planning and coordination mechanisms, including at the national and sub-national levels, where appropriate.
2. Improve the knowledge base for well-informed decision making: Improve the capacity of the Arctic Council and its partners to make well-informed decisions on the needs, priorities, and options for preventing, eradicating, and controlling invasive alien species in the Arctic by improving the knowledge base.	
2.1	Identify and assess: a) the invasive alien species and pathways that pose the greatest risk of biological invasion into, within, and out of Arctic ecosystems; b) the Arctic ecosystems, livelihoods, and cultural resources most vulnerable to biological invasion; and c) the current and projected patterns and trends of introduction and impacts of invasive alien species in the Arctic;
2.2	Produce a series of topic-specific assessments of invasive alien species issues in the Arctic considering scientific, Traditional Local Knowledge (TLK), technical, environmental, economic, socio-cultural, legal, and institutional perspectives;
2.3	Improve the collection of information on the occurrence and impacts of Arctic invasive alien species, taking advantage of new technologies for early detection, and integrate this information into circumpolar, regional, and community-based observing networks, monitoring programmes, (in particular the Circumpolar Biodiversity Monitoring Programme), and associated information systems such as (the Arctic Biodiversity Data Service); and
2.4	Facilitate full, timely, and open sharing of data and other information relevant to Arctic invasive alien species prevention and management through the Arctic Biodiversity Data Service and the CAFF Web portal.
3. Undertake prevention and early detection/rapid response (EDRR) initiatives: Protect Arctic ecosystems and human well-being by instituting prevention and early detection/rapid response programmes for invasive alien species as a matter of priority.	
3.1	Collaborate with industries, such as, tourism, energy, fisheries, mining, and shipping, and other stakeholders, as relevant, to develop and implement a wide range of biosecurity measures for points of entry and along priority pathways to reduce the initial transfer of species;
3.2	Encourage the establishment of new, or strengthen existing, surveillance, monitoring, reporting, and rapid response programmes necessary to ensure EDRR at points of entry. Consideration of TLK and community-based monitoring programmes should be encouraged;
3.3	Encourage the development and sharing of tools to enable EDRR for invasive alien species that may pose a substantial threat to the Arctic;
3.4	Actively facilitate the eradication of invasive alien species from island ecosystems throughout the Arctic as well as the recovery of native island species and habitats that have been impacted by those invasive alien species;
3.5	Develop guidance for the use and transfer of native and alien species to and throughout the Arctic environment, and identify opportunities to foster ecological resistance and resilience to environmental change;
3.6	Collect information on best practices and assess whether there is a need for the International Maritime Organization to develop Arctic specific guidance for minimising the threat posed by ballast water and biofouling as vectors for the transfer of aquatic invasive alien species from shipping; and
3.7	Foster development of the innovative research, tools, and technologies needed to advance invasive alien species prevention and EDRR capacities in the Arctic region, including through support from funding programmes.

it establishes near-term priorities for securing the future of the Arctic. These priority actions (Table 2) span terrestrial, freshwater, and marine ecosystems and take environmental, cultural and economic factors into consideration. Some of the priority actions apply to the Arctic Council as a whole, while others are best addressed at the working group level or through national implementation. The Conservation of Arctic Flora and Fauna (CAFF) and Protection of the Arctic Marine Environment (PAME) working groups of the Arctic Council hope that each Arctic State, working collaboratively with its partners, will integrate the actions from the *ARIAS Strategy and Action Plan* into national plans and employ the priority actions. This would enable the advancement of relevant decisions made under the auspices of other multi-lateral fora and instruments (e.g. the Convention on Biological Diversity and the International Maritime Organization).

The effective implementation of these priority actions will, of course, depend upon securing the resources necessary to implement them as a matter of urgency and upon collaboration with Permanent Participants, non-Arctic States (including Arctic Council Observers), regional and local authorities, industry and all others who live, work, and travel in the Arctic. Recognition by States, authorities and external organisations that collaborating with the Arctic Council provides a collective and highly desirable benefit will also be crucial. CAFF and PAME will coordinate implementation under the overall direction of the Senior Arctic Officials, drawing on other Arctic Council working groups and partners as needed. Progress reports will be submitted by CAFF and PAME to the Senior Arctic Officials and Arctic Council Ministers every two years.

Although only one of the priority actions set forth in the *ARIAS Strategy and Action Plan* (CAFF, 2017) is explicitly focused on islands, all of the action items are relevant to protecting island ecosystems. Invasive alien species issues are inherently context-specific; they change through time and across landscapes. These particular measures will need to be tailored to particular pathways, populations of non-native species, localities, type and scale of impact, and the available resources.

IMPLEMENTING PRIORITY ACTION

ARIAS Strategy and Action Plan priority action 3.4 calls for the Arctic Council and its partners to "actively facilitate the eradication of invasive alien species from island ecosystems throughout the Arctic as well as the recovery of native island species and habitats that have been impacted by those invasive alien species". The ARIAS Strategy and Action Plan Steering Committee identified this item as a priority because:

1. Island species and ecosystems are well documented as being particularly vulnerable to the impacts of invasive alien species (per previous discussion in this paper). Of particular concern are seabird species that have evolved in the absence of persistent, successful nest-site predators such as the commensal rodents.
2. The level of biological invasion on Arctic islands is relatively low. Due to a lack of other confounding variables, the likelihood for native species/ecosystem recovery following the eradication of invasive vertebrates is high.
3. There are already several examples of successful invasive vertebrate eradications from Arctic islands (Croll, et al., 2015; Jones, et al., 2016; Brooke, et al., 2017). Lessons learnt from these initiatives can be readily applied to future efforts.

To date, efforts to eradicate invasive alien species in the Arctic have been undertaken domestically by the jurisdictional governing body. Priority action 3.4 sets a new precedent for invasive alien species management and creates new opportunities for collaboration, funding, and planning across the region.

The United States Arctic Invasive Species Working Group (coordinated by the National Invasive Species Council (NISC) Secretariat: <www.invasivespecies.gov>) is exploring opportunities to collaborate with domestic and international partners to develop and begin to enact an implementation plan for priority action 3.4. As a minimum, this will include measures to:

1. Identify relevant data available in the Arctic island context and make the data available through open-access information systems, including the Threatened Island Database (TIB) and Biodiversity Information Serving Our Nation (BISON) information system.
2. Summarise the available data to generate information on current knowledge and identify gaps in key information (data gaps).
3. Develop and execute a strategy for filling data gaps.
4. Create a prioritisation schema for determining which island eradications will take precedence and why.
5. Using the schema, determine priorities for the eradication of invasive vertebrates from Arctic islands based on available information and with input from the Arctic Council members and other relevant stakeholders.
6. Based on these priorities, develop an implementation plan, including a co-financing strategy, and secure the additional resources necessary to address these priorities.
7. Implement the eradication plan for the priority island(s) identified in step 5.
8. As appropriate, develop and implement a recovery plan for native island species and habitats of concern. The recovery plan should include a monitoring programme to enable early detection and rapid response to any future invasions.

Invasive alien species have only recently become an issue of concern in the Arctic. Relatively few baseline data on species presence and impacts are available in either the continental or island context. In implementing priority action 3.4, there is a need to start with the basics: assembling/collecting baseline data and evaluating the current status and trends of invasive alien species according to island, species and pathway specific parameters. These assessments are necessary to enable governments to set priorities: which islands, where, why, and how? The findings generated by these assessments can be coupled with data on changes in human activity patterns and climate to generate projections of potential future conditions and thus strengthen and expand the programmes of work necessary to minimise the risk of impending impacts to Arctic island ecosystems (see Hendrichsen, et al., 2014; McGeoch, et al., 2016, for general discussion on assessment needs).

Unfortunately, data collection, sharing, and standardisation is a substantial challenge to filling information gaps in the Arctic. To the best of our knowledge, no one has previously assembled data on invasive alien species occurrence on Arctic islands, although some relevant data can be accessed as subsets of data contributed to national and regional biodiversity information systems [e.g. Global Biodiversity Information Facility (GBIF)]. Where information is unavailable via publicly accessible databases or published literature, information will need to be actively solicited from other available sources, including experts in the field, institutional and/or scientific networks, and traditional local knowledge.

Islands, in general, offer stronger benefits to eradication projects given their high biodiversity, high vulnerability and generally lower risks of reinvasion (compared to non-island ecosystems) that tend towards lasting eradication success (Helmstedt, et al., 2016). However, eradication projects and similar conservation initiatives are proportionately more expensive on islands than other geographical areas due to their typically restricted access and lack of infrastructure, a reality exacerbated in the Arctic (Martins, et al., 2006; Donlan, et al., 2014). Limited resources, cross-jurisdictional collaboration, and evolving techniques/technologies define our capacities to carry out eradication projects. This makes it very important to strike the right balance between the biological need for eradication and the feasibility and sustainability of operations when prioritising locations (Saunders, et al., 2011; Martinez-Abrain & Oro, 2013). Defining clear objectives and measures of performance will be vital in order to effectively and efficiently maximise the limited available funding. Consequent restoration efforts, the second half of priority action 3.4, contribute to the need for an innovative, flexible and integrated portfolio of eradication actions and strategic planning tools. Both restoration capabilities and eradication technical abilities have made exponential progress over the last decades, and yet accurate inclusion of economic costs when prioritising project scope remains a challenge due to its complexities and data gaps that require assumptions and estimates (Donlan & Wilcox, 2007; Carrion, et al., 2011; Veitch, et al., 2011).

To date, no comprehensive invasive alien species eradication prioritisation scheme has been developed for Arctic islands. Recent studies on the prioritisation of islands for invasive alien species eradication projects have highlighted and critiqued approaches to the removal of invasive alien species on a given island from multi-taxa and single-species perspectives. Helmstedt, et al. (2016) highlight the importance of including cost analyses and consideration of high-risk options or targeted, logistical options when weighing the risks and benefits of eradication (Game, et al., 2013; Joseph, et al., 2009). Helmstedt, et al. (2016) point to the value of learning from successes and failures, as well as targeting combinations of invasive alien species, and emphasise three main factors when determining the conservation benefit of various portfolios of action: ecological benefit, economic cost and feasibility of each eradication action. In addition, the study outlines the importance of cost calculations across combined portfolios of action in order to determine cost-sharing opportunities.

In the context of the Arctic islands project outlined above, detailed assessments of invasive alien species eradication options, cost-sharing opportunities and logistical feasibility will need to be conducted once the choice of candidate islands has been narrowed down with the view of maximising potential ecological and social benefits. Table 3 provides an overview of relevant prioritisation criteria to be considered during project planning and implementation. These criteria are not listed according to priority. The level of importance will be assigned during the schema development process.

Translating priorities into action on the ground can be challenging, but it is a reasonable goal when local communities, national and local government agencies, and landowners value the benefits that can be realised from the eradication of invasive alien species from islands. A key strategy to successful implementation will be the development of a "top down/bottom up" approach, where policy, regulatory, and financial support is in place, and the local island communities, landowners and agencies begin investing in the work on the ground. Implementation can be realised when the "demand" finds the resources, support and policies to move forward.

Restoration of island ecosystems is only achievable if adequate and robust funding mechanisms are in place. Projects and programmes tend to be expensive with a large upfront investment required, but the financial return on investment can be high (see Walsh, et al., this 2019). With greater demands and competition for government resources, projects tend to be funded one island at a time. Managers typically rely on blending funding from multiple grant programmes and through partnerships with non-governmental organisations, private foundations and/or philanthropy. This partnership approach to funding projects can be inefficient, and the opportunity to investigate partnerships to co-finance and implement programmatic portfolios is being considered (see Stringer, et al., 2019). Adequate financing is critical to ensure long-term sustainability and protection of the investment to respond to new introductions and facilitate active and passive restoration.

CONCLUSION

Invasive alien species impacts in the Arctic region have global implications. Arctic biodiversity is an irreplaceable asset. To envision the Arctic as ecologically, culturally and economically sustainable necessitates a focus on the factors that threaten the region's environment and human well-being. Thus, eradicating invasive alien species from Arctic island ecosystems will have cumulative benefits. If these islands are protected from invasive alien species, they may have a greater ability to resist and be resilient to other potential stressors. The achievements made through the adoption of the *ARIAS Strategy and Action Plan* present a unique opportunity for collaboration, innovation and collective action across the Arctic at all levels of governance, from regional to local community scales. Governments and their partners need to work together to make the eradication of invasive alien species from Arctic islands feasible, reduce the risks of future island invasions through commerce and other pathways by cooperating in prevention and management efforts across all shared ecosystems, and address the various factors that make island ecosystems particularly vulnerable to the adverse impacts of invasive alien species.

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Post Script Since completion of this paper, the National Invasive Species Council Secretariat and Island Conservation collaborated in the production of an analysis of available non-native species data and developed a preliminary prioritisation schema for Arctic islands. That report, *Data Matters: informing the eradication of invasive species on islands*, is available on the Council's website <https://www.doi.gov/sites/doi.gov/files/uploads/data_matters_island_conservation_report.pdf> and through Island Conservation (Gregg.howald@islandconservation.org).

Table 3 Preliminary factors for consideration in any prioritisation scheme for the Arctic.

Factor	Considerations
IUCN Red Listed species	This includes migratory bird species and should consider the current and trend status of the IUCN Red Listed species, the threat level by the target invasive, and the IUCN Red Listed species' historical recovery status.
Direct and indirect benefits	This is particularly important in understanding how to maximise project-wide benefits that may span varying islands and island systems, species, or stages in the invasion process. Direct benefits include eliminating the threat or degradation posed by the invasive alien species to targeted native ecosystems or species. Indirect benefits may include eliminating the threat or degradation posed to non-targeted ecosystems and species such as those not listed on the IUCN Red List or in other policy.
Direct and indirect consequences	Eradication projects can have significant negative and unintended impacts to native species from the techniques or technologies used, failure of control measures, or greater disruptions to ecosystem equilibriums from the removal of an established invasive alien species. It is important to assess the possibility and probability of potential consequences specific to the prioritisation scheme's target goals. Where other factors outweigh foreseen consequences, mitigation or prevention activities will need to be considered in overall cost and feasibility planning.
Reinvasion potential	The risks of anthropogenic reinvasion vary between islands depending on which pathways they connect to, their geographical proximity to other land masses such as those within swimming distance, the extent of environmental degradation or negative impacts post eradication that affect the feasibility of reestablishment, among others. This component has significant impacts on the sustainability and projected costs of a project.
Biological and ecological vulnerability and resiliency	Biological and ecological vulnerabilities serve as high conservation value components and contribute to project feasibility. Vulnerabilities include islands that come in contact with pathways and the islands' ecological resiliency capacities to biological invasion and reinvasion which impact additional prevention and restoration initiatives.
Impacts on Arctic inhabitants	This consists of not only the direct and indirect economic impacts that disrupt or limit subsistence living and local economies, but also the cultural/spiritual aspects of Arctic life that depend on natural resource identity and use. These considerations in a prioritisation scheme should make use of Traditional Local Knowledge.
Opportunities for community management	Utilising community management opportunities has the potential to not only cut costs and fill knowledge gaps, but also engage local managers and community members in complementary conservation practices such as early detection and rapid response efforts and restoration projects.
Costs and impacts on economies	This consideration needs to extend beyond the direct monetary losses to include the indirect impacts on economies and labour resources (e.g. reduced yields from natural resources, prevention of future yields, alterations and reductions in ecosystem services, and market/non-market value losses (Colautti, et al., 2006).
Feasibility and technology	Feasibility needs to include both the probability of successful eradication and the sustainability of that success. Technology feasibility/availability will differ between islands, species, and ecosystems and need to be assessed and prioritised per project proposal.
Political will of jurisdiction	Sustained political will plays a significant role in the success of any government funded project. When considering a potential site location, island system, or species, it will be important to assess the political will at each level surrounding the project's target and objectives.
Gaps in knowledge	The Arctic has relatively fewer studies regarding native species, invasive alien species, island vulnerabilities, and future risks of biological invasion that go beyond generalisations on warming climates and increasing pathways. It is important that these data gaps are recognised throughout the prioritisation process and adjusted for, where possible.
Climate change impacts	Climate change impacts the vulnerability and susceptibility for biological invasion, reinvasion, and establishment and should be taken into consideration for the long-term feasibility of an eradication project. Together, these two issues can result in exacerbated impacts to ecosystem function and biodiversity (Mooney & Cleland, 2001; Hellman, et al., 2008; Rahel & Olden, 2008).

REFERENCES

Arctic Council (2013). *Vision for the Arctic*. Kiruna, Sweden: Arctic Council.

Bellard, C., Cassey, P. and Blackburn, T.M. (2016). 'Alien species as a driver of recent extinctions'. *Biology Letters* 12: 20150623.

Bennett, J.R., Shaw, J.D., Terauds, A., Smol, J.P., Aerts, R., Bergström, D.M., Blais, J.M., Cheung, WL., Chown, S.L., Lea, M-A., Nielsen, U.N., Pauly, D., Reimer, K.J., Riddles., M.J., Snape, I., Stark., J.S., Tulloch, V.J. and Possingham, H.P. (2015). 'Polar lessons learned: long-term management based on shared threats in Arctic and Antarctic environments'. *Frontiers in Ecology and the Environment* 13: 316–324.

Birnbaum, C. (2013). 'NOBANIS – Invasive Alien Species Fact Sheet: *Neovison vison'*. *European Network on Invasive Alien Species: NOBANIS*. <http://www.nobanis.org>. Accessed 5 November 2016.

Brooke, M.d.L., Bonnaud, E., Dilley, B.J., Flint, E.N., Holmes, N.D., Jones, H.P., Provost, P., Rocamora, G., Ryan, P.G., Surman, C. and Buxton, R.T. (2017). 'Seabird population changes following mammal eradications on islands'. *Animal Conservation*. DOI: 10.1111/acv.12344.

Carrion, V., Donlan, C.J., Campbell, K.J., Lavoie, C. and Cruz, F. (2011). 'Archipelago-wide island restoration in the Galapagos Islands: reducing costs of invasive mammal eradication programs and reinvasion risk'. *PLOS One* 6: e18835.

Chapin, III, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., Mack, M.C. and Díaz, S. (2000). 'Consequences of changing biodiversity'. *Nature* 405: 234–242.

Christiansen, J.S. and Reist, J. (2013). 'Fishes'. In: H. Meltofte (ed.) *Arctic Biodiversity Assessment: Status and Trends in Arctic Biodiversity*, pp.192–245. Akureyri, Iceland: Conservation of Arctic Flora and Fauna.

Colautti, R.I., Bailey, S.A., van Overdijk, C.D.A., Amundsen, K. and MacIsaac, H.J. (2006). 'Characterised and projected costs of nonindigenous species in Canada'. *Biological Invasions* 8: 45–59.

Conservation of Arctic Flora and Fauna (CAFF) (2015). *Actions for Arctic Biodiversity, 2013–2021: Implementing the recommendations of the Arctic Biodiversity Assessment*. Akureyri, Iceland: Conservation of Arctic Flora and Fauna.

Conservation of Arctic Flora and Fauna (CAFF) (2017). *Arctic Invasive Alien Species Strategy and Action Plan*. Akureyri, Iceland: Conservation of Arctic Flora and Fauna.

Croll, D.A., Newton, K.M., McKown, M., Holmes, N., Williams, J.C., Young, H.S., Buckelew, S., Wolf, C.A., Howald, G., Bock, M.F., Curl, J.A. and Tershy, B.R. (2015). 'Passive recovery of an island bird community after rodent eradication'. *Biological Invasions* 18: 703–715.

Dahl, F. and Åhlén, P-A. (2016). 'Abstract: Egg predation by raccoon dog *Nyctereutes procyonoides* in the archipelago of northern Sweden'. *Natural Resources and Bioeconomy Studies* 21: 25.

Doherty, T.S., Dickman, C.R., Nimmo, D.G. and Ritchie, E.G. (2015). 'Multiple threats, or multiplying the threats? Interactions between invasive predators and other ecological disturbances'. *Biological Conservation* 190: 60–68.

Donlan, C.J. and Wilcox, C. (2007). 'Complexities of costing eradications'. *Animal Conservation* 10: 154–156.

Donlan, C.J., Luque, G. and Wilcox, C. (2014). 'Maximizing return on investment for island restoration and species conservation'. *Conservation Letters* 8: 171–179.

Eguíluz, V.M., Fernández-Gracia, J., Irigoien, X. and Duarte, C.M. (2016). 'A quantitative assessment of Arctic shipping in 2010–2014'. *Scientific Reports* 6: 30682.

Ehrenfeld, J.G. (2010). 'Ecosystem consequences of biological invasions'. *Annual Review of Ecology, Evolution and Systematics* 41: 59–80.

Emerson, C. and Lahn, G. (2012). *Arctic Opening: Opportunity and Risk in the High North*. London, UK: Lloyd's and Chatham House.

Fernandez, L., Kaiser, B. and Vestergaard, N. (eds.) (2014). *Marine Invasive Species in the Arctic*. *TemaNord* 2014: 547. Copenhagen, Denmark: Nordic Council of Ministers.

Game, E.T., Kareiva, P. and Possingham, H.P. (2013). 'Six common mistakes in conservation priority setting'. *Conservation Biology* 27: 480–485.

Hall, C.M., James, M. and Wilson, S. (2010). 'Biodiversity, biosecurity and cruising in the Arctic and sub-Arctic'. *Journal of Heritage Tourism* 5: 351–364.

Hellman, J.J., Byers, J.E., Bierwagen, B.G. and Dukes, J.S. (2008). 'Five potential consequences of climate change for invasive species'. *Conservation Biology* 22: 534–543.

Helmstedt, K., Shaw, J., Bode, M., Terauds, A., Springer, K., Robinson, S. and Possingham, H. (2016). 'Prioritizing eradication actions on islands: it's not all or nothing'. *Journal of Applied Ecology* 53: 733–741.

Hendrichsen, D.K., Åström, J., Forsgren, E. and Skarpaas, O. (2014). *Dispersal pathways for alien species in Norway*. NINA Report 1091. Trondheim, Norway: Norwegian Institute for Nature Research. [In Norwegian with English summary]

Howard, G.W. (1999). 'Invasive species and wetlands. Outline of a keynote presentation to the 7th Conference of the Contracting Parties to the Convention on Wetlands', Paper delivered at the 7th Meeting of the Conference of the Contracting Parties to the Convention on Wetlands, San José, 10–18 May 1999.

Jones, H.P., Holmes, N.D., Butchart, S.H.M., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Munoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P.E., Cuthbert, R.J., Ebbert, S., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R. and Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences* 113: 4033–4038.

Joseph, L.N., Maloney, R.F. and Possingham, H.P. (2009). 'Optimal allocation of resources among threatened species: a project prioritization protocol'. *Conservation Biology* 23: 328–338.

Kier, G., Kreft, H., Lee, T.M., Jetz, W., Ibisch, P.L., Nowicki, C., Mutke, J. and Barthlott, W. (2009). 'A global assessment of endemism and species richness across island and mainland regions.' *Proceedings of the National Academy of Sciences* 106: 9322–9327.

Kowalczyk, R. (2014). 'NOBANIS – Invasive Alien Species Fact Sheet – *Nyctereutes procyonoides*'. *European Network on Invasive Alien Species: NOBANIS*. <www.nobanis.org>. Accessed 7 May 2015.

Kurle, C.M., Croll, D.A. and Tershy, B.R. (2008). 'Introduced rats indirectly change marine rocky intertidal communities from algae- to invertebrate-dominated'. *Proceedings of the National Academy of Sciences* 105: 3800–3804.

Mack, R.N., Simberloff, D., Londsdale, W.M., Evans, H., Clout, M. and Bazzaz, F.A. (2000). 'Biotic invasions: causes, epidemiology, global consequences and control'. *Ecological Applications* 10: 689–710.

Magnusson, B. (2010). 'NOBANIS – Invasive alien species fact sheet: *Lupinus nootkatensis*'. *European Network on Invasive Alien Species: NOBANIS*. <https://www.nobanis.org/globalassets/speciesinfo/l/lupinus-nootkatensis/lupinus_nootkatensis.pdf>. Accessed 19 December 2016.

Martinez-Abrain, A. and Oro, D. (2013). 'Preventing the development of dogmatic approaches in conservation biology: A review'. *Biological Conservation* 159: 539–547.

Martins, T.L.F., Brooke, M.D.L., Hilton, G.M., Farnsworth, S., Gould, J. and Pain, D.J. (2006). 'Costing eradications of alien mammals from islands'. *Animal Conservation* 9: 439–444.

McGeoch, M.A., Genovesi, P., Bellingham, P.J., Costello, M.J., McGrannachan, C. and Sheppard, A. (2016). 'Prioritizing species, pathways, and sites to achieve conservation targets for biological invasion'. *Biological Invasion* 18: 299–314.

McNeely, J.A. (ed.). (2001). *The Great Reshuffling: Human Dimensions of Invasive Alien Species*. Gland, Switzerland and Cambridge, UK: IUCN.

McNeely, J.A., Mooney, H.A., Neville, L.E., Schei, P. and Waage, J.K. (eds.). (2001). *A Global Strategy for Invasive Alien Species*. Gland, Switzerland and Cambridge, UK: IUCN.

Meltofte, H. (ed.). (2013). *The Arctic Biodiversity Assessment: Status and Trends in Arctic Biodiversity*. Akureyri, Iceland: Conservation of Arctic Flora and Fauna.

Miller, A.W. and Ruiz, G.M. (2014). 'Arctic shipping and marine invaders'. *Nature Climate Change* 4: 413–416.

Mooney, H.A. and Hobbs, R.J. (eds.). (2000). *Invasive Species in a Changing World*. Washington, DC: Island Press.

Mooney, H.A. and Cleland, E.E. (2001). 'The evolutionary impacts of invasive species'. *Proceeding of the National Academy of Sciences* 98: 5446–5451.

Oug, E., Cochrane, S., Sundet, J.H., Norling, K. and Nilsson, H.C. (2011). 'Effects of invasive red king crab (*Paralithodes camtschaticus*) on soft-bottom fauna in Varangerfjorden, northern Norway'. *Marine Biodiversity* 41: 467–479.

Pejchar, L. and Mooney, H.A. (2009). 'Invasive species, ecosystem services, and human livelihoods'. *Trends in Ecology and Evolution* 24: 497–504.

Protection of the Arctic Marine Environment (PAME) (2011). *Ecosystem-Based Management in the Arctic*. Akureyri, Iceland: Arctic Council.

Rahel, F. and Olden, J. (2008). 'Assessing the effects of climate change on aquatic invasive species'. *Conservation Biology* 22: 521–533.

Reaser, J.K., Meyerson, L.A., Cronk, Q., De Poorter, M., Eldrege, L.G., Green, E., Kairo, M., Latasi, P., Mack, R.N., Mauremootoo, J., O'Dowd, D., Orapa, W., Sastroutomo, S., Saunders, A., Shine, C., Thrainsson, S. and Vaiutu, L. (2007). 'Ecological and socioeconomic impacts of invasive species in island ecosystems'. *Environmental Conservation* 34: 98–111.

Saunders, A., Parkes, J., Aguirre-Munoz, A. and Morrison, S. (2011). 'Increasing the return on investments in island restoration'. In: C.R. Veitch, M.N. Clout and D.R. Towns. (eds.) *Island invasives: eradication and management*, pp. 492–495. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Simberloff, D. (2011). 'How common are invasion-induced ecosystem impacts?' *Biological Invasions* 13: 1255–1268.

Stringer, C., Boudjelas, S., Broome, K., Cranwell, S., Hagen, E., Howald, G., Kelly, J., Millett, J., Springer, K. and Varnham, K. (2019). 'Married bliss and shotgun weddings: effective partnerships for island restoration'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 517–521. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Sundet, J.H. (2014). 'The Red King Crab (*Paralithodes camschaticus*) in the Barents Sea'. In: L. Fernandez, B. Kaiser and N. Vestergaard (eds.) *Marine Invasive Species in the Arctic*, pp. 71–82. Copenhagen, Denmark: Nordic Council of Ministers.

Sutor, A., Kauhala, K. and Ansorge, H. (2010). 'Diet of the raccoon dog *Nyctereutes procyonoides*: A canid with an opportunistic foraging strategy'. *Acta Theriologica* 55: 165–176.

Towns, D.R., Atkinson, I.A.E. and Daugherty, C.H. (2006). 'Have the harmful effects of introduced rats on islands been exaggerated?' *Biological Invasions* 8: 863–891.

Veitch, C.R., Clout, M.N. and Towns, D.R. (eds.) (2011). 'Island invasives: eradication and management'. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Walsh, A., Wilson, A. and McClelland, P. (2019). 'Winning the hearts and minds – proceeding to implementation of the Lord Howe Island rodent eradication project: a case study'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 522–530. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Walther, G-R., Roques, A., Hulme, P.E., Sykes, M.T., Pysek., P. Kühn, I. and Zobel, M. (2009). 'Alien species in a warmer world: risks and opportunities'. *Trends in Ecology and Evolution* 24: 686–693.

White House (2016). *Executive Order 13751: Safeguarding the Nation from the Impacts of Invasive Species*. Washington, USA: The White House.

Wilcove, D., Rothstein, D., Dublow, J., Phillips, A. and Losos, E. (1998). 'Quantifying threats to imperiled species in the United States'. *BioScience* 48: 607–615.

Invasive species removals and scale – contrasting island and mainland experience

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Abstract Recent years have seen large increases in the number and size of successful invasive species eradications from islands. There is also a long history of large scale removals on larger land-masses. These programmes for mammals and terrestrial plants follow the same cost-area relationship although spanning 10 orders of magnitude in scale. Eradication can be readily defined in island situations but can be more complex on larger land-masses where uncertainties defining the extent of a population, multiple population centres on the same land-mass and ongoing risks of immigration are commonplace. The term 'complete removal' is proposed to describe removal from an area with ongoing effort to maintain the area as clear, as features in many larger scale mainland programmes. Examples of complete removal to a boundary, in patches and in habitat islands are discussed. While island eradications continue to grow in scale, new legislation such as the lists of Species of European Union Concern will also drive increasing management on larger land-masses. However, these lists include large numbers of species that are already widespread. Methods are needed to prioritise species to reflect both the risks posed and the feasibility of management, including the effects of scale on cost and effectiveness.

Keywords: control, eradication, invasive alien species, non-native species

INTRODUCTION

The removal or eradication of invasive alien species is increasingly used as a conservation tool. New legislation, for example the European Union's Invasive Alien Species Regulation, will also place increasing responsibilities on states to remove or eradicate high risk species. Both of these considerations are driving an increased number of management programmes at increasing scales and there is a need to understand how the costs and constraints change in relation to scale. A large number of published eradications have been based on islands, often at relatively small scales, while a small number of larger programmes have been based on mainland experience. There is a need to pull together these different sources of evidence, to support an assessment across a wider range of scales than can be achieved by considering islands or mainland eradications in isolation.

REMOVAL AT SCALE – ISLANDS AND MAINLAND EXPERIENCE

Recent years have seen a large increase in successful invasive species eradications from islands, as well as significant increases in the size of islands involved. The number of successful eradications continues to increase, and in 2012 the Database of Invasive Species Eradications (<http://diise.islandconservation.org>) recorded 1,182 whole-island introduced invasive animal species eradication projects either completed or underway on 762 individual islands. In terms of scale, recent years have seen a number of large island eradications. Cruz, et al. (2009) describe the eradication of goats from the 584 km^2 Santiago Island in Galapagos; Parkes, et al. (2014) predicted the effort required to remove cats from the 1,680 km^2 Stewart Island in New Zealand, while the current rat removal on South Georgia will cover 3,538 km^2 (Piertney, et al., 2016).

Although the point at which an island becomes a mainland is arbitrary, there is also a long history of invasive mammal removals from larger land masses in Northern Europe (Robertson, et al., 2017). These include muskrat (*Ondatra zibethicus*) eradications from the mainlands of Britain and Ireland in the 1930s; the eradication of the Himalayan porcupine *(Hystrix brachyura*) (1970s) and coypu (*Myocaster coypus*) (1980s) from the British mainland; a variety of American mink (*Neovison vison*) and grey squirrel (*Sciurus carolinensis*) removals from the larger British islands together with the removal of Pallas' squirrel (*Callosciurus erythraeus*) from Flanders on the European mainland (since 2000). Few of the programmes covered more than a fraction of the total land mass, so size was defined as the area over which species sightings occurred and trapping took place. The larger of these species programmes have covered areas of 3,411 km^2 (the two phases of the Hebridean mink programme), 5,219 km^2 (the five separate muskrat eradications) and 19,210 km^2 (coypu) (details and full references given in Robertson, et al. 2017). The ongoing ruddy duck (*Oxyura jamaicensis*) eradication from Europe (Robertson, et al., 2015) covers six states totalling 1,535,509 km^2.

Data on the costs of eradications are available for projects covering ten orders of magnitude of scale. Studies have described the costs of successful mammal eradications from islands (Martins, et al., 2006; Howald, et al., 2007) and larger land-masses (Robertson, et al., 2017), while Rejmánek & Pitcairn (2002) describe costed plant eradications in California. For mammal eradications, those on large land-masses covered significantly larger areas than those reported from islands while successful plant eradications were confined to smaller areas. Data from these different sources, appear to follow the same relationship (Fig. 1) whereby the cost per unit area is reduced by approximately 10% as the area involved doubles (Robertson, et al., 2017). As experience of eradications on larger islands grows, the overlap between island and mainland experiences is increasing (Cruz, et al., 2009; Parkes, et al., 2014; Piertney, et al., 2016).

It is worth recording that two small datasets describe programmes that fall outside this relationship. Rejmánek & Pitcairn (2002) also record three aquatic plant eradications which appeared more expensive than comparably sized terrestrial plant programmes, while the ruddy duck

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eradication (Robertson, et al., 2015) has been significantly less expensive compared to similarly scaled mammal programmes (Robertson, et al., 2017). More data are needed on the management of other taxa in different environments before firm conclusions can be drawn. These results are based upon currently available methods of eradication. As new technologies, such as gene-drives (Webber, et al., 2015), e-DNA self-resetting (Carter, et al., 2016) and self-reporting traps (Jones, et al., 2015) become available it is likely that these costs will decrease.

Eradication and complete removal

In their classic paper, Bomford & O'Brien (1995) make a clear distinction between eradication and on-going control, presenting these as alternative objectives for management. They also identify three key criteria for successful eradication; that the rate of removal exceeds the rate of increase at all densities; there is no immigration; and all reproductive animals are at risk.

These definitions and criteria have guided many successful eradications and are particularly applicable to islands where the population extent and risks of immigration can be readily assessed. However, at the scales found on larger land masses, these criteria may be more difficult to apply or achieve, for example where the boundaries of a population remain poorly defined, where multiple population centres may occur on the same land mass, or where immigration remains a risk. Despite this, large scale programmes frequently lead to the removal of species from large areas of land. Although not meeting Bomford & O'Brien's (1995) definition of eradication, these situations are also not well described as on-going control as no active management is required across the majority of the area. In these circumstances 'complete removal' may be a better definition of the objectives, sitting between Bomford & O'Brien's (1995) definitions of eradication and on-going control.

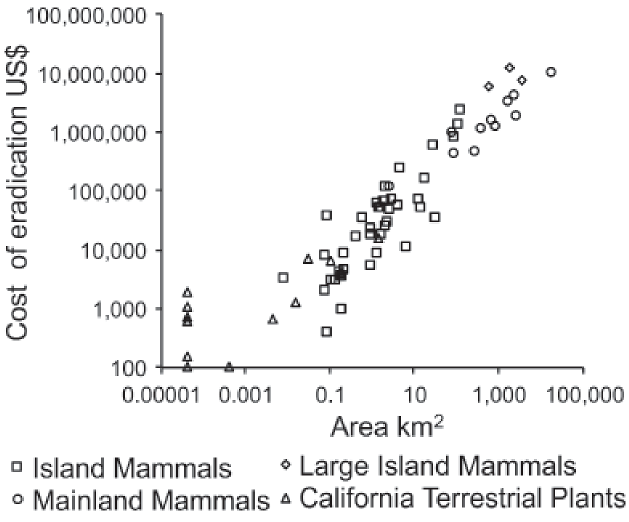

Fig. 1 The relationship between the area (km2) of a successful removal and the total cost (US$). The square symbols represent island mammal eradications reported by Martins, et al., (2006). The circles are for removals of mammals from larger land masses in Northern Europe (Robertson, et al., 2016). The three diamond symbols are recent examples or predictions of large-scale mammal eradications from islands: (Cruz, et al., 2009; Parkes, et al., 2014; Piertney, 2016). Plant eradications from California are triangles (Rejmánek & Pitcairn, 2002). Where the study recorded effort as man-years or man-days, total cost is estimated based on US$50k per man-year (Rejmánek & Pitcairn, 2002; Parkes, et al., 2014; Robertson, et al., 2016).

Eradication, the complete removal from an area, with no immediate prospect of recolonisation from neighbouring areas.

Complete Removal from an area but with ongoing effort to maintain the area as clear.

On-going Control within an area to reduce abundance, associated damage and the risk of spread.

Based on this definition, complete removal has been applied in a number of forms.

1 - Complete removal to a boundary

One objective of large scale programmes can include complete removal of a species up to a boundary across which the risk of reinvasion remains. Control along the boundary, or in a neighbouring buffer zone, can reduce the risk of reinvasion and help keep the main area clear. The nature of the boundary may vary, including fences (Saunders & Norton, 2001), landscape barriers such as water bodies or mountains (Schuchert, et al., 2014), or bottlenecks through which invading animals must move (Roy, et al., 2015). These boundaries can be permanent features of the management, requiring ongoing inputs (Saunders & Norton, 2001), or may be part of a phased programme to clear a larger area (Yamada & Sugimura, 2004; Bryce, et al., 2011; Robertson, et al., 2015; Russell, et al., 2015). If the aim is the removal of the species from a large area, but the funds or resources are insufficient for the simultaneous management of the entire population, then removal to a boundary is likely to feature.

The North American ruddy duck was introduced to the UK in the late 1940s, and its subsequent spread into Europe threatens the native white-headed duck (*Oxyura leucocephalus*) through hybridisation. The plan to eradicate the ruddy duck from Europe involves coordinated management across the continent. As the UK was the original source of this population and contained the majority of the birds, it was the focus of initial control (Robertson, et al., 2015). However, once the UK no longer contained breeding birds (currently it is thought only a few males remain), the English Channel became a boundary between a cleared area and the remaining continental populations. Control of the remaining European birds is ongoing, in the meantime the UK maintains surveillance and, if required, control along this boundary to maintain its cleared status.

In the UK, the native red squirrel (*Sciurus vulgaris*) is threatened by the ongoing spread of the invasive grey squirrel (*S. carolinensis*). This is mediated by the spread of a poxvirus by the asymptomatic greys which is typically fatal to the reds (Rushton, et al., 2000). The island of Anglesey on the north coast of Wales contained a small relict population of the native reds although greys were spreading onto the island. A control programme removed the greys (Schuchert, et al., 2014), allowing the reds to spread and recolonise the entire island. Anglesey is separated from mainland Wales by a narrow tidal channel, crossed by two bridges. There is evidence that grey squirrels can cross this boundary and the risk of recolonisation remains. To reduce this risk and maintain the island as grey squirrel-free, management has included a surveillance and rapid response programme to pick up incursions (Shuttleworth, et al., 2016), trapping to reduce the density of greys on the mainland side of the boundary, and a plan to extend the area of complete removal to clear greys from the North Wales coast up to a more distant boundary formed by a geographic bottleneck where the mountains meet the coast.

The American mink (*Neovison vison*) spread through the Western Isles of Scotland following its escape from fur farms in the 1950s. Its spread threatened internationally important populations of ground nesting birds as well as local economic activities such as salmon fishing. The decision was taken to aim for the eradication of this species from the archipelago but logistic and funding constraints, combined with the need to gain experience, led to a phased programme. In the first phase, mink were completely removed from the Uists, the southernmost islands of the chain (Roy, et al., 2015; Faulkner, et al., 2017). A buffer zone was maintained (South Harris) between this cleared area and the remaining mink population on the main island (Lewis) to the north. This buffer included a narrow, island strewn channel between the Uists and South Harris. Trapping on these 'stepping stone' islands together with South Harris itself provided an effective barrier to recolonisation. Once the Uists' work had provided confidence that eradication was feasible, a second phase extended mink control north to cover the remainder of the archipelago (Lambin, et al., 2014).

2 - Complete removal from patches

In some cases the primary objective of management may be the reduction of the impact of an invasive species with no prospect to eradicate. In many cases this constitutes ongoing control rather than complete removal (Bomford & O'Brien, 1995), although in some circumstances it can lead to complete removal. For this to occur, two criteria must be met, the species must be controlled at a rate sufficient to remove all of the resident animals in an area, and the scale of control should be such that the risk of recolonisation is so low in the centre of the controlled area that the central area is effectively maintained clear. The prospects of this occurring are scale-dependent, with the cleared area forming a larger proportion of the total as scale increases.

This approach has been used in New Zealand with the creation of 'mainland islands', areas maintained predator-free through the use of fencing combined with continuing control (Saunders & Norton 2001; Gillies, et al., 2003). The same results can be achieved without fencing, for example in Mauritius where the introduced small Asian mongoose (*Urva auropunctata*) (Patou, et al., 2009) is a major threat to the continued survival of a range of native bird species (Bunbury, et al., 2008). The mongoose is widely spread across the island, inhabiting a range of habitats, while the native birds are largely confined to remaining patches of good quality native forest. Control of the mongoose has been carried out in a number of these forest areas to create 'mongoose free' patches within the wider mongoose distribution. A network of box traps has been in place since 1989 and maintains a year-round effort to remove mongoose. As the size of the trapped area increases, the number of animals captured per unit area decreases (Fig. 2). Areas less than 5 km in extent continue to catch high numbers of mongoose per unit area, presumably because they face constant recolonisation pressure from neighbouring habitats. However, in larger areas, particularly those over 10 km^2 in area, mongoose catch per unit area drops dramatically. This is consistent with catching animals in a boundary area, with the proportion of the area maintained as mongoose-free increasing as the total area trapped increases. Achieving this requires ongoing effort, but complete removal provides many of the benefits of eradication, and has been a key element of efforts to conserve a suite of species endemic to the island. These include the Mauritius kestrel (*Falco punctatus*), the pink pigeon (*Nesoenas mayeri*), the echo parakeet (*Psittacula eques*) and a number of passerines such as the Mauritius black bulbul (*Hypsipetes olivaceus)*, and Mauritius fody (*Foudia rubra)*. Only through intensive trapping to maintain these predator-free patches, combined with a captive breeding and release programme, disease management and supplementary feeding, have these species managed to persist.

3 – Complete removal from habitat islands

Islands as blocks of land surrounded by water are widely recognised, but isolated blocks of habitat within a matrix of other land uses share many of the same characteristics. When invasive alien species are confined to discrete habitats within this matrix, they can be considered as inhabiting 'habitat islands'. In these cases, limited rates of species movement or colonisation between habitat islands may produce isolated populations, with particular opportunities for management within large land masses.

The monk parakeet (*Myiopsitta monachus*) has established a number of discrete populations in different European cities (Munoz & Real, 2006; Rodríguez-Pastor, et al., 2012). Although an attractive species widely kept as a pet, in the wild this species builds large communal nests on tall trees or man-made structures such as electricity infrastructure or radio masts. The large size and volume of nest material can lead to electrical short-outs and fire risks, with consequent economic costs (Avery, et al., 2002). The discrete nature of its current distribution, with isolated populations including London, Amsterdam and a variety of Spanish cities suggests that different populations have resulted from separate releases rather than natural spread from a single point of release. The management of this species reflects this, with some regions attempting the complete removal of isolated populations (Parrott, 2013).

The introduced Pallas' squirrel (*Callosciurus erythraeus*) also has a highly fragmented distribution within Europe, suggesting a number of separate introductions rather than spread from a single point of release. A rapid response in Flanders, Belgium removed a population whose distribution was constrained to a suburban setting in a small community surrounded by farmland (Adriaens, et al., 2015). In effect this species was present on a habitat island which aided its removal.

The current removal of rats from South Georgia (Piertney, et al., 2016) uses a similar approach. Glaciers on the island separate a number of discrete rat populations, which appear to be genetically isolated (Robertson & Gemmell, 2004). This allows the complete removal of discrete populations as steps to achieve the larger goal of island wide eradication.

These examples illustrate the potential for effective removal of isolated populations to be undertaken within larger land masses, using the principles applied to island

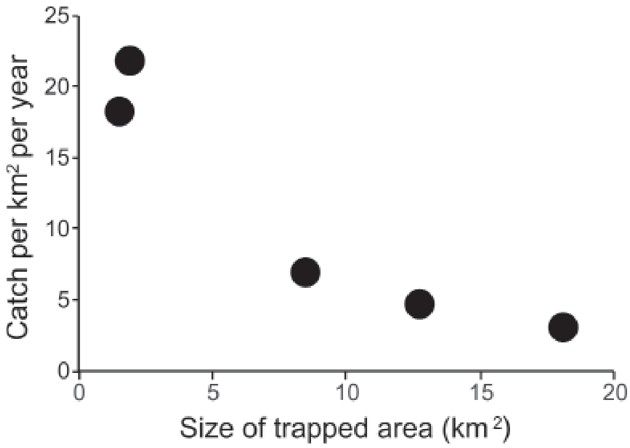

Fig. 2 The density of mongoose removed by trapping in five conservation areas in Mauritius. The control areas were surrounded by habitat containing mongoose populations.

eradications. However, as species establish and spread these discrete populations will become less pronounced. Identifying whether the distribution of a species represents a number of discrete clusters will have important implications for management, for example the decision to consider complete removal or on-going control. Spatial analysis of distributional data can be used to indicate the presence of discrete populations of a species. A range of spatial and spatio-temporal clustering algorithms (Velázquez, et al., 2016) can detect spatial point patterns and may be useful to differentiate clusters as they form.

EFFECTIVENESS AND SCALE

We used published accounts to assess the costs of removal at different scales. Doing so requires dealing with a number of biases. Firstly, it is commonly recognised that the published literature preferentially records success (Dwan, et al., 2008). For example, the successful coypu eradication in the UK is well documented (Gosling & Baker, 1987; Baker & Clarke, 1988; Gosling, et al., 1988; Gosling & Baker, 1989; Baker, 2006); the failed UK attempt to eradicate the American mink is barely recorded (Sheail, 2004) although it took place on a similar scale. Other failures are likely to have gone unrecorded. A publicly available database of island eradications is available (Keitt, et al., 2011), it would be useful to extend this to also include details of eradications on larger land masses. More importantly, the literature only records attempts, there is very little information on those situations where no action was taken, either through inaction or a judgement that it was not worthwhile. Inaction remains the most common response to invasive species. The successful island eradications are based on only a tiny proportion of the world islands, while the number of attempted eradications of alien species in Europe (Genovesi, 2005) is a similarly small proportion of the 20,000 species thought to have established.

If we are to make more objective decisions, we need to decide if, and when, management is appropriate in both island and mainland situations. Prioritisation methods have been applied to islands to identify those where management may be most beneficial (Harris, et al., 2012; Dawson, et al., 2015). Booy, et al. 2017 describe a method to assess the feasibility of eradication which incorporates the consideration of scale. If, as seems likely but has yet to be convincingly demonstrated, the prospects for successful eradication or complete removal decrease as a species spreads, then these methods offer a route to assess at what scale eradication or complete removal may no longer be a realistic outcome.

The application of methods to assess the feasibility of management is a critical need. The current EU invasive alien species regulations include the listing of species considered to be of 'Union Concern' and place reporting and management obligations on member states in which they occur. The selection of species for listing is largely based on established methods of risk assessment (Roy, et al., 2014), identifying species which pose a risk without similarly considering the feasibility of management. This focus on risk can result in the listing of species for which there are few realistic prospects for management. For example, of the 79 species currently listed or under consideration as Species of Union Concern, over half are already present in at least five member states. To date there are no successful examples of species eradication or complete removal in Europe when a species has already spread to this number of countries, although these may occur in future. Listing species based on risk assessment alone, without considering the scale and feasibility of management, risks committing resources into the on-going management of already widespread species, rather than the more productive routes of prevention and rapid response.

CONCLUSIONS

The experience of island eradications continues to grow, and to be applied at increasing scales. Alongside this, new legislation will drive increasing management on larger land masses. As island eradications grow in scale they will face many of the challenges experienced on larger land-masses, such as problems defining populations, multiple population centres on the same land mass, ongoing risks of immigration and the need for interim objectives. We suggest the term 'complete removal' to reflect the situation regularly encountered on larger land masses where a species may be removed from an area but with the need for an ongoing effort to maintain the area clear given the risk of reinvasion. The literature contains examples of successful eradications or complete removals in island and mainland situations covering 10 orders of magnitude. These island and mainland programmes appear to follow the same cost-area relationship. They also demonstrate an advantage of scale, with the costs per unit area of control reduced as the area of control increases. On larger land masses, such as the EU, care is needed to focus species listing on species where prevention, eradication or complete removal are realistic outcomes rather than committing member states to the on-going control of already widespread species. Methods of prioritisation which balance both risk and the feasibility of management, including the effects of scale on cost and effectiveness, are needed to guide future actions.

REFERENCES

Adriaens, T., Baert, K., Breyne, P., Casaer, J., Devisscher, S., Onkelinx, T., Pieters, S. and Stuyck, J. (2015). 'Successful eradication of a suburban Pallas's squirrel *Callosciurus erythraeus* (Pallas 1779) (Rodentia, Sciuridae) population in Flanders (northern Belgium)'. *Biological Invasions* 17(9): 2517–2526.

Avery, M.L., Greiner, E.C., Lindsay, J.R., Newman, J.R. and Pruett-Jones, S. (2002). 'Monk parakeet management at electric utility facilities in south Florida'. *Proceedings of the 20th Vertebrate Pest Conference*, pp. 140–145. University of California, Davis.

Baker, S.J. and Clarke, C.N. (1988). 'Cage trapping coypus (*Myocastor coypus*) on baited rafts'. *Journal of Applied Ecology* 25: 41–48.

Baker, S. (2006). 'The eradication of coypus (*Myocastor coypus*) from Britain: the elements required for a successful campaign'. *Assessment and Control of Biological Invasion Risks*, pp.142–147. Kyoto, Japan: Shoukadoh Book Sellers andOccasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Bomford, M. and O'Brien, P. (1995). 'Eradication or control for vertebrate pests?' *Wildlife Society Bulletin* 23: 249–255.

Booy, O., Mill, A.C., Roy, H.E., Hiley, A., Moore, N., Robertson, P., Baker, S., Brazier, M., Bue, M., Bullock, R. and Campbell, S. (2017) 'Risk management to prioritise the eradication of new and emerging invasive non-native species'. *Biological Invasions*, 19(8): 2401–2417.

Bunbury, N., Stidworthy, M.F., Greenwood, A.G., Jones, C.G., Sawmy, S., Cole, R.E., Edmunds, K. and Bell, D.J., (2008). 'Causes of mortality in free-living Mauritian pink pigeons *Columba mayeri*, 2002–2006'. *Endangered Species Research* 9(3): 213–220.

Bryce, R., Oliver, M.K., Davies, L., Gray, H., Urquhart, J. and Lambin, X., (2011). 'Turning back the tide of American mink invasion at an unprecedented scale through community participation and adaptive management'. *Biological Conservation* 144(1): 575–583.

Carter, A., Barr, S., Bond, C., Paske, G., Peters, D. and van Dam, R. (2016). 'Controlling sympatric pest mammal populations in New Zealand with self-resetting, toxicant-free traps: a promising tool for invasive species management'. *Biological Invasions* 18(6): 1723–1736.

Cruz, F., Carrion, V., Campbell, K.J., Lavoie, C. and Donlan, C.J. (2009). 'Bio-economics of large-scale eradication of feral goats from Santiago Island, Galápagos'. *Journal of Wildlife Management* 73(2): 191–200.

Dawson, J., Oppel, S., Cuthbert, R.J., Holmes, N., Bird, J.P., Butchart, S.H., Spatz, D.R. and Tershy, B. (2015). 'Prioritizing islands for the eradication of invasive vertebrates in the United Kingdom overseas territories'. *Conservation Biology* 29(1): 143–153.

Dwan, K., Altman, D.G., Arnaiz, J.A., Bloom, J., Chan, A.W., Cronin, E., Decullier, E., Easterbrook, P.J., Von Elm, E., Gamble, C. and Ghersi, D. (2008). 'Systematic review of the empirical evidence of study publication bias and outcome reporting bias'. *PLOS One*, *3*(8): e3081.

Faulkner, S.C., Verity, R., Roberts, D., Roy, S.S., Robertson, P.A., Stevenson, M.D. and Le Comber, S.C. (2017). 'Using geographic profiling to compare the value of sightings vs trap data in a biological invasion.' *Diversity and Distributions* 23(1): 104–112.

Genovesi, P., (2005). 'Eradications of invasive alien species in Europe: a review'. *Issues in Bioinvasion Science* 7: 127–133.

Gillies, C.A., Leach, M.R., Coad, N.B., Theobald, S.W., Campbell, J., Herbert, T., Graham, P.J. and Pierce, R.J. (2003). 'Six years of intensive pest mammal control at Trounson Kauri Park, a Department of Conservation "mainland island", June 1996–July 2002'. *New Zealand Journal of Zoology* 30(4): 399–420.

Gosling, L.M. and Baker, S.J. (1987). 'Planning and monitoring an attempt to eradicate coypus from Britain'. *Symposia of the Zoological Society of London* 58: 99–113.

Gosling, L.M., Baker, S.J. and Clarke, C.N. (1988). 'An attempt to remove coypus (*Myocastor coypus*) from a wetland habitat in East Anglia'. *Journal of Applied Ecology* 25: 49–62.

Gosling, L.M. and Baker, S.J. (1989). 'The eradication of muskrats and coypus from Britain.' *Biological Journal of the Linnean Society* 38(1): 39–51.

Harris, D.B., Gregory, S.D., Bull, L.S. and Courchamp, F. (2012). 'Island prioritization for invasive rodent eradications with an emphasis on reinvasion risk'. *Biological Invasions* 14(6): 1251–1263.

Howald, G., Donlan, C., Galván, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M. and Saunders, A. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21(5): 1258–1268.

Jones, C., Warburton, B., Carver, J. and Carver, D. (2015). 'Potential applications of wireless sensor networks for wildlife trapping and monitoring programs'. *Wildlife Society Bulletin* 39(2): 341–348.

Keitt, B., Campbell, K., Saunders, A., Clout, M., Wang, Y., Heinz, R., Newton, K. and Tershy, B., 2011. The global islands invasive vertebrate eradication database: a tool to improve and facilitate restoration of island ecosystems. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 74–77. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Lambin, X.L., Cornulier, T., Oliver, M.K. and Fraser, E.J. (2014) *Analysis and Future Application of Hebridean Mink Project Data*. Scottish Natural Heritage Commissioned Report No. 522.

Martins, T.L.F., Brooke, M.D.L., Hilton, G.M., Farnsworth, S., Gould, J. and Pain, D.J. (2006). 'Costing eradications of alien mammals from islands'. *Animal Conservation* 9(4): 439–444.

Muñoz, A.R. and Real, R. (2006). 'Assessing the potential range expansion of the exotic monk parakeet in Spain'. *Diversity and Distributions* 12(6): 656–665.

Parkes, J., Fisher, P., Robinson, S. and Aguirre-Muñoz, A., (2014). 'Eradication of feral cats from large islands: an assessment of the effort required for success'. *New Zealand Journal of Ecology* 38: 307–314.

Parrott, D. (2013) 'Monk parakeet control in London'. In: C. van Ham, P. Genovesi and R. Scalera (eds.) *Invasive alien species: the urban dimension*, pp 83–85. Gland, Switzerland: IUCN.

Patou, M. L., Mclenachan, P. A., Morley, C. G., Couloux, A., Jennings, A. P. and Veron, G. (2009). 'Molecular phylogeny of the Herpestidae (Mammalia, Carnivora) with a special emphasis on the Asian Herpestes'. *Molecular Phylogenetics and Evolution* 53(1): 69–80.

Piertney, S.B., Black, A., Watt, L., Christie, D., Poncet, S. and Collins, M.A. (2016). 'Resolving patterns of population genetic and phylogeographic structure to inform control and eradication initiatives for brown rats *Rattus norvegicus* on South Georgia'. *Journal of Applied Ecology* 53: 332–339.

Rejmánek, M. and Pitcairn, M.J. (2002). 'When is eradication of exotic pest plants a realistic goal.' In: C.R. Veitch and M.N. Clout (eds.) *Turning the tide: the eradication of invasive species,* pp. 249–253. IUCN, SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Robertson, B.C. and Gemmell, N.J. (2004). 'Defining eradication units to control invasive pests'. *Journal of Applied Ecology* 41(6): 1042–1048.

Robertson, P.A., Adriaens, T., Caizergues, A., Cranswick, P.A., Devos, K., Gutiérrez-Expósito, C., Henderson, I., Hughes, B., Mill, A.C. and Smith, G.C. (2015). 'Towards the European eradication of the North American ruddy duck.' *Biological Invasions* 17(1): 9–12.

Robertson, P.A., Adriaens, T., Lambin, X., Mill, A., Roy, S., Shuttleworth, C.M. and Sutton-Croft, M. (2017). 'The large-scale removal of mammalian invasive alien species in Northern Europe'. *Pest Management Science* 73(2): 273–279.

Rodríguez-Pastor, R., Senar, J.C., Ortega, A., Faus, J., Uribe, F. and Montalvo, T. (2012). 'Distribution patterns of invasive Monk parakeets (*Myiopsitta monachus*) in an urban habitat.' *Animal Biodiversity and Conservation* 35(1): 107–117.

Roy, H., Schonrogge, K., Dean, H., Peyton, J., Branquart, E., Vanderhoeven, S., Copp, G., Stebbing, P., Kenis, M., Rabitsch, W. and Essl, F. (2014). *Invasive Alien Species–Framework for the Identification of Invasive Alien Species of EU Concern*. Report to the European Commission ENV.B.2/ETU/2013/0026.

Roy, S.S., Chauvenet, A.L.M. and Robertson, P.A., (2015). 'Removal of American mink (*Neovison vison*) from the Uists, Outer Hebrides, Scotland'. *Biological Invasions* 17(10): 2811–2820.

Rushton, S.P., Lurz, P.W.W., Gurnell, J. and Fuller, R. (2000). 'Modelling the spatial dynamics of parapoxvirus disease in red and grey squirrels: a possible cause of the decline in the red squirrel in the UK?' *Journal of Applied Ecology* 37(6): 997–1012.

Russell, J.C., Innes, J.G., Brown, P.H. and Byrom, A.E. (2015). 'Predator-free New Zealand: conservation country'. *BioScience* 65(5): 520–525.

Saunders, A. and Norton, D.A. (2001). 'Ecological restoration at mainland islands in New Zealand'. *Biological Conservation* 99(1): 109–119.

Schuchert, P., Shuttleworth, C.M., McInnes, C.J., Everest, D.J. and Rushton, S.P. (2014). 'Landscape scale impacts of culling upon a European grey squirrel population: can trapping reduce population size and decrease the threat of squirrelpox virus infection for the native red squirrel?' *Biological Invasions* 16(11): 2381–2391.

Shuttleworth, C, Robertson, P. and Halliwell, L. (2016). 'Identifying incursion pathways, early detection responses and management actions to prevent grey squirrel range expansion: an island case study in Wales.' In: C.M. Shuttleworth, P.W.W. Lurz and J. Gurnell (eds.) *The Grey Squirrel: Ecology and Management of an Invasive Species in Europe*, pp. 475–495. UK: European Squirrel Initiative.

Sheail, J. (2004). 'The mink menace: the politics of vertebrate pest control'. *Rural History* 15(02): 207–222.

Velázquez, E., Martínez, I., Getzin, S., Moloney, K.A. and Wiegand, T. (2016). 'An evaluation of the state of spatial point pattern analysis in ecology'. *Ecography* 39(11): 1042–1055.

Webber, B.L., Raghu, S. and Edwards, O.R. (2015). 'Opinion: Is CRISPR-based gene drive a biocontrol silver bullet or global conservation threat?' *Proceedings of the National Academy of Sciences* 112(34): 10565–10567.

Yamada, F. and Sugimura, K. (2004). 'Negative impact of an invasive small Indian mongoose *Herpestes javanicus* on native wildlife species and evaluation of a control project in Amami-Ohshima and Okinawa Islands, Japan'. *Global Environmental Research* 8(2): 117–124.

Strategic environmental assessment for invasive species management on inhabited islands

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Abstract Over the past decade the challenges of managing invasive species on inhabited islands have clearly become limiting factors to scaling-up the area of invasive species eradications. Step-change is required to unleash the conservation and restoration potential of biodiversity on inhabited islands around the globe and avoid the pitfalls previous attempts to eradicate invasive species on inhabited islands have fallen into. Strategic environmental assessment (SEA) is a systematic decision support process, aiming to ensure that environmental and other sustainability aspects are considered effectively throughout policy, plan and programme making. Within the framework of SEAs, on target islands eradication planners could implement a number of tools including stakeholder engagement, social impact assessment and economic cost-benefit analysis alongside existing environmental impact assessment. Such a suite of assessments captures the immediate impacts of an eradication operation on a range of values, alongside predicted long-term changes in these tightly coupled socio-ecological systems. In this paper we outline what SEA is, and then contrast invasive species management attempts occurring outside an SEA framework on two similar but also contrasting UNESCO World Heritage islands; Lord Howe Island, Australia and Fernando de Noronha, Brazil. We then demonstrate how an SEA approach to invasive species management would assist planning in New Zealand to eradicate introduced mammalian predators from two large offshore islands in New Zealand; Aotea (Great Barrier Island) and Rakiura (Stewart Island). We conclude with future prospects for applying SEA to invasive species management on inhabited islands.

Keywords: eradication, mammals, New Zealand, social impact assessment, World Heritage

INTRODUCTION

Over the past decade the challenges of managing invasive species on inhabited islands have clearly become limiting factors to scaling-up the area of invasive species eradications on islands (Oppel, et al., 2011; Glen, et al., 2013). This is particularly the case for eradication of small mammalian predators, where step-change in technology (e.g. use of helicopters for aerial delivery of toxin; Howald, et al., 2007) coupled with ongoing incremental advances (e.g. non-target mitigation; Hanson, et al., 2015) mean very large islands are now potential targets of whole-island small mammal eradication, but there has not been a commensurate increase in the knowledge around engaging with resident communities (Russell, et al., 2018). Scaling-up eradications to larger islands is also confounded by additional complexities such as mixed land-tenure and land-use (Holmes, et al., 2015) on larger islands, further complicating the suite of appropriate methods for social engagement and technical implementation.

There are many reasons why there should be an increased emphasis on inhabited islands as targets for biodiversity conservation. Most uninhabited islands are small and, although the number of islands from which invasive species have been eradicated is impressive (e.g. Jones, et al., 2016), as a percentage island land area the total is still low (Russell, et al., 2016a). Some endangered species can only be conserved on large islands (PCE, 2017), while climate change increases the long-term risk profile for small islands as resilient conservation sites (Courchamp, et al., 2014). In the tropics, even small islands can be inhabited (Russell & Holmes, 2015), and small island developing states (SIDS) are particularly poorly represented in invasive mammal eradication statistics (Russell, et al., 2017a). Eradication of invasive mammals on inhabited islands also brings about many other benefits beyond biodiversity conservation, including benefits to agriculture, economies, public health and culture (Russell, et al., 2017a).

To date approaches to community engagement in anticipation of mammal eradication on inhabited islands have been designed and led mainly by biologists with a particular set of values and priorities (e.g. Bell, 2019). They have tended to be ad hoc and have not always drawn upon existing scholarship in community engagement. A new step-change is required to unleash the conservation and restoration potential of biodiversity on inhabited islands around the globe and avoid the pitfalls previous attempts to eradicate invasive species on inhabited islands have fallen in to. In this paper we outline the potential for strategic environmental assessment to enable more consistent assessment of options and engagement with island communities in the context of invasive mammal eradication. We then provide two contrasting illustrative examples of approaches to invasive predator management on two similar UNESCO World Heritage island sites, followed by examples from the two largest inhabited offshore islands of New Zealand. We conclude with recommendations for implementing strategic environmental assessment during planning for invasive species management. We emphasise that much of the scholarship we present here is built upon reflection over the past decade on attempts to eradicate invasive mammals from inhabited islands. These lessons come from the benefit of hindsight and could not be anticipated in advance, so they should not be taken as reflecting poorly on those who initially invested themselves in advocating for invasive mammal eradication. Our purpose is to suggest a way towards better processes and improved outcomes from eradications on inhabited islands.

Strategic Environmental Assessment

Strategic environmental assessment (SEA) is a widely accepted approach to applying impact assessment to policies, plans and programmes, contributing to the planning processes, decision making and the ongoing management of change (Tetlow & Hanusch, 2012). Sustainability assessment is another approach often linked to SEA (Morgan 2012). SEA has been described as "analytical and participatory approaches that aim to integrate environmental considerations into policies,

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plans and programmes and evaluate the inter linkages with economic and social considerations" (OECD, 2006). Applications of SEA include spatial planning, sector planning (e.g. fisheries, energy) and catchment planning (Tetlow & Hanusch, 2012; Taylor & Mackay, 2016).

Importantly, SEA provides an over-arching framework of a collection of tools rather than a single, fixed and prescriptive approach. Such an approach is therefore analogous to best practice in technical implementation of eradications on islands (Keitt, et al., 2015), where just as islands differ ecologically, it is also recognised they differ socially. Thus, in any particular case variations with regards to best, or complete, practice will still take place. Its application is an ongoing adaptive and iterative process which adds value to and builds capacity in existing systems (e.g. island human communities). The sorts of tools that can be considered as contributing to the SEA toolbox for island eradications include:

- Community and stakeholder engagement techniques
- Social profiles/baselines and social impact assessments (SIA)
- Health impact assessments
- Cost benefit analyses
- Ecological baselines and impact assessments (EIA)
- Technical feasibility studies
- Livelihoods analyses
- Social marketing/environmental education
- Environmental and social monitoring
- Institutional analyses and change management (includes ongoing biosecurity planning).

As a toolbox, SEA has been around since about the early 1990s when it developed from a growing realisation that local and project-specific applications of environmental impact assessment are insufficient when environmentally damaging decisions are being made at a more strategic level. SEA has not been widely applied in the context of wildlife management (Taylor, et al., 2004). However, in some countries SEA-like frameworks have been implemented in all but name (e.g. the Resource Management Act in New Zealand provides for the application of SEA and the development of policies and plans for the purposes of natural resource management). Strategic environmental assessment is widely accepted internationally as a critical tool in development planning (e.g. by the World Bank and OECD), where the focus is on impact analysis through to institutional assessment. Strategic environmental assessment is accepted in international development as a way to incorporate environmental considerations across all levels of strategic decision-making including plans, programmes, and policies, setting the context for environmental and social impacts assessments of development projects.

In the context of wildlife management on inhabited islands, we adopt the definition of Russell, et al. (2018) for an inhabited island. Namely that "inhabitation on an island incorporates the basic infrastructure to enable a community to function socially and economically, such as any of schools, churches, community buildings or general shared spaces, alongside enterprises delivering goods and services, and opportunities for residents to pursue a range of livelihood opportunities in the public and private sectors". However, we hasten to add that even when an island is uninhabited, a social framework process may still be required during wildlife management planning where stakeholders and others with vested interests in the island can be identified.

Poor or inconsistent planning is well known in other sectors to delay project completion (Flyvbjerg, 2014). To avoid this problem, we consider wildlife management on islands, and particularly eradication of invasive species, should be treated in the same way as any large-scale, multi-component development project, whereby SEA is a valuable unifying framework that draws together a collection of tools. Many of the tools under SEA are already becoming increasingly applied when planning invasive species management, such as social profiling (Russell, et al., 2018), social impact assessment (Crowley, et al., 2017b), and participatory processes (McEntee & Johnson, 2016). Other tools, such as EIA and economic cost-benefit analyses, can work under the umbrella of SEA for specific eradication projects, once the strategic framework is in place. In particular, eradication practitioners globally should adopt a best practice approach when working with communities on inhabited islands, as they already do for technical best practice when planning the operational elements of eradications on islands (Keitt, et al., 2015).

Most importantly, SEA provides the policy tool by which the role of invasive species eradication as a conservation intervention can cascade throughout all levels of the decision-making process on islands, including deliberative and more participatory approaches (Sims, 2012). This more comprehensive approach applies not just to decisions about wildlife management, but around sustainability of the environment and the livelihoods of human communities on islands. We see this as critical to avoid the pitfalls that previous eradication propositions on inhabited islands haven fallen into – namely where invasive species eradication is considered only as a technical solution to a wildlife management problem on a project by project basis (isolated from other island issues and strategies), and where the support for eradication is seen as merely needing to gain a public consensus through democratic process.

UNESCO WORLD HERITAGE ISLANDS

Many island groups are listed as UNESCO World Heritage sites based on their cultural and natural heritage values. A subset of the islands listed for natural heritage values are also inhabited. In this section we explore the contrasting experiences of enabling introduced small mammal predator management on two similar inhabited UNESCO World Heritage islands where such predator management has been proposed; Fernando de Noronha, Brazil and Lord Howe Island, Australia. These are not the only UNESCO World Heritage islands where predator management takes place. Predator management is also undertaken on Fraser Island, Australia but within the context of a suite of different social and environmental issues related to dingo management (Allen, et al., 2018), and has also been considered on Gough Island (Varnham, et al., 2011), and undertaken on islands in the Galapagos (Carrion, et al., 2011) and Ogasawara Islands (Hashimoto, 2010).

Fernando de Noronha

Fernando de Noronha and Atol das Rocas Reserves in Brazil was assigned UNESCO World Heritage status in 2001. Fernando de Noronha is an archipelago, comprising the primary island of the same name and 20 secondary islands and islets, lying 345 km north east of Brazil in the tropical Atlantic Ocean. The inhabited centre of the island is classified as an Environmental Protection Area (APA), while the uninhabited forested outer areas of the island are part of the Marine National Park (PARNAMAR). Both areas are environmentally administered at the federal level by ICMBio, but socio-politically administered at the state level by neighbouring Pernambuco state on the continent. The resident population of Fernando de Noronha is estimated at around 3,000 people (IBGE 2016). Tourism is the major enterprise on Fernando de Noronha (de Oliveira, 2003), and an estimated 500 tourists arrive and depart each day. This has led the state government to impose a daily tourist tax for environmental protection <http://www.ilhadenoronha.com.br/ailha/taxadepreservacao_em_noronha.php>. However, it is only the regulation of visitor numbers and not proceeds of the tax which contribute directly to environmental protection.

Today the major invasive species on Fernando de Noronha are cats (*Felis catus*), black (*Rattus rattus*) and brown rats (*R. norvegicus*), and the introduced tegu (*Salvator merianae*) lizard (Abrahão, et al., 2019). In Brazil, invasive species are not widely acknowledged as a threat to biodiversity (Bellard & Jeschke, 2016), and any management of invasive species on Fernando de Noronha typically reflects a public health and continental mind set. Wildlife is managed only in the context of vectors of disease (Magalhães, et al., 2017) while cats are managed as companion animals with strict laws administered from the governing Pernambuco state which do not permit lethal control of cats unless their own welfare is suffering (Dias, et al., 2017). The tegu is a CITES listed native species from continental South America, which is also likely to be having severe predatory impacts on the island fauna (Abrahão, et al., 2019).

Management of invasive species on Fernando de Noronha lacks an island conservation context which acknowledges the severe impact such species are having on the biodiversity of the island, and does not engage in lethal control (Russell, et al., 2016b). These biodiversity impacts are not able to be considered alongside other social and economic issues on Fernando de Noronha, as independent levels and agencies of government are in charge of each separately. Strategic environmental assessment would allow proposals for the management of invasive species on Fernando de Noronha to be placed within their broader social context, where invasive species can be considered both as public health pests and companion animals. Impact assessments of invasive species on both the environment and society are absent but could be contemporaneously created. The island's environmental aesthetic (e.g. beaches) is known to be the main driver of tourism, and generates considerable wealth each year, but it is unknown what role the island's biota (e.g. unique endemic species) play in tourism. Strategic environmental assessment would allow the costs of invasive species on the wider economy to be properly calculated, alongside the potential added value to tourism from invasive species management if not eradication. It would play a role in assessing institutional preparedness for embarking on invasive species management and incorporating invasive species management in wider environmental issues such as pollution and island development. In doing so this would ensure that invasive species management was not marginalised against other critical development issues on the island such as poverty and unemployment.

Lord Howe Island

Lord Howe Island, in Australia, was inscribed UNESCO World Heritage status in 1982. Lord Howe Island is an archipelago, comprising the primary island of the same name and 27 secondary islands and islets, 600 km east of Australia in the Tasman Sea. The island is administered as part of the state of New South Wales and for legal purposes is regarded as an unincorporated area administered by the Lord Howe Island Board which reports to the New South Wales Minister for Environment and Heritage. The resident population of the island is around 350 people. Tourism is the primary enterprise on Lord Howe Island but the Kentia palm (*Howea forsteriana*) industry also contributes to the local economy (Gillespie & Bennett, 2017).

The major invasive species on Lord Howe Island are black rats and mice (*Mus musculus*) (Wilkinson & Priddell, 2011). Eradication of rodents from Lord Howe Island would accrue both biodiversity (Hutton, et al., 2007) and economic benefits (Gillespie & Bennett, 2017). It would specifically facilitate reintroduction of the critically endangered Lord Howe Island stick insect (*Dryococelus australis*) (Hutton, et al., 2007) from its last remaining wild habitat on nearby tiny, precipitous Ball's Pyramid. Eradication of the rats and mice on Lord Howe Island was first proposed in 2001 followed by a series of technical feasibility studies (Saunders & Brown, 2001, Parkes, et al., 2003). Planning commenced in 2006 (Wilkinson & Priddell, 2011) and a draft eradication plan was published in 2009 (LHI Board, 2009). Whereas a number of other eradications of invasive species have occurred on islands belonging to Australia (Priddell, et al., 2011), the eradication of rodents on Lord Howe Island would be the first to take place on an inhabited island, particularly in the strict sense of our more comprehensive definition of inhabitation (i.e. communities and facilities). However, the original proposal to eradicate rodents from Lord Howe Island was met with prolonged resistance by elements of the island community.

Management of invasive species on Lord Howe Island is undertaken in an island conservation context which acknowledges the severe impact such species are having on the biodiversity and economy of the island and engages in lethal control. Nonetheless, on Lord Howe Island resistance to rodent eradication was prolonged from a lack of application of social tools (Russell, et al., 2018), although at the time Lord Howe Island was one of the first inhabited islands where rodent eradication was being actively pursued. Ultimately, a number of tools from SEA have now been applied independently, including an environmental impact assessment (LHI Board, 2016), economic cost-benefit analysis (Gillespie & Bennett, 2017), and human health risk assessment (O'Kane, 2017) Strategic environmental assessment would have allowed the planning of rodent eradication on Lord Howe Island to take place using the most appropriate tools for engaging with a resident community that had unanticipated levels of hostility towards the overall proposal. Tools from an SEA framework would have helped identify the various underlying threads of the resistance to rodent eradication in a community that was already accepting of lethal rodent control for the same values at those proposing rodent eradication. Whereas it was initially believed providing more evidential information on the need for eradication and the expected biodiversity benefits alone would be sufficient to gain support for rodent eradication (Wilkinson & Priddell, 2011), this is now known to play only a small role in invasive species planning (Crowley, et al., 2017a), and SEA would have provided tools for a greater participatory process in the rodent eradication planning on Lord Howe Island.

Summary

Although Fernando de Noronha and Lord Howe Island are very similar in geography, they share only a few consistencies in governance and structure, e.g. on both islands the government remains the land-owner and residents are all lease-holders. Otherwise, the generally vast differences in cultures and governance (Reis & Hayward, 2013) mean that planning for invasive species management must be considered in very different contexts on each island. On Fernando de Noronha SEA would have fostered the consideration of invasive species impacts within wider environmental and societal issues, whereas on Lord Howe Island SEA would have provided guidance on the appropriate tools for community engagement to move beyond rodent control to eradication. Thus, the over-arching framework of SEA would have been applied differently on each island to reflect their different contexts and experiences.

NEW ZEALAND ISLANDS

New Zealand has led the world in invasive mammal eradications, with about one third of its islands having been cleared of all invasive mammals (Towns, et al., 2013). These successes have spurred the country to propose the Predator Free New Zealand ambition to eradicate stoats (*Mustela erminea*), rats and brushtail possums (*Trichosurus vulpecula*) from the entirety of the archipelago by 2050 (Russell, et al., 2015). A necessary stepping stone to this goal would entail removing invasive mammals from the large offshore islands of Aotea (Great Barrier Island) and Rakiura (Stewart Island), which would immediately raise the amount of offshore island predator-free land area from 10% to 50%. Discussions and limited planning for invasive mammalian predator eradication from both islands have taken place but using different methods to understand the wider context of, and barriers to, invasive mammal eradication.

Aotea

Aotea comprises a main island of 27,761 ha and numerous surrounding islands and islets, located 17 km north-east from the northern North Island of New Zealand. The island falls within the rohe (tribal boundaries) of Ngati Rehua and has about 800 residents. Seventy percent of the land is owned by the New Zealand Government and is administered by the Department of Conservation. Invasive mammalian predators include cats, black rats, Pacific rats (*R. exulans*) and mice. Mustelids, brushtail possums and hedgehogs (*Erinaceus europaeus*) are notably absent. Large predator control projects at the sub-island level currently occur at Windy Hill Sanctuary (770 ha) and Glenfern Sanctuary (230 ha), and invasive mammals have been removed from numerous surrounding offshore islands (Clout & Russell 2006). A number of bird species are currently at risk of island extirpation including red-crowned parakeets (*Cyanoramphus novaezelandiae*) and tomtits (*Petroica macrocephala*), and the last remaining kokako (*Callaeas wilsoni*) were removed in 1994 to nearby Hauturu. Whole-island eradication of feral cats and rodents was first proposed in 2003, but was met with prolonged resistance by elements of the island community (Ogden & Gilbert, 2011).

A number of tools from SEA have been applied independently on Aotea to better understand the position of the local community towards invasive mammalian predator eradication. In 2015 a participatory process was initiated in the community to understand community perspectives and aspirations towards the overall ecology of the island (McEntee & Johnson, 2015; McEntee & Johnson, 2016). This participatory process identified that the community's perspective on invasive mammal eradication could not be disassociated from their broader economic and social aspirations, and that any investment in invasive mammal eradication had to be part of a broader investment in the community itself. It also identified underlying conflicts in the community such as the tension between the value of isolation versus the desire to increase tourism, and between the desire to control invasive predators versus the value of a toxin-free environment.

A social profiling exercise was also undertaken in 2015 alongside an assessment of the community's attitudes to invasive species management (Aley, 2016; Russell, et al., 2018). This exercise found that there was a higher level of uncertainty with respect to supporting eradication than found on other neighbouring islands, but the social profile of Aotea was not markedly dissimilar to other neighbouring islands in the Hauraki Gulf, although all the islands were markedly different from a corresponding sample in neighbouring Auckland city. This suggested overall that the community's position on invasive mammal eradication was potentially driven by unique recent experiences and exposure to ideas, rather than anything in its social profile, although there did appear to be an overriding island archetype for all the islands in the study, even though one had already had invasive rats eradicated from it (Russell, et al., 2018).

Rakiura

Rakiura comprises a main island of 174,600 ha and numerous surrounding islands and islets, located 27 km south from the southern South Island of New Zealand. The island falls within the rohe of Ngai Tahu and has about 450 residents. Eighty-five percent of the land is owned by the New Zealand Government and is administered by the Department of Conservation. Invasive mammalian predators include cats, black rats, brown rats, Pacific rats, brushtail possums and hedgehogs. Mustelids and mice are notably absent. Large predator control projects at the sub-island level currently occur at Mamaku Point Conservation Reserve (172 ha; previously Dancing Star Conservation Estate), and invasive mammals have been removed from numerous surrounding offshore islands (Clout & Russell, 2006). Although a number of endangered bird species rare on the main islands of New Zealand are abundant on Stewart Island, the last remaining kakapo (*Strigops habroptilus*) were removed in 1992 to nearby offshore Whenua Hou. Whole-island eradication of feral cats and rodents was first proposed in 2008 (Beaven, 2008), but was also met with local resistance.

Rakiura is another case where a number of tools from SEA have been applied independently in an ad hoc manner, not preceded by any attempt to better understand the position of the local community towards invasive mammalian predator eradication. In 2013 a technical feasibility study for removing all invasive mammal predators from Rakiura was undertaken (Bell & Bramley, 2013). This technical feasibility study found that the eradication of invasive mammalian predators from Rakiura was not possible with today's technology, but a sub-island level project around Halfmoon Bay would be feasible. Subsequently a sub-island level project (4,800 ha) consisting of a predator-proof fence protecting the northern peninsula at Halfmoon Bay was proposed as an interim step to achieving a predator-free Rakiura, including technical reports on the predator-proof fence design (Bell, 2014a) and predator eradication methodology (Bell, 2014b). The report on the fence design emphasised the necessity of a predator-proof fence in order to achieve invasive mammalian predator eradication on the peninsula, while the report on predator eradication methodology presented a suite of options for the community to be consulted upon.

In 2014, an economic cost-benefit assessment of invasive mammalian predator eradication for both Rakiura and Halfmoon Bay was also undertaken (Morgan & Simmons, 2014). This report found that eradication was unlikely to have a net positive economic gain from tourism alone but became positive with the addition of ecosystem service valuation. The report also emphasised that anticipated economic and social benefits from invasive mammal eradication may not necessarily eventuate unless the community had a plan and processes in place to capitalise upon them. Despite the substantial investment in technical scoping and community lobbying for a predator-free Rakiura and the Halfmoon Bay project, there remains a level of resistance to both projects on the island along with multiple local proposals and efforts towards enhanced biodiversity (Russell, et al. 2017b).

Summary

The human communities on both Aotea and Rakiura exist in a similar cultural space, and the islands have remarkably similar ecological histories of bird loss, despite being at opposite latitudes of New Zealand. However, both islands illustrate the importance of drawing on the full set of tools available in SEA to build a comprehensive understanding of the perceived and real barriers to implementing an invasive species eradication programme. On Aotea, an SEA approach would have brought the technical and economic aspects of predator eradication into the community discussion earlier, alongside the social elements. When done properly this could have reduced uncertainty in the technical aspects of the proposed eradication, and addressed broader livelihood elements, particularly with respect to the economy, which are important issues on the island. In contrast, on Rakiura an SEA approach would have identified much earlier in the planning process the importance of including social assessment alongside technical and economic cost-benefit assessment, and drawn all three threads together simultaneously to identify that the most immediate barriers to predator eradication on Rakiura, or even in Halfmoon Bay, reflect existing political structures and economic development issues on the island.

DISCUSSION

In this paper we have outlined the process of SEA and how it might specifically be applied to wildlife management, with an emphasis on invasive species management and eradication on inhabited islands. We have reflected on lessons learnt from case studies on four inhabited islands around the world where invasive species are the primary threat to biodiversity, while also impacting on other elements of island livelihoods. Strategic environmental assessment captures a broad suite of tools, including EIA and SIA. Not all of the tools which are a part of SEA may need to be implemented on every island, and SEA allows the application of more context-specific tools such as SIA, and subsequent community engagement and collaborative planning. Importantly, SEA is not a single, linear or one-off process. As stated at the outset, it is an ongoing adaptive and iterative process which adds value to and builds capacity in existing systems. For eradications on inhabited islands the target system is the island community itself, including both human and non-human organisms. This should come as no surprise as it is now readily accepted that environments with humans in them must be managed as joint socio-ecological systems (González, et al., 2008).

We encourage eradication project managers to identify at the outset which SEA tools should be applied in any given project (e.g. Crandall et al. 2018), and to implement those tools in a consistent manner across projects. Governments should also develop a standardised planning and reporting process for invasive species eradication programmes. However, it is important to note that SEA is not a panacea to the challenges faced by practitioners wishing to implement invasive species eradication programmes on islands. Strategic environmental assessment can still be prone to biases either towards values or technical evidence in decision-making processes (Kørnøv & Thissen, 2000). In some cases, whole-island eradication may simply not be an optimal nor achievable goal, due to technical, ecological social, economic or political barriers (Russell, et al., 2015). This does not mean eradication should not remain an aspirational goal (e.g. Predator Free New Zealand), but that in the meantime focus is directed to conservation interventions which maximise return-on-investment in the broadest sense, e.g. invasive species management at the sub-island level.

The application of SEA in a conservation context has the added benefit of bringing wildlife management and invasive species eradication more strongly into the ambit of a broader application of SEA to island development. This would enable the wider benefits of invasive species eradication to be realised, such as on public health (de Wit, et al., 2017) and in primary industry (Nimmo-Bell, 2009). It would also allow the benefits to be incorporated into the international Sustainable Development Goals, such as reduced inequalities through the more equitable distribution of resources for invasive species eradication across developed and developing island nations. For instance, in small island developing states (Russell, et al., 2017a), which are predominantly tropical and home to unique biodiversity not found elsewhere and at risk from invasive species such as mammalian predators (Russell & Holmes, 2015). Undertaking an SEA approach to invasive species eradication on islands will ultimately ensure the longevity of eradications on islands, and alongside enabling eradications on islands in the first instance, will have immediate benefits in the implementation and maintenance of biosecurity on islands (e.g. Russell, et al., 2017a).

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REFERENCES

Abrahão, C., Russell, J.C., Silva, J.C.R., Ferreira, F. and Dias, R. (2019). 'Population assessment of a novel island invasive: tegu (*Salvator merianae*) of Fernando de Noronha'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 317–325. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Aley, J. (2016). 'Environmental and Pest Management Attitudes of Hauraki Gulf Island Communities'. MSc thesis. Auckland, New Zealand: University of Auckland.

Allen, B.L., Cox, T.E., Fleming, P.J., Meek, P.D. and Russell, J.C. (2018). 'Wildlife conservation management on inhabited islands'. *Australasian Journal of Environmental Management* 25(1): 1–4.

Beaven, B. (2008). *Scoping the Potential to Eradicate Rats, Cats and Possums from Stewart Island/Rakiura*. Oban: Stewart Island/Rakiura Community & Environment Trust.

Bell, E.A. (2019). 'It's Not All Up in the Air: The development and use of ground-based rat eradication techniques in the UK'. In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) *Island invasives: scaling up to meet the challenge,* pp. 79–87. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

Bell, P. and Bramley, A. (2013). *Eliminating Predators from Stewart Island: Scoping Report to Investigate Issues of Technical Feasibility*. Wellington, New Zealand: Department of Conservation.

Bell, P. (2014a). 'Predator Free Halfmoon Bay Project: *Technical Aspects of the Project – Preliminary Outline*'. Wellington, New Zealand: Department of Conservation.

Bell, P. (2014b). *Predator Free Halfmoon Bay Project: The Predator Fence – Preliminary Outline'*. Wellington, New Zealand: Department of Conservation.

Bellard, C. and Jeschke, J.M. (2016). 'A spatial mismatch between invader impacts and research publications'. *Conservation Biology* 30(1): 230–232.

Carrion, V., Donlan, C.J., Campbell, K.J., Lavoie, C. and Cruz, F. (2011). 'Archipelago-wide island restoration in the Galápagos Islands: reducing costs of invasive mammal eradication programs and reinvasion risk'. *PLOS One* 6(5): e18835.

Clout, M.N. and Russell, J.C. (2006). 'The Eradication of Mammals from New Zealand Islands'. In: F. Koike, M.N. Clout, M. Kawamichi, M. de Poorter and K. Iwatsuki (eds.) *Assessment and Control of Biological Invasion Risks,* , pp. 127–141. Gland, Switzerland: IUCN.

Courchamp, F., Hoffmann, B.D., Russell, J.C., Leclerc, C. and Bellard, C. (2014). 'Climate change, sea-level rise, and conservation: keeping island biodiversity afloat'. *Trends in Ecology & Evolution* 29(3): 127–130.

Crandall, S.G., Ohayon, J.L., de Wit, L.A., Hammond, J.E., Melanson, K.L., Moritsch, M.M., Davenport, R., Ruiz, D., Keitt, B., Holmes, N.D., Packard, H.G., Bury, J., Gilbert, G.S. and Parker, I.M. (2018). 'Best practices: social research methods to inform biological conservation'. *Australasian Journal of Environmental Management* 25(1): 6–23.

Crowley, S.L., Hinchliffe, S. and McDonald, R.A. (2017a). 'Conflict in invasive species management'. *Frontiers in Ecology and the Environment* 15(3): 133–141.

Crowley, S.L., Hinchliffe, S. and McDonald, R.A. (2017b). 'Invasive species management will benefit from social impact assessment'. *Journal of Applied Ecology* 54(2): 351–357.

de Oliveira, J.A.P. (2003). 'Governmental responses to tourism development: three Brazilian case studies'. *Tourism Management* 24(1): 97–110.

de Wit, L.A., Croll, D.A., Tershy, B., Newton, K.M., Spatz, D.R., Holmes, N.D. and Kilpatrick, A.M. (2017). 'Estimating burdens of neglected tropical zoonotic diseases on islands with introduced mammals'. *The American Journal of Tropical Medicine and Hygiene* 96(3): 749–757.

Dias, R.A., Abrahão, C.R., Micheletti, T., Mangini, P.R., de Oliveira Gasparotto, V.P., de Jesus Pena, H.F., Ferreira, F., Russell, J.C. and Silva, J.C.R. (2017). 'Prospects for domestic and feral cat management on an inhabited tropical island'. *Biological Invasions* 19(8): 2339–2353.

Flyvbjerg, B. (2014). 'What you should know about megaprojects and why: an overview'. *Project Management Journal* 45(2): 6–19.

Gillespie, R. and Bennett, J. (2017). 'Costs and benefits of rodent eradication on Lord Howe Island, Australia'. *Ecological Economics* 140: 215–224.

Glen, A.S., Atkinson, R., Campbell, K.J., Hagen, E., Holmes, N.D., Keitt, B.S., Parkes, J.P., Saunders, A., Sawyer, J. and Torres, H. (2013). 'Eradicating multiple invasive species on inhabited islands: the next big step in island restoration?' *Biological Invasions* 15(12) :2589–2603.

González, J., Montes, C., Rodríguez, J. and Tapia, W. (2008). 'Rethinking the Galapagos Islands as a complex social-ecological system: Implications for conservation and management'. *Ecology and Society* 13(2): 13.

Hanson, C.C., Jolley, W.J., Smith, G., Garcelon, D.K., Keitt, B.S., Little, A.E. and Campbell, K.J. (2015). 'Feral cat eradication in the presence of endemic San Nicolas Island foxes'. *Biological Invasions* 17(4): 977–986.

Hashimoto, T. (2010). 'Eradication and ecosystem impacts of rats in the Ogasawara Islands'. In: K. Kawakami and I. Okochi (eds.) *Restoring the Oceanic Island Ecosystem*, pp.153–159. Tokyo, Japan: Springer.

Holmes, N.D., Griffiths, R., Pott, M., Alifano, A., Will, D., Wegmann, A.S. and Russell, J.C. (2015). 'Factors associated with rodent eradication failure'. *Biological Conservation* 185: 8–16.

Howald, G., Donlan, C., Galván, J.P., Russell, J.C., Parkes, J., Samaniego, A., Wang, Y., Veitch, D., Genovesi, P., Pascal, M., Saunders, A. and Tershy, B. (2007). 'Invasive rodent eradication on islands'. *Conservation Biology* 21(5): 1258–1268.

Hutton, I., Parkes, J.P. and Sinclair, A.R.E. (2007). 'Reassembling island ecosystems: the case of Lord Howe Island'. *Animal Conservation* 10(1): 22–29.

IBGE. 2016. Censo Demográfico Fernando de Noronha (Pernambuco). <http://cod.ibge.gov.br/V6F>.

Jones, H.P., Holmes, N.D., Butchart, S.H., Tershy, B.R., Kappes, P.J., Corkery, I., Aguirre-Muñoz, A., Armstrong, D.P., Bonnaud, E., Burbidge, A.A., Campbell, K., Courchamp, F., Cowan, P.E., Cuthbert, R.J., Ebbert, S., Genovesi, P., Howald, G.R., Keitt, B.S., Kress, S.W., Miskelly, C.M., Oppel, S., Poncet, S., Rauzon, M.J., Rocamora, G., Russell, J.C., Samaniego-Herrera, A., Seddon, P.J., Spatz, D.R., Towns, D.R. and Croll, D.A. (2016). 'Invasive mammal eradication on islands results in substantial conservation gains'. *Proceedings of the National Academy of Sciences, USA* 113(15): 4033–4038.

Keitt, B., Griffiths, R., Boudjelas, S., Broome, K., Cranwell, S., Millett, J., Pitt, W. and Samaniego-Herrera, A. (2015). 'Best practice guidelines for rat eradication on tropical islands'. *Biological Conservation* 185: 17–26.

Kørnøv, L. and Thissen, W.A. (2000). 'Rationality in decision-and policy-making: Implications for strategic environmental assessment'. *Impact Assessment and Project Appraisal* 18(3): 191–200.

Lord Howe Island Board (2009). *Draft Lord Howe Island Rodent Eradication Plan*. Lord Howe Island: Lord Howe Island Board.

Lord Howe Island Board (2016). *Lord Howe Island Rodent Eradication Project – Public Environment Report*. Lord Howe Island: Lord Howe Island Board.

Magalhães, F.J., Ribeiro-Andrade, M., Souza, F.M., Lima Filho, C.D., Biondo, A.W., Vidotto, O., Navarro, I.T. and Mota, R. A. (2017). 'Seroprevalence and spatial distribution of *Toxoplasma gondii* infection in cats, dogs, pigs and equines of the Fernando de Noronha Island, Brazil'. *Parasitology International* 66(2): 43-46.

McEntee, M. and Johnson, S. (2015). *Enabling an Ecological Vision for Aotea Great Barrier Island: Understanding Community Perspectives and Aspirations*. Auckland, New Zealand: Aranovus Ltd.

McEntee, M. and Johnson, S. (2016). *Aotea Great Barrier Island Ecology Vision: Weaving the Tapestry*. Auckland, New Zealand: Aranovus Ltd.

Morgan, G. and Simmons, G. (2014). *Predator-Free Rakiura: An Economic Appraisal*. Wellington, New Zealand: The Morgan Foundation.

Morgan, R.K. (2012). 'Environmental impact assessment: The state of the art'. *Impact Assessment and Project Appraisal* 30(1): 5–14.

Nimmo-Bell. (2009). *Economic Costs of Pests to New Zealand*. Technical Paper No: 2009/31. Wellington, New Zealand: MAF Biosecurity.

O'Kane, M. (2017). *Report on the Human Health Risk Assessment for the Lord Howe Island's Proposed Rodent Eradication Program*. NSW Chief Scientist & Engineer.

OECD (2006). *Applying Strategic Environmental Assessment: Good Practice Guidance for Development Co-operation*. Organisation for Economic Co-operation and Development. DAC Guidelines and Reference Series, OECD Publishing.

Ogden, J. and Gilbert, J. (2011). 'Running the gauntlet: advocating rat and feral cat eradication on an inhabited island – Great Barrier Island, New Zealand'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management,* pp. 467–471. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Oppel, S., Beaven, B.M., Bolton, M., Vickery, J. and Bodey, T.W. (2011). 'Eradication of invasive mammals on islands inhabited by humans and domestic animals'. *Conservation Biology* 25(2): 232–240.

Parkes, J., Ruscoe, W., Fisher, P. and Thomas, B. (2003). *Benefits, Constraints, Risks and Costs of Rodent Control Options on Lord Howe Island*. Unpublished report to the Lord Howe Island Board. Lincoln, New Zealand: Landcare Research.

PCE. (2017). *Taonga of an Island Nation: Saving New Zealand's Birds*. Wellington, New Zealand: Parliamentary Commissioner for the Environment.

Priddel, D., Carlile, N., Wilkinson, I. and Wheeler, R. (2011). 'Eradication of exotic mammals from offshore islands in New South Wales, Australia'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 337–344. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Reis, A. and Hayward, P. (2013). 'Pronounced particularity: A comparison of governance structures on Lord Howe Island and Fernando de Noronha'. *Island Studies Journal* 8(2): 285–298.

Russell, J.C. and Holmes, N.D. (2015). 'Tropical island conservation: rat eradication for species recovery'. *Biological Conservation* 185: 1–7.

Russell, J.C., Innes, J.G., Brown, P.H. and Byrom, A.E. (2015). 'Predator-free New Zealand: Conservation country'. *BioScience* 65(5): 520–525.

Russell, J.C., Cole, N.C., Zuël, N. and Rocamora, G. (2016a). 'Introduced mammals on Western Indian Ocean islands'. *Global Ecology and Conservation* 6: 132–144.

Russell, J.C., Jones, H.P., Armstrong, D.P., Courchamp, F., Kappes, P.J., Seddon, P.J., Oppel, S., Rauzon, M.J., Cowan, P.E., Rocamora, G., Genovesi, P., Bonnaud, E., Keitt, B.S., Holmes, N.D., Tershy, B.R. (2016b). 'Importance of lethal control of invasive predators for island conservation'. *Conservation Biology* 30(3): 670–672.

Russell, J.C., Meyer, J.Y., Holmes, N.D and Pagad, S. (2017a). 'Invasive alien species on islands: impacts, distribution, interactions and management'. *Environmental Conservation* 44(4): 359–370.

Russell, J.C., Taylor, C.N. and Aley J.P. (2018). 'Social assessment of inhabited islands for wildlife management and eradication'. *Australasian Journal of Environmental Management* 25(1): 24–42.

Russell, K.J., Taylor, C.N., Balanovic, J.X., Aley, J.P., Harbow, M.A. and Russell, J.C. (2017b). *Predator Free Rakiura Social Impact Assessment*. A report for the Department of Conservation. Auckland, New Zealand: Uniservices Ltd.

Saunders, A. and Brown, D. (2001). *An Assessment of the Feasibility of Eradicating Rodents from the Lord Howe Island Group*. Unpublished report to the Lord Howe Island Board.

Sims, L. (2012). 'Taking a learning approach to community-based strategic environmental assessment: results from a Costa Rican case study'. *Impact Assessment and Project Appraisal* 30(4): 242–252.

Taylor, C.N., Bryan, C.H., Goodrich, C.G. (2004) *Social Assessment: Theory, Process and Techniques*. Middleton, Wisconsin: Social Ecology Press.

Taylor, C.N. and Mackay, M. (2016). 'Practice issues for integrating strategic social assessment into the setting of environmental limits: insights from Canterbury, New Zealand'. *Impact Assessment and Project Appraisal* 34(2): 110–116.

Tetlow, M.F. and Hanusch M. (2012). 'Strategic environmental assessment: the state of the art'. *Impact Assessment and Project Appraisal* 30(1): 15–24.

Towns, D.R., West, C.J. and Broome, K.G. (2013). 'Purposes, outcomes and challenges of eradicating invasive mammals from New Zealand islands: an historical perspective'. *Wildlife Research* 40(2): 94–107.

Varnham, K., Glass, T. and Stringer, C. (2011). 'Involving the community in rodent eradication on Tristan da Cunha'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 504–507. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Wilkinson, I.S. and Priddel, D. (2011). 'Rodent eradication on Lord Howe Island: challenges posed by people, livestock, and threatened endemics'. In: C.R. Veitch, M.N. Clout and D.R. Towns (eds.) *Island invasives: eradication and management*, pp. 508–514. Occassional Paper SSC no. 42. Gland, Switzerland: IUCN and Auckland, New Zealand: CBB.

Towards a guidance document for invasive species planning and management on islands

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Abstract In 2012 a process was initiated to produce a guidance document for invasive species management on islands, as an objective of a regional invasive species project in the Western Indian Ocean (WIO) islands, implemented by IUCN. The consultative process for producing the document began with requests and discussions via regional and global island and invasives email distribution lists. Initial responses revealed a consensus on the need for a guidance document for programmatic planning. A draft was therefore constructed around existing Pacific regional guidelines and a draft manual that had initially been written for the WIO, with new supplementary sections suggested by respondents. The new draft was discussed and revised in workshops at two international conferences. The document is now organised into three main sections: the first on how to use it, the second a checklist of the essential components of a comprehensive island invasives programme (to ensure nothing is overlooked when planning), and the third providing detailed guidance on the planning and decision-making processes. The document is intended to provide a comprehensive framework and procedural guide for invasive species planning on islands. Further consultations took place by email, and a later draft was tested by a number of users writing various kinds of invasive species strategy and action plan. Publication will be in English, French (both published 2018) and Spanish (scheduled for 2019).

Keywords: best practice, consultative planning, NBSAP, networks, NISSAP, prioritisation

INTRODUCTION

The intentional and unintentional movement by people of organisms around the world, many of which become invasive in the areas to which they are introduced, is an international problem of particular concern to islands. The effective management of invasive species on islands therefore requires comprehensive and coordinated action by international agencies, governments, NGOs, the private sector and local communities. Small islands and developing states do not have the resources to tackle all invasive threats by themselves, and in some regions collaborative regional plans and strategies have been developed to promote coordinated planning and action between islands and nations and also to guide international agencies in providing assistance to them. Many countries have also developed National Invasive Species Strategies and Action Plans (NISSAPs), as have a number of individual islands (for brevity, all such plans are herein termed NISSAPs). The Convention on Biological Diversity recognises invasive species as a serious threat, including in its Aichi Target 9, and encourages countries to include plans for managing invasive species in their National Biodiversity Strategies and Action Plans (NBSAPs). However, the NBSAPs and NISSAPs of different islands and island countries vary greatly in their comprehensiveness in dealing with invasive species problems (Doherty & Boudjelas, 2010; Boudjelas, in press).

The Inva'Ziles Project, implemented by IUCN from 2012 to 2018, provided assistance to the islands of the western Indian Ocean (WIO) region in managing biological invasions. One of the project's objectives was to develop guidance for islands and island nations to help them prevent and manage the spread of invasive species and reduce their impacts on biodiversity and people's livelihoods. This paper describes the process leading to the production of a guidance document specifically for invasive species planning on islands worldwide. It explains the purpose of the document and outlines progress towards its publication, including input from the 3rd Island Invasives conference in Dundee, July 2017.

FIRST STEPS

The Inva'Ziles project began with a broad interpretation of its commitment to produce a guidance document for invasives management on islands, by compiling a first draft of a manual attempting to cover the whole range of actions necessary for an invasive species programme, in the following chapters:

INTRODUCTION

- Importance of biological diversity
- Significance of biological invasion as a disruption of biodiversity

BIOLOGICAL INVASION AS A PROCESS

ELEMENTS OF AN INVASIVES SPECIES STRATEGY

- Regional coordination and exchange
- Risk Assessment
- Prevention without quarantine
- Pathways of introduction
- Early detection and rapid response
- Management of established invasions

MONITORING

CAPACITY

- Institutions
- Awareness
- Information
- Conventions

IMPLEMENTATION OF INVASIVES SPECIES MANAGEMENT

- Policies, laws and regulations
- Institutions and capacity
- Roles and responsibilities of the public
- International and regional responsibilities

GLOBAL CHANGE AND INVASION

In: C.R. Veitch, M.N. Clout, A.R. Martin, J.C. Russell and C.J. West (eds.) (2019). Island invasives: scaling up to meet the challenge, pp. 698–702. Occasional Paper SSC no. 62. Gland, Switzerland: IUCN.

While all these topics are important, there are good resources already available to help planners and managers with many of these activities, including project design, border biosecurity, methods of controlling various species of invasive animal, plant and other organisms, raising awareness, etc. Examples include the guidelines and toolkits of the Global Invasive Species Programme (www.issg.org/gisp_guidelines_toolkits.htm) on marine biofouling (Jackson, 2008), marine pest management (Hilliard, 2005), legal and institutional frameworks (Shine, et al., 2000; Shine, 2008), best prevention and management practice (Preston, et al., 2000; Wittenberg & Cock, 2001), and economic analysis (Emerton & Howard, 2008) along with their accompanying training courses (www.issg.org/gisp_training_coursematerials.htm). There are also many excellent materials developed in individual regions, such as the rodent and cat eradication resource kits for the Pacific (Pacific Invasives Initiative, 2011) and UK (GB Non-native Species Secretariat, 2017), as well as the Pacific kits for invasive plant (Pacific Invasives Initiative, 2015) and ant (New Zealand MFAT, 2016) management and their accompanying training courses. It would be impossible within a single document to improve on all of these and others. Further, it was considered doubtful whether general explanations of biological invasions and their impacts on biodiversity would be necessary for the intended primary users of the document: invasive species planners, managers and researchers on islands. The introductory material covering these topics and the discursive style adopted in the first Inva'Ziles draft limited the amount and clarity of the guidance provided; for example, the draft did not give clear guidance on the steps to be taken when planning an invasives programme, nor on how to prioritise when faced with many problems and limited financial and human resources. It was felt that a short document with a clear purpose and direct guidance would be more useful and used than something longer and more discursive.

It was therefore decided to carry out consultation in order to find out what kind of guidance invasive species workers themselves thought they needed most, so as to be able to focus the planned document more precisely on priority gaps in available resources.

CONSULTATION AND REDRAFTING

Given the Inva'Ziles Project's primary responsibility to provide assistance to the WIO region, an initial consultation was carried out by e-mailing a simple questionnaire to the c. 325 members of the Western Indian Ocean Network on Invasive Species (WIONIS), asking what kind of guidance they felt was most needed. It was essential to give respondents an idea of what might be possible for the project to produce within the limitations of its timespan and budget, so, to encourage realistic answers three possibilities were suggested: a manual-style document resembling the first draft produced by Inva'Ziles, something focused more precisely on the planning and decision-making processes, using the example of the *Guidelines for Invasive Species Management in the Pacific* (SPREP, 2009: hereafter termed the 'Pacific *Guidelines*'), or something else.

This was followed by a similar worldwide consultation using the following global and regional e-mail distribution lists: aliens-l (1,400 subscribers, global), islands-l (360, global), carib-ias (310, Caribbean), the Pacific Invasives Initiative list (1,210), and the Pacific Invasives Partnership (c. 40). In addition, the same request was sent to specifically compiled lists of known contacts in the Atlantic and Mediterranean islands (c. 20 people).

These consultations generated responses from invasive species planners, scientists and managers, including experts in all major island biomes, marine and terrestrial (all contributors up to the submission date of the present article are named in the Acknowledgments). Of the 43 respondents who indicated a clear preference for the kind of document they would like to see developed, two wanted an operations manual for field management and 41 preferred guidance on planning, with no-one suggesting any other kind of document. These choices, taken together with written comments from many other respondents, indicated a consensus that specific guidance on programmatic planning was scarce and lacking in detail, and that this represented a particular resource gap. The Pacific *Guidelines* have been widely adopted and used in that region, and many respondents felt that an updated and internationalised version of this would be highly appropriate for other island regions.

The decision was therefore made to produce a document addressing this need for planning guidance, taking the Pacific *Guidelines* as a model, updating and hopefully improving it, and at the same time endeavouring to make the document as useful and relevant as possible to islands worldwide. A skeleton was then produced by adapting the text of the Pacific *Guidelines* for a global set of users, and adding ideas for new sections suggested by the drafting team, questionnaire respondents and others. The new sections were then partially populated by adapting text from the Inva'Ziles first draft manual.

To expand the consultation process, we used opportunities created by international and regional meetings to obtain further input. Workshops were therefore organised at the IUCN 'World Conservation Congress' (WCC) in Hawai'i, September 2016, and the 3rd 'Island Invasives' conference (3II), Dundee, July 2017. The first of these meetings attracted (as expected) a broad cross-section of conservationists, while the second drew a substantially different group, consisting primarily of invasive species management practitioners and researchers. Both meetings generated contributions from people working on a wide range of aspects of the invasives threat to islands, from all parts of the world.

At the WCC, two events were organised with the objective of obtaining input. First, the IUCN held a major introductory event on '*Islands at risk: meeting the global challenge of Invasive Alien Species*', at which three initial presentations (one on the guidance document) were followed by work-groups on the three topics. The guidance work-group attracted some 30 people, of whom 14 offered to make additional contributions later, as the drafts developed. The second WCC event was a roundtable discussion organised by the Pacific Invasives Partnership, which attracted about 20 people, most of whom had not attended the first working session. At both of these sessions, input was obtained not only for the global guidance document, but also for a planned revision of the Pacific *Guidelines*, led by the Pacific Invasives Partnership.

Comments and ideas received at the WCC were incorporated into a second draft, which included supplementary sections solicited meanwhile from volunteer experts on particular topics. During this process it became clear that guidance on two areas in particular was desired: the planning process itself, including prioritisation and decision-making, and how to increase support for invasives management among politicians, their electorates (the public), and local communities experiencing problems caused by invasives. As a result, these two areas grew to constitute the largest supplementary sections.

At the 3II, the IUCN gave an introductory presentation in plenary to explain the purpose of a working session on the guidelines that evening. Some 50 people came to the evening session (approximately 15% of the conference attendees), which was organised into three work-groups covering different sections of the draft, namely: planning and decision-making; awareness, support and capacity; research and practical management. Twenty of those who attended offered to contribute further.

The steps towards producing this document are illustrated in Fig. 1, and the location or geographical interest of the identified contributors summarised in Table 1. At the time of writing this paper, we were in the process of incorporating comments from the 3IIsland A major outcome from 3II was confirmation from practitioners that the fundamental need for this document was genuine and widespread, and also that guidance on how to use the document should be given clearly within.

THE CURRENT DRAFT

The aim thus became to provide a comprehensive framework and procedural guide for anyone planning an invasives programme on islands, including international and regional agencies, conservation NGOs, relevant government agencies (agriculture, biosecurity, environment …), conservation managers, research planners, and anyone else who has to find, plan and prioritise funds and resources for invasives management.

The latest and final draft met the target limit of 48 pages plus covers (the Pacific *Guidelines* comprises 24 pages including covers), has now been organised into three main sections (plus a "Resources" section). The first of the main sections explains the purpose of the document, how to use it, and who the intended users are. The second section is a checklist of the essential components of a comprehensive island invasives programme, to ensure nothing is overlooked when planning (this part still resembles the Pacific *Guidelines*, which consists mainly of such a checklist). The third section describes in detail how to conduct the processes mentioned by many people as being particularly problematic, especially how to plan, how to prioritise, how to make decisions, and how to increase collaboration, support and involvement by different target groups ranging from local communities to senior policy- and decision-makers. Throughout, there are links to additional resources on each topic.

The document provides decision-making guidance at both programmatic planning and field project planning levels, including how to prioritise, how to choose management goals, and how to win political and community support for the actions planned. It should help international agencies to identify their niche for invasives work on islands and to identify island priority needs that match their agency's expertise. It will help national and local agencies and managers to identify and prioritise actions within their jurisdiction, design a NISSAP, benefit from the experience of other countries and organizations, and justify projects to decision-makers and donors. Content of the three main sections is organised as follows:

INTRODUCTORY MATERIAL

- Purpose of the document, intended users, how to use
- Background

THE GUIDELINES CHECKLIST

- Foundations (planning, decisions, support, capacity, legal)
- Information (baseline, monitoring, prioritisation, research)
- Management (borders, established invaders, restoration)

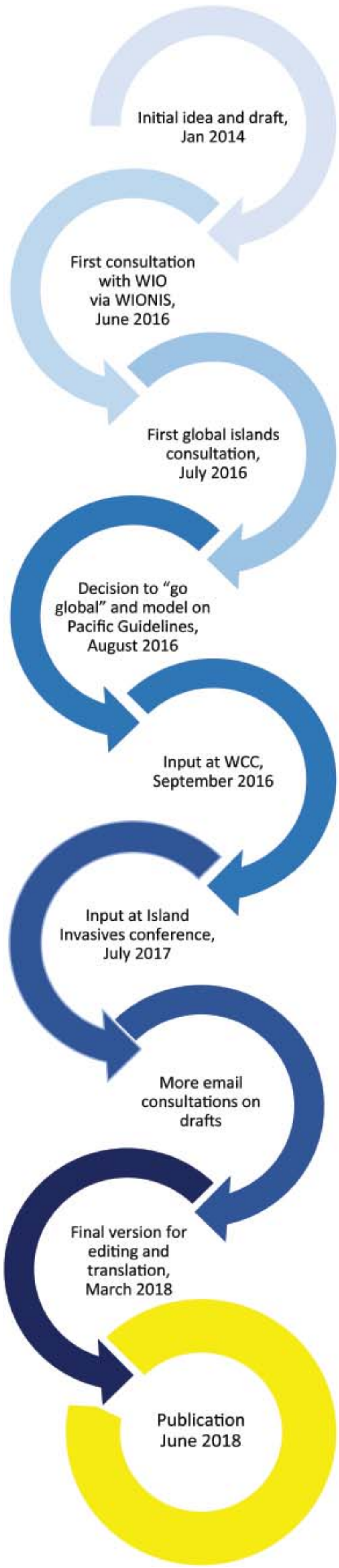

Fig. 1 Timeline of the process of producing the guidance document.

HOW TO PLAN

- Getting people involved, raising support for the plan, mutual help networks
- Programme planning, NISSAPs and others
- Prioritising, hard decisions, decision tools, risk analysis
- Neglected areas
- Planning for global change
- Project planning, other decision tools

The identification of these priority areas for guidance has largely been determined by the views of the respondents. These priorities differ somewhat from the critical areas for action identified almost 20 years ago in the Pacific, when the following were considered to need special attention (SPREP, 2000):

- Shortage and inaccessibility of information on invasive species and best management practice
- **Lack of awareness of the impacts of invasive species**
- **Insufficient networking, coordination and collaboration**
- Inadequate legislation, regulations, cross-sectoral policies, and enforcement
- Shortage of trained personnel, and inadequate facilities
- **Insufficient funding.**

The three items in bold are closely related to the current priority needs identified by our recent consultations. The differences reflect both the fact that some of the other areas, particularly best management practices, have since been addressed by resources specially designed to assist with them, but also the fact that our new document is aimed at perceived needs for guidance itself, rather than at other kinds of need (e.g. adequate facilities, trained personnel, laws etc.).

FINAL STEPS

The final draft was circulated once more to the core group of committed contributors (ultimately just over 100 people contributed) as well as to all of the e-mail distribution lists cited above. Special contributions were solicited from experts on particular themes. The later drafts were tested in a number of planning processes, including for the first NISSAP of the Comoro Islands in mid-2018. The English and French versions were published in print and online in mid-2018 and the Spanish version published online in early 2019.

ACKNOWLEDGEMENTS

The Inva'Ziles Project (official title *Preparation and testing of a comprehensive model for preventing and managing the spread of invasive species on island ecosystems*) was implemented by the International Union for Conservation of Nature (IUCN), funded by the European Union and hosted by the Indian Ocean Commission. Geoffrey Howard and Olivier Tyack (IUCN) drew up the initial concept and first draft of the guidance document. Kosi Latu (SPREP) kindly gave permission to use the Pacific *Guidelines* as a model. Kevin Smith, Olivier Hasinger (IUCN) and David Moverley (SPREP) organised working sessions at the 2016 'World Conservation Congress', and Kevin, together with Souad Boudjelas (Pacific Invasives Initiative) and Jill Key (GB Non-Native Species Secretariat), also organised the session at the 3rd 'Island Invasives' conference; they have all provided invaluable support and contributions throughout. Dick Veitch encouraged me to write this article for the current proceedings, Katharina Lapin (IUCN) constructed Fig. 1, and Olivier Hasinger and Kevin Smith reviewed a draft. The following had contributed to drafts by the date of submission of this article by providing texts, commenting on versions, suggesting areas of concern, or in other ways (this list does not include people who contributed at conferences but did not leave their details): Ademola Ajagbe, Katy Beaver, Alex Bond, Elsa Bonnard, Olaf Booy, Rafael Borroto, Souad Boudjelas, Nancy Bunbury, Earl Campbell, Dario Capizzi, Juli Caujapé-Castells, Alison Copeland, Ana Costa, Franck Courchamp, Phil Cowan, Steve Cranwell, Cathleen Cybèle, Curt Daehler, Maria Cristina Duarte, Julia Dunn, Rui Bento Elias, Marko Filipovic, Julian Fitter, Frauke Fleischer-Dogley, Jason Goldberg, Ines Gómez, Viliami Hakaumotu, Sjurdur Hammer, Olivier Hasinger, Ben Hoffmann, Geoffrey Howard, Stephanie Hudin, Jason Jack, Patricia Jaramillo, Marie-May Jeremie-Muzungaile, Gabe Johnson, Chris Kaiser-Bunbury, Springer Kaye, Inti Keith, John Kelly, Jill Key, Michael Kiehn, Cynthia Kolar, Christoph Kueffer, Janice Lord, Ian MacDonald, Gwen Maggs, Christy Martin, Kelly Martinou, John Mauremootoo, Mathilde Meheut, Tommy Melo, Jean-Yves Meyer, Joel Miles, Aileen Mill, James Millett, Nitya Mohanty, Craig Morley, David Moverley, Bradley Myer, Rachel Neville, Ray Nias, Kimberley O'Connor, Warea Orapa, Shyama Pagad, Julián Pérez, John Pinel, Bruce Potter, Parmenanda Ragen, Frida Razafinaivo, Tim Riding, Gérard Rocamora, James Russell, Susana Saavedra, Adrian Schiavini, Richard Selman, Nirmal Shah, Andy Sheppard, Greg Sherley, Junko Shimura, Didier Slachmuylders, Kevin Smith, Antonio Soares, Yohann Soubeyran, Vikash Tatayah, Anna Traveset, Olivier Tyack, Magdalena Vicens,

Table 1 Islands and island regions represented by the 96 identified contributors so far (each contributor assigned to only one category). OIT signifies overseas island territories of any kind, irrespective of their political status; n = no. of contributors.

Islands	**n**	**Islands**	**n**	**Islands**	**n**
Australia	1	Indian Ocean	3	Pacific	8
Azores	3	Isle of Man	1	Palau	1
Bermuda	1	Japan	1	Papua New Guinea	1
Canaries	2	Kosrae	1	Seychelles	7
Cape Verde Islands	2	Lord Howe Island	1	Tonga	1
Caribbean	1	Madagascar	1	UK & OITs	4
Cuba	1	Mauritius	4	USA, Hawai'i & OITs	9
France & OITs	4	Mediterranean	3	Global, multi-regional or unknown	30
Galapagos	2	New Zealand	3		

Jeanne Wagner, Josua Wainiqolo, Katherine Walls, Andrew Walsh, Masahito Yoshida, Glyn Young, Kristi Young. Thanks to all, and if I've missed anyone, please let me know!

REFERENCES

Boudjelas, S. (in press). A review of national and island plans for the management of invasive species in the Western Indian Ocean region. Pp. XX–XX in Tye, A. (ed.) The State of Knowledge and Planning for Invasive Species Management in the Western Indian Ocean. IUCN, Gland.

Doherty, N. and Boudjelas, S. (2010). *Invasive Species Management in the Pacific: A Review of National Plans and Current Activities*. Unpublished report for the Pacific Invasives Partnership. Auckland: Pacific Invasives Initiative. <www.issg.org/cii/Electronic%20references/pii/references/pii_ism_in_the_pacific_a_review_of_national_plans_and_current_activities.pdf>.

Emerton, L. and Howard, G. (2008). *A Toolkit for the Economic Analysis of Invasive Species*. Global Invasive Species Programme.

GB Non-native Species Secretariat. (2017). *UK Rodent Eradication Best Practice Toolkit*. <www.nonnativespecies.org/index.cfm?pageid=613>.

Hilliard, R. (2005). *Best Practice for the Management of Introduced Marine Pests - A Review*. Global Invasive Species Programme.

Jackson, L. (2008). *Guidelines for the Prevention and Management of Marine Biofouling and Invasive Species*. Global Invasive Species Programme.

New Zealand MFAT. (2016). *Pacific Invasive Ant Toolkit*. <www.piat.org.nz/>.

Pacific Invasives Initiative (2011). *Resource Kit for Rodent and Cat Eradication*. <http://rce.pacificinvasivesinitiative.org/>.

Pacific Invasives Initiative. (2015). *Resource Kit for Invasive Plant Management.* <http://pacificinvasivesinitiative.org/ipm/ipm.pacificinvasivesinitiative.org/index.html>.

Preston, G., Brown, G. and van Wyk, E. (eds.) (2000). *Best Management Practices for Preventing and Controlling Invasive Alien Species*. Cape Town: Working for Water Programme.

Shine, C. (2008). *A Toolkit for Developing Legal and Institutional Frameworks for Invasive Alien Species*. Global Invasive Species Programme.

Shine, C., Williams, N. and Gündling, L. (2000). *A Guide to Designing Legal and Institutional Frameworks on Alien Invasive Species.* Environmental Policy and Law Paper No. 40. Gland, Switzerland: IUCN.

SPREP. (2000). *Draft Invasive Species Strategy for the Pacific Islands Region*. Apia: Secretariat of the Pacific Regional Environment Programme.

SPREP. (2009). *Guidelines for Invasive Species Management in the Pacific*. Apia: Secretariat of the Pacific Regional Environment Programme.

Wittenberg, R. and Cock, M.J.W. (eds.) (2001). *Invasive Alien Species: A Toolkit for Best Prevention and Management Practices*. Global Invasive Species Programme.

Chapter 4: Abstracts

These abstracts are for papers which were presented at the conference, either as oral presentations or poster papers, but for which the authors have chosen not to prepare and publish a full written paper.
These abstracts are given in the alphabetical order of the prime author of the paper with the address of only that first author included.

Mexico's progress and commitment to comprehensive island restoration

A. Aguirre-Muñoz, F. Méndez-Sánchez, L. Luna-Mendoza, A. Ortiz-Alcaraz, J. Hernández-Montoya, Y. Bedolla-Guzmán, M. Latofski-Robles, E. Rojas-Mayoral, N. Silva-Estudillo, F. Torres-García, M. Félix-Lizárraga, A. Fabila-Blanco, A. Hernández-Ríos, E. Bravo-Hernández, F. Solís-Carlos, C. Jáuregui-García and D. Munguía-Cajigas

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For the past 18 years, Mexico has taken bold steps to systematically eradicate invasive mammals. Mexico´s 4,111 islands host 8.3% of the country's plants and land vertebrates. They harbour one in three seabirds worldwide, placing Mexico as the third most diverse country. Invasive mammals have had a big toll on Mexico's biodiversity, with 17 out of 21 confirmed vertebrate extinctions occurring on islands. The Mexican conservation organisation Grupo de Ecología y Conservación de Islas (GECI), in collaboration with Mexico's federal government, and a wide network of national and international donors, has been leading the National Programme for Island Restoration that has grown in scope. The first eradications on small islands fostered trust amongst partners, setting the foundations for complex eradications on bigger islands requiring innovation, capacity development, and research. Island biosecurity is now a priority for long-term tangible results. This programme evolved to be truly comprehensive, including post-eradication restoration to strengthen island resilience, and the social construction of a cultural approach integrating interests from conservation and local fishing communities. Results to date include: (1) eradication of 58 populations of invasive mammals from 37 islands; (2) publication of both a National Island and Invasive Species Strategy, identifying conservation priorities; (3) ongoing active restoration of seabird colonies and native plant communities; (4) original applied research and ad hoc infrastructure and equipment to support restoration; (5) legal protection of all Mexican islands; (6) assessing the effects of climate change on islands' biodiversity and human populations; and (7) formation of in-house specialists through postgraduate studies in collaboration with research institutes and universities from Mexico and elsewhere. As for the future, we foresee two priorities: (1) remove invasive mammals from all Mexican islands by 2030; and (2) promote the creation of an "International Islands Institute" that could operate under a wide international collaboration and interdisciplinary approach.

The Pacific invasives partnership – a model for regional collaboration on invasive alien species

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Invasive Alien Species (IAS) are a fundamental challenge facing Pacific Island Countries and Territories (PICTS), impacting economies, habitats, food security, biodiversity, livelihoods and quality of life. These negative and substantial impacts are being acknowledged by PICTs leaders as well as on the international stage. As the inter-relatedness of IAS and other fundamental challenges such as climate resilience, oceans and sustainability are understood and acknowledged, strategies to integrate IAS and biosecurity concepts into international efforts will require invasive species expertise and guidance. The Pacific Invasives Partnership (PIP) is a group created by the Pacific Roundtable for the Conservation of Nature that has evolved into a broad advocate for IAS outreach and an incubator for collaborative IAS efforts in the Pacific. PIP comprises volunteer IAS experts from regional, national, NGO and international groups that work in two or more PICTS and want to advance IAS issues. By taking a "rising tide floats all boats" approach, PIP members work to raise the profile and understanding of IAS as a fundamental, underpinning issue to PICT economies, environments and future sustainability. PIP successes over the past five years include reports and briefing materials prepared for the Pacific Islands Forum Leaders meeting, provision of advice and assistance for Pacific invasive species Global Environment Facility projects, leading and supporting regional and sub-regional projects on regional biosecurity, invasive ant and rodent eradication and prevention, and the successful raising of the IAS profile at various international fora. PIP is a successful model of regional collaboration on invasive alien species and could be used as a model for similar efforts in other island regions of the world.

A review of monitoring of biodiversity responses to island invasive species eradications

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A recent review examined the benefits from invasive species eradications on islands worldwide. While the review concluded that island eradications are overwhelmingly beneficial for native biodiversity, a response to eradication was only demonstrated for 22 of the 532 islands treated. While many studies advocate monitoring, there appears to be a gap, either between eradication effort and monitoring effort, or between monitoring and analysing/reporting responses. We focussed on regions of the Pacific, Australia and the Caribbean to document the level of monitoring on islands where eradications have taken place. We collated published and unpublished literature and spoke to key practitioners in the region to investigate targets for monitoring, duration and frequency of monitoring, and the ability of implemented monitoring work to detect responses. We also investigated drivers of monitoring such as type of funder or implementing organisation behind the eradication operation. The study's findings highlight apparent biases in monitoring effort, they provide a benchmark of current monitoring effort, and open the debate on when and where monitoring should be undertaken and how best to develop optimal monitoring strategies.

A review of seabird recovery on Lundy Island, England, over a decade following the eradication of brown and black rats

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Lundy, a 450 ha island situated 19 km off the Devon coast in the UK's Bristol Channel, is internationally important for its marine life and its waters were established as the UK's first Marine Nature Reserve in 1971. Lundy is home to eleven seabird species, including Manx shearwater (*Puffinus puffinus*), for which the UK has a global responsibility and Atlantic puffin (*Fratercula arctica*), a globally threatened species. Steep declines in Lundy's seabird populations, with puffins nearing extinction and low numbers of Manx shearwaters, led to the establishment of the Seabird Recovery Project in 2001. The project aimed to improve the conditions for these burrow-nesting seabirds through the eradication of brown and black rats. From 2002–2004 a ground-based operation was undertaken, and in 2006 Lundy was officially declared rat-free. The seabird populations of Lundy have been well studied with detailed regular data spanning the last 35 years. Over the last decade, as a result of rat removal, seabird numbers on the island have doubled and storm petrels have colonised. By 2013, the breeding population of Manx shearwaters increased more than ten-fold to an estimated 3,451 pairs. In 2004, the puffin population on Lundy fell to an all-time low with only five individuals, but in 2013, more than 80 individuals were recorded. Here we discuss the observed seabird responses to the eradication and present the most recent results of the monitoring surveys from 2017. These impressive results highlight the importance of and need for effective biosecurity to reduce the risk of re-incursion of rats. Lundy is a popular tourist destination with a working farm; therefore, the regular transportation of cargo remains a high biosecurity risk. A revised biosecurity and incursion response plan is now being finalised.

Eradicating invasive ants in conservation areas

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Established invasive invertebrates, such as Argentine ants (*Linepithema humile*), can have long-term and cascading adverse ecological impacts for native communities. In Mediterranean ecosystems, they out-compete most native ant species and harm plants such that they interfere with pollination, reducing seed set. In 2013–2016 we developed and carried out a unique treatment protocol on four Argentine ant infestations on Santa Cruz Island, California, totalling 410 ha. We used polyacrylamide beads, hydrated with 6 ppm thiamethoxam and 25% sucrose water distributed at a rate of 148 litres per hectare via helicopter and hopper. We treated the four infestation areas 14 times, for total cost of US$1,400 per ha. Two monitoring strategies used lures and visual searching on 74 ha in 2013–2015, with costs at US$2,200 and US$500 per ha. The less costly, targeted strategy revealed one spot population totalling 0.3 ha. This population was located at the edge of a treatment site, possibly indicting that the 50 m buffer added to that delimited infestation was insufficient. Follow up treatments were conducted on that site and Argentine ants were not detected in subsequent monitoring rounds. Monitoring will continue 2016–2020 throughout all four treatment areas, aided by a fine-scale model of probability of detection and probability of persistence by vegetation type, and detection dogs. Packaged with patience and persistence, these treatment and monitoring protocols show promise as an eradication tool. Preliminary data indicate that the treatment may also be effective in eradication programmes for other invasive ant species.

Big island, small invader: eradicating invasive fish on a national scale

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Topmouth gudgeon (*Pseudorasbora parva*) is an invasive non-native cyprinid from Asia. Listed as a Species of Union Concern under the EU Invasive Alien Species Regulations, it is considered one of the most potentially damaging non-native fish species to invade Western Europe. Introduced to Great Britain (GB) in 1984, evidence indicated that if topmouth gudgeon established in GB, the impacts on our native species and habitats could be severe. The threats were clear, and the case for action robust. However, in 1980s and 1990s GB authorities lacked a coherent invasive species strategy, regulatory powers were ineffective, there was no focused expertise or capacity and the tools and techniques necessary to control such a tenacious invasive species had not been developed or adopted. Topmouth gudgeon spread inexorably across England and Wales, until 2004. By 2004, with seven populations identified, the authorities were no closer to a solution. However, using an innovative biocide-based approach, a local Environment Agency team successfully eradicated topmouth gudgeon from a fishery in the Lake District. This led to a number of small scale, ad hoc eradications, but as confirmed populations climbed to 14, sustainable removal of the species from GB was not considered feasible. In 2011, supported by the GB Invasive Species Strategy, the Environment Agency utilised their growing expertise and capacity to develop a specialist team and equipment and implemented a Water Framework Directive National Programme; their ambitious objective: total eradication of topmouth gudgeon from GB by 2018. Scaling up from small scale, localised eradication to a national landscape scale programme to eradicate an aquatic invasive species was unprecedented and presented significant strategic, legal, operational, economic and political challenges. This paper documents that 12-year journey, highlighting the challenges, discussing how they were overcome, the lessons learnt, and considers the future potential and direction of this work.

Population growth of seabirds after the eradication of introduced mammals

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Eradication of introduced mammals to restore island ecosystems has become increasingly common, with more than 1,000 successful projects around the world. Various benefits for native fauna have been documented, including reduced predation and positive demographic response. However, evidence that these eradications lead to increases in populations of seabirds, which are important island ecosystem engineers, is sparse. The limited amount of monitoring is partly because of seabirds' long life cycle, meaning that several years or even decades may elapse before populations respond to eradication. Drawing on data from across the world, we assemble population growth rates (lambda, λ) of 181 seabird populations of 69 species following successful eradication projects. After successful eradication, the median growth rate was 1.12 and populations with positive growth ($\lambda > 1$; n = 151) greatly outnumbered those in decline ($\lambda < 1$; n = 23) and those that exhibited no change ($\lambda = 1$; n =7). Population growth was faster at newly-established colonies compared to those already established, and in the first few years after eradication before the species' age of first breeding. Because λ was higher before first-time breeders are recruiting back into the colony, this suggests that immigration is important for colony growth. Population growth was also faster among gulls and terns compared to other seabird groups and when several invasive mammals were eradicated together in the course of the restoration project. This reflects the relative lack of philopatry among gulls and terns and reinforces current best practice – the removal of all invasive mammals where feasible. These results may help prioritise sites for future eradication projects and determine where active seabird population management is required after eradication.

Assessment of the possible effects of biological control agents of *Lantana camara* and *Chromolaena odorata* in Davao City, Mindanao, Philippines

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Invasive plants have an impact on global biodiversity and ecosystem function, and their management is a complex and formidable task. Two of these invasive plant species, *Lantana camara* and *Chromolaena odorata*, are found in the Philippines. *Lantana camara* has the ability to suppress the growth of and outcompete neighbouring plants. *Chromolaena odorata* causes serious agricultural and economical damage and causes fire hazards during dry season. In addition, both species have been reported to poison livestock. One of the known global management strategies to control invasive plants is the introduction of biological control agents. These natural enemies of the invasive plants reduce population density and impacts of the invasive plants, resulting in the balance of the nature in their invasion. Through secondary data sources, interviews, and field validation (e.g. microhabitat searches, sweep netting, opportunistic sampling, photo-documentation), we investigated whether the biocontrol agents previously released by the Philippine Coconut Authority (PCA) in their Davao Research Center to control these invasive plants are still present and are affecting their respective host weeds. We confirm the presence of the biocontrol agent of *L. camara*, *Uroplata girardi*, which was introduced in 1985, and *Cecidochares connexa*, a biocontrol agent of *C. odorata* released in 2003. Four other biocontrol agents were found to affect *L. camara*. Signs of damage (e.g. stem galls in *C. odorata,* and leaf mines in *L. camara*) signify that these biocontrol agents have successfully established outside of their release site in Davao. Further investigating the extent of the spread of these biocontrol agents in the Philippines and their damage to the two weeds will contribute to the management of invasive plant species in the country.

Black rat eradication from Linosa Island: work in progress

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The black rat (*Rattus rattus*) is one of the most successful colonising mammals and one of the 100 world's worst invasive species. It is a generalist and opportunistic predator, particularly of seabird eggs and chicks on islands where it has been transported by humans on ships. The Life project "Pelagic Birds: Conservation of the main European population of *Calonectris diomedea* and other pelagic birds on Pelagic Islands" on Linosa Island involves the eradication of black rats, since it is considered the major cause of Scopoli's shearwater breeding failure. From 15 May to 10 October 2013, a preliminary phase was carried out to determine the abundance and the distribution of black rats and house mice (*Mus musculus*) through captures. In four sessions of captures in eight different representative habitats, a total of 197 rats and 247 mice have been captured. In the same year rats impacted negatively the 34% of the 400 shearwater nests monitored, having a similar impact on eggs and chicks. On February 2016 we set 2,700 rodenticide stations all around the island. Then, the rodenticide was replaced in April, June, October and November, with positive results. The rat take of baits has decreased significantly. In November, an average of 86% of baits were left in the stations, indicating a strong decrease of the rat population. Continuing the action and the distribution of rodenticide is essential in order to reach the eradication of this aggressive predator by the end of the year.

Effects of cat, rat, and human predation on Scopoli's shearwater (*Calonectris diomedea*) breeding success and nest-site occupancy on Linosa Island

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Biodiversity on islands is seriously threatened by invasive species, that have been voluntarily or accidentally introduced by humans. Seabirds, especially small and medium ground-nesting Procellariiformes, are particularly vulnerable to introduced predators which can negatively affect breeding success and nest occupancy. Linosa is a small Mediterranean island where thousands of Scopoli's shearwaters (*Calonectris diomedea*) breed each year. Their survival is endangered by the presence of 400 inhabitants, 300 free-roaming cats (*Felis catus*), and a conspicuous population of rats (*Rattus rattus*). Our study aims at evaluating the effects of cat, rat and human predation on the shearwaters' breeding success and the effects of breeding failure on nest-site occupancy. From 2013 to 2016 we monitored shearwater nests and collected data on burrow occupancy, egg deposition, egg hatching, and chick fledging taking notes of cases of failure. Nest characteristics were also measured. Overall, the shearwater breeding success was 65% and predation by mammals was the major cause of breeding failure (19%). We analysed the effects of cat and rat predation and poaching on the nest occupancy in the following year, using generalised linear mixed effect models. We also analysed if nest characteristics (depth and diameter) and nest position, in terms of distance from houses, roads, trails and coastline, were related to the probability of predation by cats, rats and poaching. Egg-poaching had a negative effect on the occupancy of the following year, whereas predation upon eggs by rats and predation upon chicks by cats had a minor effect. We also found that the nest position didn't affect the probability of predation by rats and cats and egg poaching. However, increasing in cavity depth reduces the probability of cat predation.

Invasive plants: what can be done about this continuing threat to biodiversity?

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Human intervention has led to plants being moved around the world for centuries. This practice has been both unintentional and intentional. Unintentional when seeds and/or vegetative propagules were transported vicariously alongside other materials being moved. Intentional when desirable and useful plants were moved around the world, often linked with colonialisation, arguably the fore-runner of today's globalisation. Many of these plants became naturalised only locally or required careful nurturing to survive in their new habitats. However, some of these plant species found their new environments highly conducive to spread. Removed from controlling factors such as pests and herbivores, they became established over significant areas posing a serious threat to native biodiversity. Invasive species are now recognised as a major driver of biodiversity loss globally, with particularly severe impacts on islands. We have reviewed six global invasive species databases to determine the number of invasive plants globally. Taxonomic reconciliation has demonstrated that 6,075 vascular plant species are currently documented as invasive. The first part of this talk will review this in its historical context and consider the implications of the continuing increase in the number and spread of invasive plant species globally. The second part of the talk will review work by Kew's UK Overseas Territories team on invasive plants. The UK Overseas Territories support the most significant UK biodiversity in terms of unique species and habitats. This biodiversity is under severe threat from invasive species. We have been identifying and mapping invasive plants, and developing actions plans for their control. The talk will include examples from St Helena, Ascension, Falkland Islands and British Virgin Islands. Wider implications from this work for dealing with this global threat will be considered.

Partnerships in the restoration of tropical Pacific islands

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The eradication of invasive alien species from islands is a highly effective conservation action for the recovery of declining and threatened native species. Among the characteristics necessary for the success of these operations and the sustainability of the conservation outcomes is a range of technical expertise, cultural and political support, and financial and organisational capacity. In the tropical Pacific, civil society organisations including the BirdLife Partnership have taken a lead in implementing invasive vertebrate eradications, and despite capacity limitations have successfully delivered operations for 40 sites in five countries since 2007. The scale and complexity of these eradications have increased over time, from focusing on single target species on individual islands to simultaneously addressing multiple invasives and islands. This growing experience has highlighted the strengths of locally based civil society organisations, particularly in addressing the cultural and political issues associated with vertebrate eradications, but also the essential role of partnerships in supporting their technical preparation and financing. The operations to date have benefited multiple threatened species. However, if invasive species management is to fulfil it's potential to reduce biodiversity loss on Pacific islands, political support and local capacity must increase, particularly for biosecurity. Stronger partnerships between governments and non-governmental organisations are also necessary, both to engage local communities and to meet the specialised technical preparations and significant financing needs, so that the challenges of island restoration are met with a response of the requisite pace and scale.

Vespapp: citizen science to detect the invasive species *Vespa velutina*

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The yellow-legged hornet (*Vespa velutina*) is an Asian native species recorded across Europe, including mainland Spain, since 2004. Its first detection in Majorca (Balearic Islands; Spain) took place by researchers at the Laboratory of Zoology in collaboration with local beekeepers in October 2015. This invasive species has an important impact on biodiversity, apiculture and human health. Adult wasps are predators of bees, therefore contributing to the loss of honeybee colonies. For efficient actions to minimise the harms of the invasive species, early detections are crucial. Thus, civic collaboration may offer an important source of information to determine the presence and distribution of *V. velutina.* Current technological advances offer the opportunity for citizens to become active participants of the scientific research (citizen science). Vespapp is a software, either as a cell phone app or a website, which aims to identify any suspicious observation (hornets and nests) by sending a picture to a global database. The received information is subsequently confirmed or discarded by an expert panel. In case of a positive identification, an action protocol is implemented including the placement of traps, nest removal and monitoring the area. Since the Vespapp launch in June of 2016, the app has been downloaded 1436 times, has received more than 450 photos and 31 of them have been positive in the Balearic Island and the Iberian Peninsula. These results have enabled detection and removal of a total of nine nests during 2016, which is of great importance in controlling the expansion of the *V. velutina* considering the early stage of invasion in the Balearic Islands.

Wild ginger, a beautiful menace to island ecosystems – can a natural solution be found?

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Kahili or wild ginger, (*Hedychium gardnerianum* (Zingiberaceae)) poses a serious threat to many unique island ecosystems worldwide including: the Federated States of Micronesia, Cook Islands, French Polynesia, Hawaii, New Zealand, La Réunion, the Macaronesian Archipelago and Jamaica. Introduced from the foothills of the Himalayas for its ornamental/commercial value, kahili has escaped cultivation and to become an aggressive coloniser in its introduced, sub-tropical range. Adaptable to a wide range of habitats, from native wetlands and riparian areas through to forest understorey, road verges and scrubland, wild ginger forms large, herbaceous, shade tolerant monocultures which outcompete native vegetation. It has the potential to prevent regeneration of native forests and cause wide scale ecosystem collapse and biodiversity loss. Wild ginger forms deep rhizome beds, reproduces vegetatively as well as through seed and is spreading unchecked across extensive and rugged terrain, which make chemical and mechanical control largely ineffectual. Classical biological control is widely believed to be the only long-term solution for this intractable invader. A biocontrol initiative for kahili ginger was initiated by CABI in 2008 for Hawaiian and New Zealand stakeholders. Surveys in the native range identified a number of damaging and limiting natural enemies which continue to be evaluated for specificity in the UK. The progress, prioritised agents and future prospects are further described.

Is poisoning rodents a health hazard?

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As large-scale island eradication projects expand, it is inevitable that aerial baiting will occur on inhabited islands. However, when anticoagulant bait is to be spread all around living areas, community concerns about the safety of such projects are likely and understandable. Health monitoring of bait handling personnel on the largest island aerial baiting projects (including Macquarie Island and South Georgia), has shown no significant poisoning. Given the exposure of these individuals is orders of magnitude beyond that of community members, such monitoring can provide reassurance to far less exposed individuals. Additionally, lessons can be learnt on how to manage the community perceptions of these issues for critical conservation projects.

When our enemy is our friend: new approaches to managing alien vegetation in Seychelles catchment forest

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Invasive alien plants are one of the major causes of biodiversity loss with impacts on ecosystems such as alterations of biogeochemical and hydrological cycles. The Seychelles' forest is dominated by non-native vegetation arising from plantation agriculture often referred to as novel ecosystems. Under some circumstances native vegetation shows signs of recovery, particularly in low light conditions that occur under a forest canopy dominated by exotic species. Conversely, high light conditions arising from forest disturbance benefit invasive exotic species especially vines such as *Merremia peltata* which outcompete native vegetation. The Ecosystem Based Adaptation in the Seychelles project aims to enhance water-catchment management formulating recommendations for vegetation rehabilitation and establishing post-rehabilitation monitoring. The project will rehabilitate 600 ha of forest, an ambitious target that requires forestry management capacity development, policy development and community support to ensure long-term protection and management of catchment forest. Catchment vegetation quality was assessed using plant endemism, species diversity and forest rejuvenation indices. Sampling was conducted by transects and permanent monitoring plots in 10 intensive monitoring sites in water catchments. The project also deployed drone monitoring and light level monitoring using images taken with a fish eye lens. Rehabilitation has been implemented first on sites with high vegetation quality indices where management is expected to assist natural regeneration. Management has focused on removal of exotic saplings and under-canopy shrubs leaving a forest canopy dominated by exotic species including *Tabebuia pallida* and *Falcataria moluccana* intact. This counterintuitive approach is expected to maintain the shade conditions and the microclimate that will benefit native species over non-native species and facilitate the regeneration of palm dominated native forest. Initial indications are that closed canopy forest rehabilitation and community supported protection of forests from disturbance are important management measures for these novel ecosystems and hence for water catchments.

Eleonora's falcon (*Falco eleonorae*) benefiting from cat eradication – the case of Andros, Greece

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Three colonies of Eleonora's falcon (*Falco eleonorae*) on the island of Andros (Cyclades, Greece) have been monitored since 2006. On two of these colonies, presence of rats has been recorded at least since 2006, while the third colony was invaded by rats in 2011. The latter provided a unique situation to study the short-term impacts of rats on the breeding performance of the Eleonora's falcon. On the newly rat-infested islet within a single year the number of active nests, the breeding success and the total number of fledglings were reduced by 47%, 23% and 58% respectively. Rat eradication operations were successfully carried out on all three islets in 2012 and 2014. At all colonies the breeding performance improved immediately. At the colony where rats were present for only one breeding season (2011), all breeding parameters recovered to pre-invasion levels within the rat eradication year. In all colonies, vegetation degradation resulting from rat foraging had consequently led to lower nesting site quality for falcons. Therefore, rat eradications were followed by construction of artificial nests which further improved the breeding habitat. In the years following the eradications 14–25% of active nests were artificial and the breeding success in artificial nests was in general higher than in natural nests. The rat eradication operations in combination with the construction of artificial nests on the islets of Andros indicate the benefits of these management measures on the breeding performance of the Eleonora's falcon and highlight the importance of immediate response to rat infestation. The conservation measures were implemented as part of the LIFE Nature project ANDROSSPA (LIFE10 NAT/GR/000637).

A review of 12 years of rat eradication operations for the conservation of priority island nesting birds in Greece

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Since 2005, rat eradication operations have been carried out on a total of 41 uninhabited islets and islands in the Aegean Sea in Greece ranging in size from less than 1 ha up to almost 300 ha with the total area exceeding 1,050 ha. The initial eradication methodology was developed with the support of the Royal Society for the Protection of Birds and further optimised through implementation of consecutive eradication operations. The operations were carried out on 16 different groups of islets and islands with the aim of improving the breeding habitat of a significant proportion of island nesting bird species of conservation concern, including Eleonora's falcon (*Falco eleonorae*), Mediterranean shag (*Phalacrocorax aristotelis desmarestii*), Audouin's gull (*Larus audouinii*), yelkouan shearwater (*Puffinus yelkouan*), Scopoli's shearwater (*Calonectris diomedea diomedea*) and European storm-petrel (*Hydrobates pelagicus*). While the most recent eradication operations are still underway, previous operations have successfully removed all rats, eliminating egg and chick predation, as well as, degradation of bird nesting habitats. All rat eradication operations were carried out using brodifacoum-based bait, deployed mainly through placement of bait stations in association with hand broadcast. No significant negative impacts on non-target species due to baiting have been recorded. All rat eradication operations have been carried out through six different LIFE projects co-financed by the European Commission.

Improving nesting habitats for the Eleonora's falcon and seabirds

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Rat invasion is considered a major environmental issue regarding the Aegean islands (Greece), which are characterised by a rich biodiversity of faunistic and floristic taxa of high conservation concern. One of the most emblematic bird species at national level, Eleonora's falcon (*Falco eleonorae*), is severely affected by rat invasion. The Aegean islands constitute the core of its breeding range, holding more than 80% of the species' breeding population. In the framework of the LIFE Nature project "LIFE ElClimA" (LIFE13 NAT/GR/000909), rat eradication operations take place at two uninhabited island complexes, hosting approximately 6% of the species' national population, as well as important colonies of other priority seabird species that are also affected by rat predation, namely the yelkouan shearwater (*Puffinus yelkouan*) and Scopoli's shearwater (*Calonecrtis diomedea*). Removing rats from a total area of 705 ha is the largest rat eradication operation ever attempted in the country. Rodenticide baits have been primarily deployed in bait stations to minimise primary poisoning risk to non-target species, e.g. partridges (*Alectoris chukar*) and rabbits (*Oryctolagus cuniculus*), as well as their predators such as Bonelli's eagles (*Aquila fasciata*) and long-legged buzzards (*Buteo rufinus*), which could be deprived of their food source. After several months of regular baiting, bait consumption is minimal and the eradication operations are considered to be at their final stage. Close cooperation with regional and local stakeholders throughout the field operations aims to ensure optimal involvement of local communities and authorities as well as minimal risk of future rat reinvasion. The rat eradication operations implemented in the framework of the current project are expected to contribute to the preservation of the high ecological value of the two island complexes in general and, in particular, to the improvement of the nesting habitat and conservation status of important bird species in the area.

Broadening the context of invasive species eradications

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Our ability to eradicate harmful organisms has greatly progressed in the last decades, and available information shows that currently this management tool not only is one of the most effective conservation weapons, but also that it permits protection of human livelihood. In conservation we tend to refer to eradications as the total and permanent removal of an invasive species' population by means of a time-limited campaign; this term is more often used for eradications carried out on islands, where some general rules apply, such as that all individuals need to be vulnerable to the removal methods, and that there should be no risk of reinvasions. However, there is a growing number of interventions that go beyond this definition. Eradications can now target multiple species, and campaigns carried out in densely inhabited regions need to address significant risks of reinvasions through long term surveillance and rapid response efforts. Furthermore, there have been eradications carried out at much larger scales than small islands, such as those implemented for human or animal health purposes (e.g. smallpox or the rinderpest virus eradicated from the globe), or of eradications in mainland areas, requiring complex geographical planning, and that may set context specific objectives such as management to zero density in key areas through permanent control efforts. To fully exploit the potential of invasive species control for conservation, it is important to adapt the lessons learnt in islands eradications, rethinking the paradigms of this conservation tool to the new challenges that need to be met, as also highlighted by the Honolulu Challenge on Invasive Alien Species adopted in 2016. The New Zealand Predator Free 2050 campaign, planning to eradicate several key invasives at an unprecedented scale, is indeed a milestone in this direction, providing a basis for broadening the global vision of invasive species management.

Recovery of Santa Luzia Nature Reserve and translocation of the globally endangered Raso lark

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The inhabited island of Santa Luzia is a priority KBA located in Cabo Verde. It holds the entire world population of the Critically Endangered Raso lark (*Alauda razae*) and the most important colony of Cabo Verde shearwater (*Calonectris edwardsii*). Since 2013 SPEA, Biosfera1 and RSPB have developed a feasibility study for habitat recovery of Santa Luzia, including an operational plan for cat (*Felis catus*) eradication and several baseline studies on the local species, both native and alien. The current project aims to translocate part of the population of Raso larks to the nearby island to increase the resilience of the population to long periods of droughts that have been increasing with global climatic changes. The feral cat population, estimated at 126 animals (95% CI 87.5 – 189) individuals, has strong negative impacts on several species of fauna on the island and will have to be removed to increase the chances of success of the re-introduction of Raso larks. Mice (*Mus musculus*) are also present, but at very low densities. Abundance index was calculated throughout the year and peaked at 0.06 and 0.067 captures/trap/night in February and March respectively (mean abundance index throughout the year 0.026). Recent data on the cat diet shows high levels of reptile predation and was found to change markedly depending on annual conditions. In 2010 mice were 79.6% of prey species identified in cat diet, while in 2013 and 2014 cats preyed mostly upon reptiles (91.67% of scats and >70% of prey item biomass). The project will rely strongly on local staff and will involve local communities in order to build local capacity and to increase awareness of the problems caused by IAS on islands. We aim to achieve sustainable protection of the habitats and threatened biodiversity of Raso, Branco and Santa Luzia marine protected areas.

Setting-up a predator-free area on a Macaronesian island using a pest-proof fence

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The island of Corvo, with an area of 17.1 km^2, is the smallest, westernmost and least populated of the nine islands of the Azorean Archipelago; 41% of the island is classified as a Special Area for Conservation and Special Protection Area thus included in the Natura 2000 Network and classified as UNESCO's Biosphere Reserve. Azorean settlers brought a number of associated threats to the local fauna and flora, such as the introduction of invasive mammals (rats (*Rattus* spp.), mice (*Mus musculus*), cats (*Felis catus*), goats (*Capra hircus*) and sheep (*Ovis aries*)), which jeopardize the breeding populations of seabirds. The archipelago still remains of critical importance for the conservation of several petrel species, namely Cory's shearwater (*Calonectris borealis*), little shearwater (*Puffinus lherminieri*) and Madeiran storm-petrel (*Hydrobates castro*). From 2009 to 2012, with funds from the EU LIFE program, a 100% pest proof fence 800 m long was built on Corvo, Azores. This solution was adopted to create a safe nesting area of 3 ha for shearwaters and petrels breeding in the island and subject to high predator pressure from feral cats, dogs (*Canis familiaris*), black-rats (*Rattus rattus*) and mice. Following the closure of the area, all predators were removed and biosecurity procedures were adopted. The vegetation cover inside the fenced area was cleared of alien plants and native flora was abundantly replanted to recover the natural habitats. Acoustic and visual luring methods for prospecting seabirds were employed, and for three consecutive years small groups of juveniles Cory's shearwaters were translocated to the area. The fence withstood hurricane type winds and very frequent harsh weather conditions for long periods of time with minor maintenance necessary. After four years the fence demonstrated to be a feasible solution for adequate areas and the first breeding pairs of seabirds were recorded inside the area during the 2016 breeding season.

Green iguana (*Iguana iguana*) monitoring and control efforts on Grand Cayman

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Effective control management of invasive alien species (IAS) is limited by our understanding of population dynamics. Monitoring and modelling are essential components of control management. The green iguana (*Iguana iguana*) is overabundant on Grand Cayman (estimated density ± SE = 41.363 ± 16.813), and this can cause significant economic losses (e.g., damage to roads and agricultural crops), pose serious health and safety hazards (e.g., diseases and accidents), and trigger negative ecological interactions with endemics (e.g., hybridisation with the Sister Island rock iguana (*Cyclura nubila caymanensis*)). Therefore, control management is a priority for the Cayman Islands Department of the Environment (DOE). In this poster, we provide information about green iguana population surveys conducted on Grand Cayman in August 2014, 2015 and 2016. With the abundance estimates derived from these surveys, we conducted a model-based assessment of population response to sustained removal effort. Although the green iguana is exposed to human-induced mortality (e.g. hunting at private property, depredation by feral cats (*Felis catus*) and dogs (*Canis familiaris*), and road kills), the population increased at an annual rate of 60% between 2014 and 2015 and 98% between 2015 and 2016 (not including hatchlings). Herein, we present the results from experimental culls organised by the DOE in June 2016 in which 18,838 green iguanas were removed mainly from western Grand Cayman. Bounty hunter groups and skilled hunters under contract both averaged about 100 iguanas killed per day. Removal effort, technique used and crippling loss (i.e., shot but not retrieved) were among the variables quantified, which also included biological data to establish a baseline understanding of green iguana population dynamics and response to control management. Applying basic concepts of harvest theory and decision analysis, the DOE and USFWS are developing cost effective strategies going forward.

Predicting the potential habitat of the invasive coral vine (*Antigonon leptopus*) using remote sensing and species distribution modelling

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The spread of invasive plant species often outpaces the capacity to manage the invasions. Remote sensing can be used to map the distribution of invasive plant species at a snapshot in time, but it is difficult to predict the future distribution without incorporating the habitat preferences of the invasive species. Habitat suitability modelling is predictive, but often suffers from an insufficient number of training points. In this study we combine vegetation classification models based on remotely sensed imagery with habitat suitability models to predict the potential distribution of an invasive vine, *Antigonon leptopus* (Polygonaceae), on two neighbouring Caribbean islands, St. Eustatius and Saba. A Support Vector Machines (SVM) classification was produced for two WorldView-2 images of St. Eustatius (images acquired on 8 February 2011 and 24 August 2014) to produce maps of presence/absence of the vine. Pixels from the SVM classifications where *A. leptopus* was present in both years were used as the dependent variable in the species distribution model for St. Eustatius. The independent variables tested for the species distribution model were slope, elevation, soil hardness, soil moisture, drainage area, distance to nearest building, and distance to nearest road. The results suggest that the potential for *A. leptopus* invasion can be readily assessed for other islands in the Lesser Antilles. We illustrate this potential for the neighbouring island of Saba, revealing that the expansion of *A. leptopus* may approach that of St. Eustatius if no preventive actions are taken.

The diet of 'Viking mice' on Nólsoy, Faroe Islands

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Burrowing seabirds can be very vulnerable to rodents. Although there is abundant evidence for the negative impact of rats, there is some recent evidence that mice (*Mus musculus*) can also have a detrimental effect on seabird populations. Introduced by Vikings, mice are the only rodent on Nólsoy in the Faroe Islands, which also hosts one the largest European storm-petrel colonies in the world. Using stomach dissections and stable isotope analysis we examined for evidence of storm petrel consumption (eggs or chicks) in mice on Nólsoy. The findings may have implications for rodent management on Nólsoy and other Ramsar sites in the Faroe Islands.

Seabird restoration and advances towards the eradication of feral cats on Guadalupe Island, Mexico

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Guadalupe Island (24,172 ha; 1,298 m) is located in the Pacific Ocean, 260 km off the Baja California Peninsula. It is inhabited (ca. 150 people) and is part of a Biosphere Reserve, managed by Mexico's National Commission for Natural Protected Areas (CONANP) in collaboration with Grupo de Ecología y Conservación de Islas, A.C, a professionalised Mexican NGO. Guadalupe has 223 vascular plant species (12% endemic), and hosts 139 taxa of birds, including seven endemic races, six of which are considered extinct. Goats (*Capra hircus*), cats (*Felis catus*) and house mice (*Mus musculus*) were introduced by the end of 19th century. Now a goat-free island, the feral cat is the most serious threat to biodiversity, especially to surface- and burrow-nesting birds. The island hosts the most important breeding colony of Laysan albatross (*Phoebastria immutabilis*) in the Eastern Pacific. Upon its colonisation in 1983, albatross adults and chicks have been subject to severe predation by feral cats. To protect the albatross population, since 2003 we have done cat control around the breeding area, now improved by the construction of a 700 m exclusion fence that protects 65 ha. Thanks to these efforts, the number of breeding pairs has increased exponentially, with more than 400 to date. With a long-term vision and the support from the National Fish and Wildlife Foundation (NFWF) and the Alliance WWF-Fundación Carlos Slim, as of March 2017 we have moved from cat control to eradication. Timeframe for the eradication campaign will be 4.5 years. The methods will involve hunting, trapping (leg-hold traps) and detection dogs. Since the island is inhabited, biosecurity measures are crucial since the eradication's start. The achievement of the eradication will benefit native and endemic seabirds and landbirds–especially those endangered–preventing their extinction.

From island studies to mainland management

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Islands have been the first and foremost natural habitats impacted by alien invasive species. With some delay, mainland ecosystems are going through the same effects. Difficulties to manage and mitigate the effects of AI on indigenous species are even greater, and the task to define a strategy more complex, on the continent. The numerous studies and reports have helped taking the challenge up in some territories, and in France, it was decided to plan and organise efforts at the hydrological scale of the great river, the Loire. Since 2002, exchanges between on-field managers and stakeholders have permitted the creation of a network that has emerged as an example, as it edited a first interregional strategy of management of alien species. The flow of information, the common edition of documents and supports for the management were the first on the to-do list. Now, as Europe has announced its first 37-long list of priority species, the Loire working group is revising its third version of a prioritised list and editing a first mapping of more than 60 species. Most of these species came or were helped by the connected water system of the large river and its tributaries. As an interconnected habitat system within the continent, the Loire basin can be compared to an island and as such has a lot to inform the managers from the island alien invasives techniques used to eradicate the species. The creation of an atlas helps visualise which species should be targeted for such efforts, and where to start. So, the achievement of more than 15 years is only the start.

Differential effects of human impact and habitat type on exotic and native species diversity on oceanic islands

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Human land use is considered one of the main drivers of species compositional change. While some species experience population decline as a result of human activities, others optimally exploit human-impacted environments. We hypothesised that such contrasting responses could in part be attributable to species' native or exotic origin. Our objective was to assess the effect of human impact, defined as the addition of man-made substrates, on the taxonomic and functional composition of exotic and native reptile assemblages of two anthropogenically impacted Caribbean islands. We extensively surveyed insular reptile communities and recorded species abundance and richness data. Functional traits were obtained from literature and used to construct functional diversity metrics for every sampled community. Of the composite environmental variation among 114 sample plots, 46% could be reduced onto two PCA axes, resulting in a habitat structure axis (29%) as well as a human impact axis (17%). PCA axes were subsequently regressed against various taxonomic and functional abundance and diversity indices. Habitat structure and human impact independently affect abundance and diversity indices across both islands. The direction of these effects largely depends on exotic or native origin. Exotic species are never found in forest habitat, whereas native abundances peak in tropical forest. Exotic abundances are primarily affected by human impact levels while native abundances show no significant association. Exotic species occur in higher numbers on St Martin, which is likely due to regional shipping intensity rather than within-island factors. Furthermore, on St Martin, exotic species significantly increase functional trait diversity by occupying unique functional niche space in impacted environments. However, we found no indication of environmental filtering of functional trait values as a result of human impact, rather habitat structural change seems to shift community trait values towards beneficial levels for survival in non-forested environments.

Genetic pest management technologies to control invasive rodents

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Many strategies exist to manage invasive pests on islands, ranging from poison to trapping, with varying degrees of success. Genetic technologies are increasingly being applied to insect pests, but so far, not to vertebrates. We are implementing a genetic strategy to eradicate invasive mouse populations as another tool for pest control. *Mus musculus*, the common house mouse, is one of the most widespread invasive species. Mice threaten human health, agriculture, and biodiversity on many islands, particularly seabirds. Seabirds are endangered indirectly through competition for resources or predators being attracted by the mice or directly with mice attacking chicks and eggs. Rodenticides are the most common method of eradicating mice, but their use leads to poisoning of non-target species and has limited efficacy against mice. An approach that could eliminate non-target species impact would be to engineer daughterless mice linked to a gene drive system for self-sustained propagation. For this project, we have investigated exploiting a naturally occurring gene drive, the t-complex. Using the t^{w2} haplotype of the t-complex, we observed the t^{w2} haplotype being transmitted to offspring with a transmission distortion ratio of 95.3%. The daughterless phenotype is being accomplished by inserting the *Sry* gene (male sex-determining gene) into an autosome containing the t^{w2} haplotype via CRISPR/Cas9 gene editing. The presence of *Sry* will induce testis formation, regardless of the sex chromosomes naturally inherited. When *Sry* is inserted into the t-complex, the desired gene will spread through the population, eliminating female offspring. This model system will support studies to evaluate the effectiveness of crashing an invasive population without adversely affecting other species. While still in the beginning stages, this is a novel idea and once this method has been perfected, it will open the way to use this genetic strategy for the eradication of other invasive mammal species.

A new look at Galapagos fouling communities

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The maritime history of the Galapagos Islands begins in 1535 with the accidental discovery of the archipelago. For the past 500 years the islands have endured a significant amount of terrestrial plant and animal introductions and, to some extent, freshwater invasions; however, the number of marine introductions reported has been significantly lower. Research has been conducted looking at the fouling communities of the Galapagos Marine Reserve (GMR) to provide a clearer picture of the true scale of marine non-native species present in ports and harbours of the GMR. Settlement plates were deployed for three and 14 months on floating docks on the Islands of Santa Cruz and San Cristobal. As a result, numerous new records of introduced species of hydroids, polychaete worms, bryozoans, and ascidians, amongst other taxa, have been documented for the Galapagos. The continued increase of marine traffic from many sources to the Galapagos Islands concomitantly increases the risk of arrival of non-native species to this region. While research on terrestrial invasive species is well established, research on marine invasive species and their impacts in the GMR has been less investigated. The Charles Darwin Foundation (CDF), the Galapagos National Park Directorate (GNPD) and the Galapagos Biosecurity Agency (ABG) have been working together to improve the marine biosecurity standards for the GMR, and some clear advances are now in place. A synthesis of marine biosecurity based on prevention, early detection and management of marine non-native species is presented and potential management strategies discussed.

Planning processes for eradication of mice on Gough Island

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Gough Island, part of the remote Tristan da Cunha group in the South Atlantic, is considered one of the most important seabird islands on the planet. A UK Overseas Territory and World Heritage Site, Gough supports millions of breeding seabirds and the UK's only Critically Endangered bird species, the Tristan albatross and Gough bunting. Invasive house mice (*Mus musculus*) were introduced in the 1800s and prey on hundreds of thousands of chicks each year. It has been predicted that the Tristan albatross faces extinction within c. 30 years unless the mice are eradicated. Led by the RSPB and the Government of Tristan da Cunha, the Gough Island Restoration Programme aims to eradicate mice from Gough Island using aerial baiting containing anticoagulant toxin; a methodology established during previous island eradications. Now in the operational planning phase, the programme aims for mouse eradication on Gough in 2019. Applications for various approvals are required and a captive bird management programme designed to protect land birds vulnerable to secondary poisoning. As well, robust operational planning and detailed logistical planning need to be completed. Situated around 2,800 km from Cape Town, South Africa, Gough Island presents challenges including its remoteness, terrain, weather and cave systems. Long lead in times for planning are required, reflecting the scale and complexity of logistics and regulatory requirements.

Perils of saving the smallest for the last: lessons learnt about sequencing eradications on Santa Cruz Island, CA

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The biota of many islands have been damaged by invasive species, but in a growing number of cases island invaders have been successfully eradicated. Many eradication projects target vertebrate species whose size and harmful effects make them particularly conspicuous. Unfortunately, smaller and less conspicuous invaders, including invertebrates and plants, may be overlooked before or following successful eradications, and their continued presence can limit the attainment of some of the management goals that may have motivated the earlier eradications. For example, vegetation recovery that often follows removal of herbivores can make eradication of remaining invaders more difficult. Vertebrate eradications can result in the release of perceived "secondary" invaders, which can compromise the benefits of the initial eradication. We review the suite of eradications that have occurred or are underway on Santa Cruz Island, USA, which have focused on plant, invertebrate, avian, and mammalian taxa. We discuss the biological impacts of – including the long-term management challenges created by – decisions regarding which taxa were eradicated when. We recommend that prior to undertaking any eradication all invasive species and the resources they threaten be evaluated with regard to how the sequence of eradications may positively or negatively affect any eradication efforts that may follow.

Citizens' attitude towards the removal of grey squirrels in Italy: what support do we need?

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Grey squirrels (*Sciurus carolinensis*) were introduced in Umbria, Central Italy, in 2000. Since then, they have successfully occupied a range of about 50 km^2. The LIFE U-SAVEREDS Project now aims to eradicate this isolated population, but the squirrel distribution is centred on the city of Perugia and animals are particularly abundant in public urban parks and private house gardens. Thus, part of the public opinion opposes the project activities. For this reason, the overall management strategy involves both direct (capture and euthanasia) and indirect (capture and surgical sterilisation) removal of the animals. Further, a Decision Support System including the evaluation of social issues was specifically developed. It identified spatial intervention priorities and it allowed the start-up of grey squirrel management in areas where the overall social context was favourable. At the same time, we implemented a targeted information campaign to increase the population's knowledge on the issue of invasive alien species and, most important, to actively involve citizens in the Project. As a consequence, several citizens agreed to collaborate on the eradication campaign. Following the intervention in different management units, characterised by a different acceptance level of the eradication campaign, we now evaluate how the citizens' collaboration affected the outcome of Project activities. The percentage of accessible land (ranging from 84 to 21%) for each management unit was quantified through mapping and modelling in GIS environment, and was compared to the outcome of direct removal of the animals. In 2016, 470 animals were removed, and preliminary results suggest that the spatial configuration of accessible lands also plays an important role in the eradication. Considering both social and technical issues, simulations were finally implemented to assess the success probability of the eradication campaign at local scales.

Computer modelling of complex interstitial spaces to protect endemic island lizards from invasive mice

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New Zealand is home to a large diversity of endemic lizards, with 42 gecko (Diplodactylidae) and 55 skink (Scincidae) taxa, ~ 84% of which are classified as Threatened or At Risk. Habitat destruction and invasive mammalian predators are responsible for much of this decline. Endemic lizard species are afforded legal protection in New Zealand, meaning that when populations are threatened by human activity such as road construction, individual animals must be salvaged and moved to a safe location (mitigation translocation). Mitigation translocations of lizards in New Zealand often involve habitat enhancement, for instance building new rock pile habitat. However, there is little research to show if habitat enhancement actually has the intended effect of providing better habitat for lizards, or if there might be undesirable side effects such as creating habitat for invasive predators like mice (*Mus musculus*). I describe a novel technique using a computer game physics engine (Unity, PhysX) to investigate the best rock pile design to protect translocated skinks while hindering the movement of mice. I achieve this by measuring the interstitial spaces in virtual rock piles to determine which compositions (sizes, shapes of constituent rocks) will maximise spaces skinks are able to fit through while minimising spaces mice are able to fit through, enabling skinks to avoid predation by mice. My virtual approach to this problem allows me to model complex spaces which were unable to be measured using previous, physical techniques. Predictions from modelling are confirmed using data from computed tomography (CT) scans of real rock piles. The design that results from this research will be tested in a real mitigation translocation to determine whether skinks have higher survival in my rock pile designs. This research will inform understanding of invasive predator/prey interactions and conservation of species threatened by invasive mammals.

An integrated physical control method on *Spartina alterniflora*

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Spartina alterniflora is a noxious invasive plant due to its ecological impact. An integrated method of mowing plus shading (MS) was conducted for control of *Spartina alterniflora* in Guangxi, China. Plant height, node number, node length, basal stem diameter, aboveground biomass and population density of this weed were used to compare the effectiveness of mowing and MS. Results showed that all characters of *S. alterniflora* were significantly decreased by mowing plus shading ($P<0.\ 05$), and only node number, plant height and aboveground biomass were suppressed by mowing alone. It was indicated that clonal growth and sexual reproduction of *S. alterniflora* were absolutely inhibited by mowing plus shading in the whole growth season. We also found the restraining effect of mowing plus shading was positively correlated with shading degree. The light transmittances of single layer shading net, double layers shading net and triple layers shading net were 15.27%, 2.29% and 0.31%, respectively, and rhizome survival rate were 3.68%, 2.09% and 1.70% in November respectively. Above-ground parts were all dead in November before mowing plus single layer shading treatment, while they were all dead at July in mowing plus double layers shading treatment and mowing plus triple layers shading treatment. In the future, mowing plus shading may be used as an effective method of controlling *S. alterniflora*.

Predicting the risk of plant invasion on islands: the case of *Miconia calvescens* in the Marquesas, French Polynesia (South Pacific)

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Miconia calvescens (Melastomataceae), a small tree native to Central and South American rainforests, is a dominant plant invader in the Society Islands (French Polynesia), Hawaii, New Caledonia, and tropical Australia, thus listed as one of the world's 100 worst invasive species. This fast growing, early reproducing and prolific seed producer (small fleshy fruits dispersed by birds over long distances) with a long-lasting soil seed bank (several decades) was first detected 20 years ago in the Marquesas (French Polynesia), a remote archipelago with a unique and endangered native flora (48% of endemism and 145 threatened species). Despite some eradication efforts, several new outbreaks have been located in the last few years on the largest island of Nuku Hiva. In this alarming context, it is urgently needed to determine the potential distribution of the species in order to assess the risk of invasion and refine the areas for further surveys and control. Species distribution models (SDMs) are numerical tools that project species distribution from the combination of species occurrences with environmental variables. Fitting an SDM on the basis of Marquesas populations to predict the future of *Miconia* over the archipelago would violate the equilibrium assumption behind SDMs. Moreover, projecting the environmental envelope occupied by the species in its native range would ignore inherent characteristics of island ecosystems (e.g. low species richness, low functional redundancy, competitive release, vacant niches, restricted and specialised habitats) that leave them much more vulnerable than continents to biological invasions. As a result, the environmental distribution of *Miconia* across the similarly-sized high-elevation islands of the Society and the Hawaiian archipelagos was projected over the Marquesas. The different SDMs agree that *Miconia* will spread over a large area of native lowland rainforest and montane cloud forest in Nuku Hiva unless appropriate control strategies are rapidly adopted.

The secret life in Switzerland of an island pest, the house mouse

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House mice (*Mus musculus*) can have harmful effects on island biota, and are frequently the targets of eradication efforts. The success of eradication strategies will be influenced by how well the biology of the house mouse is understood. We have carried out a long-term study of a free-living population of wild house mice in Switzerland, following mice in the population from cradle to grave (or disappearance). Adult mice are chipped and a system of antennas installed at the entrances to nests have allowed us to monitor the movements of house mice and observe their social lives in unprecedented detail. House mice live in large but fairly closed social groups of males and females, sharing several nests. Competition between males and between females has led to dramatic reproductive skews in both sexes and high rates of infanticide, despite *ad libitum* food availability. Multiple paternity within litters is common. Cooperation between breeding females within a social group also occurs, in communal nursing of all pups present in the same nest. Population density has increased over time, giving rise to larger group sizes. How this increase in social tolerance is achieved is unclear. Furthermore, population size recovered rapidly from an epidemic that killed ca 30% of adults. We are currently focused on understanding factors influencing reproductive suppression, dispersal likelihood, social tolerance and cooperation between females, including genetic influences, such as the t haplotype. Our studies may be useful in predicting the outcome of interventions to house mouse populations.

Catalysing conservation of islands through collaboration: a North American perspective

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The countries of North America are inextricably linked through shared species, habitats, and ecosystems. Over the last several decades, significant efforts have been made to protect and restore unique island ecosystems within the three nations. Many of the significant advances have been through bi and trilateral collaboration. In recognition of the value of cross border collaboration, in 2014, the governments of Canada, United States and Mexico signed an agreement to protect fragile island ecosystems and their imperilled species. This agreement, endorsed under the scope of the Trilateral Committee for Wildlife and Ecosystem Conservation and Management, strengthens the on-going collaboration between the three nations on the conservation and restoration of island ecosystems and their adjacent coastal and marine environments. Through coordinated efforts, government and NGO partners are accelerating investment in island conservation programmes across North America with a focus on invasive species, biosecurity, restoration, and regulatory processes. Activities include prioritisation of invasive species on a continental scale, sharing of expertise and technology, strengthening institutional capacities, and leveraging (shared) funding and support. These partnerships have accelerated conservation outcomes across North America, including the eradication of invasive species in Canada, protection of rare species and ecosystems in the United States, and a systematic and comprehensive programme to conserve and restore islands in Mexico.

The value of monitoring and the price of uncertainty in the management of an invasive population

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Improving decision-making regarding resource allocation for the control of invasive populations often requires monitoring to obtain information on the state of the population. The cost incurred by monitoring detracts from the resources available for direct control, and so, for monitoring to be feasible, the information gained must have greater value to management than the costs of obtaining it. We aim to provide generalisable recommendations on the use of monitoring data to inform the management of invasive species. Here we present a simulation study inspired by the control of invasive American mink in Scotland. Mink populations exhibit seasonal dynamics with highly dispersive juvenile and intrasexually territorial adult life stages. Control effort was simulated to be dependent on season and perceived variation in the abundance of settled adults. Imperfect monitoring can result in false positive or negative detections of adults, allowing the value of reducing uncertainty by increasing monitoring effort to be explicitly considered in terms of its impact on the invasive population and unplanned overspending of effort budgets. The modelling framework allows the relative value of monitoring effort to be assessed for different control strategies. Future work will utilise large-scale mink control data and surveys of a threatened endemic prey species, the water vole, to estimate the level of mink control required for a high probability of persistence of water vole metapopulations. This will inform future simulation work identifying the balances between monitoring and intervention that maximise the probability of favourable conservation outcomes for fixed cost.

Invasive arthropods of ecological, agricultural and health importance recently introduced in the Balearic Islands (Spain)

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The Balearic Islands archipelago (Mallorca, Menorca, Ibiza and Formentera) is located in the western part of the Mediterranean Sea. Like other places in the area, the Balearic Islands are exposed to the introduction of several highly invasive species, some of them even world-wide distributed. In fact, the Balearic Islands have a long record of introduced species including different taxa of animals. Here we focus on those invasive arthropod species that were introduced during the last decade and have high impact on ecosystems, agriculture and human health. We present a description of the current situation of the incursion, spread and impact of the tomato leafminer (*Tuta absoluta*, Gelechiidae); the red palm weevil (*Rhynchophorus ferrugineus*, Curculionidae); the Asian tiger mosquito (*Aedes albopictus*, Culicidae) and the Asian hornet (*Vespa velutina*, Vespidae). We conducted an analysis of the path of entry of the different species to the Balearic Islands, considering means of transport including commodities and human transportation. We also analysed the current impact of the presence of the above-mentioned species on agriculture (i.e. increase use of insecticides), landscape (i.e. palm trees destruction), human health (i.e. vector-borne diseases) and ecosystems (i.e. impact on bee population). Results indicate that some invasive species, such as *T. absoluta* could be effectively managed by farmers after a period of adaptation of control procedures to the new pest. The impact on landscape by species such as the red palm weevil has notably increased since its introduction and its expansion is currently uncontrolled. Species such as the Asian tiger mosquito have changed the perception of citizens on the risk of vector-borne diseases, due to the current expansion and its possible implication on arbovirus transmission. Finally, the recent detection of the Asian hornet, has deeply increased concern about the role of bees as an essential component of ecosystems.

Using key-informant surveys to reliably and rapidly estimate the distributions of multiple insular invasive species

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Knowledge of invasive species' distributions is critical to manage established populations. Distribution at large spatial scales can be rapidly estimated through public surveys, though reliability of such information must be tested. We gathered detection/non-detection data for the Indian bullfrog (*Hoplobatrachus tigerinus*), the common myna (*Acridotheres tristis*), the house sparrow (*Passer domesticus*), and the giant African snail (*Achatina fulica*) through interviews in 91 sites on inhabited islands of the Andaman Archipelago. We interviewed 855 key informants comprising farmers, plantation workers, and aquaculturists, from January to March and September to December 2015. Additionally, we obtained detection/non-detection data for the Indian bullfrog (75 sites), the common myna (65 sites), the house sparrow (39 sites), and the giant African snail (29 sites) through systematic visual encounter surveys and opportunistic records. We corrected the informant data for false positive detections in an occupancy framework and estimated the distribution of the four species. The Indian bullfrog occurred on all islands, except Baratang, Long, and Little Andaman Islands. The giant African snail was ubiquitous, occurring on all islands. The distribution of the common myna was most likely influenced by roads, while ports might be significant for the house sparrow invasion. The findings substantiate the efficacy of public surveys in generating rapid distribution information on multiple invasive species simultaneously.

Time germination response to temperature and light conditions in *Ulex*

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The Canary Islands are widely recognised as an outstanding biodiversity hotspot worldwide. Biological invasion, together with wildfire, are two of the main factors of biodiversity loss in the islands, due to low habitat diversity, their simplified trophic webs and the high rates of endemism. *Ulex europaeus* is an invasive species, which is in the early period of its naturalisation, but it is already affecting two of the richest ecosystems of the island: laurel and pine forests. Previous studies were focused on shade and post-fire conditions as key factors in the growth of young plants, while less attention was oriented to factors linked to seeds germination. The goals of this study are to understand the role of light exposure and temperature shocks in *U. europaeus* germination, and to highlight the optimal conditions. In this study, seeds experienced three different light exposures (total darkness, 70% shade and full light) with eight different temperature ranges (from 30º to 130º C). Then seeds were exposed to temperature shocks for 1, 5 and 10 minutes. The results of DCA and standard statistical analysis show that light exposure has a low relationship with seed germination. Significant differences were found between temperature and time germination: a short exposure to temperatures between 40º to 70 ºC has a positive effect on the germination of *U. europaeus*, although higher temperatures inhibit germination. These results enable a greater understanding of the relationships of *U. europaeus* and environmental conditions of fire zones, but further studies that take into consideration the role of litter and ashes are needed also.

Rat eradication from Berlengas Island, Portugal

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The Berlengas Archipelago, six miles off the coast of Portugal, is composed of a main island of 95 ha and five small islets. It holds the only colony of Cory's shearwaters (*Calonectris borealis*) on continental Portugal, and the largest Portuguese colonies of shag (*Phalacrocorax aristotelis*) and yellow-legged gull (*Larus michaellis*). A breeding population of Madeiran storm-petrel (*Hydrobates castro*) of unknown size also breeds on the nearby islets. The native vegetation includes three endemic species of conservation concern. The presence of IAS in Berlengas (black rat, *Rattus rattus*) is considered to have a significant impact on several seabird species and on the island vegetation. It is also thought to prevent colonisation of the main island by prospecting Madeiran storm-petrels that are often recorded there. Within the scope of an EU funded LIFE programme, a full rat eradication started in 2014, and is still underway, to restore the local ecosystem. A grid of 1,000 closed baiting stations (25 m x 25 m) was used with cereal pellets containing the anticoagulant brodifacoum. Special care was taken to prevent secondary poisoning of non-target species, and a full assessment of the invasive alien species populations was made before any control action started. Species abundance, local distribution, inter-annual abundance variation, and genetic characterisation was determined prior to the baiting operations that started on September 2016. The last confirmed rat sign was registered at the end of October during the weekly monitoring surveys. After December 2016, the remaining toxic baits were removed from the baiting stations and non-toxic scented baits were used to detect any remaining signs of rat activity. The operational phase is expected to last at least two years after the first baiting station was set and we expect that after the eradication the subsequent recovery by seabirds and native plants will make a substantial conservation contribution at European level.

Response of an open feral cat population to an intensive control programme for improving the critically endangered Fatu Hiva monarch conservation strategy

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The Fatu Hiva monarch (*Pomarea whitneyi*) is an endemic terrestrial bird of Fatu Hiva Island (Marquesas, French Polynesia) red-listed by IUCN as Critically Endangered since 2000. Recent decline of the remaining populations is particularly alarming with 30 individuals currently remaining while 275 were still present 10 years ago. Introduced predators have been identified as the main cause of extirpation, especially ship rat (*Rattus rattus*), introduced in the 1980s and feral cats (*Felis catus*) that reatly impact the remaining population at all bird demographic stages (chicks at nest, fledging chicks, and adults). An intensive feral cat culling programme has therefore been progressively implemented over the past five years by SOP-Manu (Birdlife representative in FP) on a 290 ha controlled area to secure part of the Fatu Hiva monarch population. By using data from 43,845 trap-nights and > 189,000 camera-trap images we evaluated the effects of this intensive cat control on feral cat abundance in the treated area (three different indices: abundance index, minimum number of individuals and individual capture histories using the spatially explicit capture–recapture (SECR) model to calculate densities). In parallel, we fitted cats with GPS collars to (i) understand the recolonisation process from the untreated adjacent areas and (ii) assess the risk due to domestic and stray cats from the nearby village. These results will help to refine and optimise feral cat control strategy in this large, mountainous and inhabited island where eradication could be considered, although difficult. The protected and treated area includes 25 of the 30 remaining individuals whose only three breeding pairs of this species are on the verge of extinction.

Feral cats threaten the outstanding endemic fauna of the New Caledonia biodiversity hotspot: implications for feral cat management strategy

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Among invasive species, feral cats (*Felis catus*) are one of the most successful and harmful predator species leading to dramatic loss of biodiversity on the world's islands. Effective feral cat management (eradications, controls) on numerous islands generally resulted in positive effects for native biodiversity conservation. The lack of feral cat diet study in the New Caledonia archipelago was an obstacle (i) to assess the importance of feral cat issues and (ii) to provide relevant guidelines for feral cat population management to mitigate their impacts. Our study aims to evaluate feral cat threats to the outstanding biodiversity at this major biodiversity hotspot in order to provide recommendations to prioritise management and preservation of native biodiversity. We investigated feral cat predation by analysing 5,300 cat scats sampled at 14 selected representative sites giving an accurate picture of the four main natural habitats. Feral cats prey upon at least 43 vertebrate species, 20 of which are IUCN Red List threatened species. New Caledonia is the home of 30.8% of IUCN threatened species preyed on by feral cats, while representing only 0.12% of the total area of islands (including Australia). Thus, this study increases at least by 44.4% the number of IUCN threatened species vulnerable and preyed upon by feral cats across islands worldwide. Threatened vertebrate species preyed on by feral cats are skinks, flying foxes and petrels, and their predation mainly occur in humid forest and maquis mosaic sites. The results of this study prompted feral cats to be listed among the top-five priority species for future management in New Caledonia. We therefore recommend that future actions be prioritised based upon the most critical species situations (most impacted and endangered native species, i.e. skinks, flying foxes, seabirds), and targeting first some geographic areas of manageable size already offering some management facilities and support.

Scaling up invasive plant management for ecosystem restoration in Mauritius: successes and challenges

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Objectives: To document progress made in the last 30 years in restoration of Mauritian terrestrial ecosystems with a primary focus on the invasive plant management component. Methodology: Invasive plant management activities and results have not been systematically monitored so much of the evidence for management effectiveness is anecdotal. As part of the UNDP-GEF PAN Project (Expanding coverage and strengthening management effectiveness of the protected area network on the island of Mauritius) practitioners' knowledge of plant restoration practices undertaken to date has been synthesised in a 'Good Practice Guide for Native Forest Restoration in Mauritius'. This synthesis has allowed us to take stock of management effectiveness. Results: The area under restoration in mainland Mauritius has increased from < 10 ha in the 1980s to almost 100 ha from the 1990s to the 2000s to nearly 500 ha today. Per hectare weeding costs in real terms have been reduced by more than half during this period, principally by moving away from pure manual weeding to an approach that involves a mixture of manual and chemical approaches, and more effective implementation arrangements. There are certain common practices in invasive plant management but there are also site and species-specific weeding approaches, and initiatives that could be scaled up such as utilising weed biomass as a cost-recovery option, and using mulching as a weed suppression technique. Conclusions: Much progress has been made at both the site level and nationally for the country's entire PA estate. The Good Practice Guide will help disseminate this knowledge among new and existing practitioners as a contribution to management effectiveness.

Implementing an early detection programme on Catalina Island: prioritising landscaped grasses

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Invasive species pose a significant threat to native plant species by increasing the risk of wildland fires, displacing native species, and altering native habitat. Recent trends in Southern California landscaping have increased the demand for drought resistant grasses, and often these are non-native species. Catalina Island Conservancy's Catalina Habitat Improvement and Restoration Program's invasive plant project developed an early detection and rapid response project, the Avalon Grasses Initiative, in 2016 to address recent introductions of three highly invasive grass species installed in landscaping. The Avalon Grasses Initiative implements "target-based" early detection methodology created by previous research and early detection efforts conducted on mainland California. Roadside surveys detect populations and staff walks through the community going door to door to request permission to remove target species and offer native plants as replacement. Initial surveys detected 30 populations of *Cortaderia selloana*, *Pennisetum setaceum*, and *Stipa tenuissima*. Control and survey efforts are on-going, but more than 1,000 plants have already been removed and replaced with native Catalina Island plant species grown in the Conservancy's native plant nursery.

Challenges and opportunities for lethal and non-lethal management of non-native ungulates on islands: feral pigs, goats and cows

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The National Wildlife Management Centre (NWMC), which is part of the UK Government's Animal and Plant Health Agency (APHA), has supported and delivered the management of non-native species including commensal rodents and ungulates on a variety of islands across the world. The NWMC utilises a range of both lethal and non-lethal approaches in these projects. We will present two ungulate case studies highlighting the merits and limitations of each of these approaches. This includes NWMC's recent work to reduce the population of feral goats on Great and Little Tobago in the British Virgin Islands. NWMC worked with the RSPB and the National Parks Trust of the Virgin Islands to directly reduce this population through humane culling and trained locally-based staff to increase their capacity to deliver similar projects in the future. Although this project proceeded as intended, lethal control is not suitable in all situations. We have found that although it can deliver rapid reductions in populations in the short term, and is often the best option where complete eradication is the aim of the management intervention, it may be unfeasible or be unacceptable due to its impact on the environment and on animal welfare. Fertility control is increasingly being considered as an alternative long-term solution to reduce population sizes of problematic species. This non-lethal method can offer a humane, publicly acceptable method to reduce population sizes. Recent advances in research and development have led to the registration of novel fertility control agents for wildlife. Species-specific systems to deliver baits containing oral contraceptives to target species are now available. In addition, the development of new software and mathematical models has allowed researchers to make predictions of the effects of fertility control on population size. In our second case study, we present experimental data on the efficacy of fertility control agents on model wildlife species and illustrate examples of species-specific bait delivery systems.

Diet of introduced black rats (*Rattus rattus*) on Christmas Island: setting the scene with stomach and stable isotope analysis

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The black rat (*Rattus rattus*) is an introduced and invasive rodent, negatively affecting endemic species on many islands worldwide. Black rats have existed on Christmas Island for more than 100 years, and feral cats (*Felis catus*), also on the island, are poised for imminent eradication. The risk of meso-predator release needs to be considered, and a combination of stomach and stable isotope analyses of rats was used to determine potential impacts on native fauna should such a release occur. Samples of rat stomach, muscle and fur, along with baseline and consumer reference groups were collected in plateau forest and coastal terrace for stable isotope analysis during the wet and dry season of 2015/16. Stomach analysis revealed an omnivorous diet, with reproductive parts (flowers, fruits and small seeds) of plants significantly dominating the invertebrate component. One reptile was found in a single gut, the introduced blind snake (*Indothyphlops bramini*) but no birds were detected in stomach contents. Stable isotope analysis showed an omnivorous to predatory role compared with stomach analysis, but no association with nesting seabird sources. The effect of habitat and season did not result in major diet shifts, with rats consuming items that primarily followed the C3 pathway. Omnivory was predominant in plateau forest and carnivory dominated the coastal terraces, while trophic niche width broadened on the coastal terraces. Homogeneity of diet across habitat and season suggests persistent plant and invertebrate resources may satisfy nutritional requirements through opportunity or necessity year-round. Little evidence of significant dietary overlap was shown with feral cats based on stomach data from previous diet studies. Further investigation into the diets and relative abundance of rats over time is required to reliably gauge their impacts on vulnerable species and communities on Christmas Island, to justify future rat control actions in the wake of feral cat eradication.

The prospects for biological control of *Rubus niveus* in the Galapagos Islands

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Following its introduction for its sweet edible fruit in the 1970s, *Rubus niveus,* native to Indochina, has become one of the worst invasive weeds on the Galapagos archipelago. It invades open vegetation, scrub and forests where it can grow to 4 m in height and form dense, impenetrable thickets. As a result, *R. niveus* can out-compete native flora and decrease biodiversity; the endemic *Scalesia pedunculata* forest on Santa Cruz Island is currently threatened by *R. niveus*. It is also a serious problem for agricultural land where it increases the cost of weed control and may render land unsuitable for cultivation. Current control methods are based on mechanical removal followed by chemical control. However, due to the long-lived seed bank and rapid growth of *R. niveus*, this has to be repeated, which is both labour intensive and costly. Classical biological control using coevolved, host-specific natural enemies from the native range of an invasive species can be an economic and self-sustaining method of weed control. It is important to select natural enemies for further evaluation that are best-adapted to populations of *R. niveus* on the Galapagos Islands. The results of on-going molecular research undertaken to determine which area in the native range the archipelago biotype originated from, will be presented. In addition, the results of a desk-based analysis and preliminary natural enemy surveys in India and China, which have revealed a suite of insects and fungal pathogens that target *R. niveus*, will be discussed.

A tool for biodiversity conservation within Chile: renewed interest in island eradications sparked by successful European rabbit (*Oryctolagus cuniculus*) eradication

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Choros Island (301 ha) and Chañaral Island (517 ha) are the two largest islands which make up the Humboldt Penguin National Reserve (RNPH), in northern Chile, within Chile's National Protected Areas System (SNASPE) designed to protect the rich resources of the Humboldt Current. The European rabbit (*Oryctolagus cuniculus*) was introduced to both islands in the mid-20th Century, triggering erosion and negative impacts on native vegetation and two seabird species endemic to the Humboldt Current: the Humboldt penguin (*Spheniscus humboldti*) and the Peruvian diving petrel (*Pelecanoides garnotii*). Island Conservation and CONAF (Corporación Nacional Forestal; Chile's National Forestry Corporation and RNPH manager) initiated the eradication of European rabbits from Choros Island in 2013 – the first eradication of invasive species from a Chilean island in a decade. The project was successfully confirmed in 2014, prompting the partnership to pursue ecological restoration of Chanaral Island in 2015, beginning with the removal of invasive rabbits. Utilising lessons learnt from work on Choros Island, the eradication on Chañaral Island was initiated in 2016 and is currently in a monitoring phase. The opportunity to remove all invasive vertebrates from the entirety of a protected area – RNPH – has built confidence in planning, implementation and monitoring among government officials and local stakeholders (ecotourism operators) and has facilitated increased momentum in Chile for island biodiversity conservation through the eradication of invasive species. As a result, CONAF seeks to achieve greater biodiversity conservation within other islands in the SNASPE, such as the Juan Fernández Archipelago National Park (PNAJF), representing unique ecosystems severely affected by multiple invasive vertebrates.

Finders keepers? Discovering and securing the rare species rediscovered in weeded restoration plots

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Objectives: To document the role of the expansion of weeded areas in increasing the discovery of threatened plants and how this process can be integrated into systematic plant species recovery programmes. Methodology: Mauritius hosts some of the most threatened plant species in the world. More than 80% of its remaining 273 endemic plant species are considered to be threatened. The expansion in weeded areas in recent years has resulted in a number of species rediscoveries and increases in the known wild populations for other species. Written and verbal records of species rediscovery from different agencies are consolidated. Results: Results are summarised by numbers of species and number of individuals rediscovered, location of these discoveries and the fate of the discovered individuals. Some rediscovered individuals have been successfully utilised for their germplasm for propagation and subsequent reintroduction. However, in most cases, this process has not been systematic. Conclusions: Finding previously unrecorded species and populations is clearly a positive thing. However, there are challenges. Weeding, although essential for the long-term health of Mauritian native forest, can cause short-term negative effects for rare plants and other threatened taxa. Therefore, it is important to develop weeding approaches that take the requirements of rare plants into account, for example leaving certain exotic species which act as substrates to epiphytic plants, and gradually removing species in the vicinity of rare plants so that they are not exposed to a sudden change in micro-climate. These actions have been implemented in certain instances but have been neglected in others, chiefly because of the lack of knowledge of labourers who are not trained to recognise rare native plants. Rediscovery does not mean that the species concerned are 'out of the woods' so it must be considered to be a part of an overall rare species recovery plan.

Impacts and control of invasive species: trading off actions

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Environmental conservation is chronically underfunded, as a result of both an absolute shortfall in funding, and poor funding prioritisation. Control of impacts of invasive species on native ecosystems is recognised as of high global conservation priority, but also requires significant economic investment. Improving prioritisation of invasive species control options, and identifying alternative funding sources, would therefore greatly improve efficiency in mitigating degradation caused by invasive species. We incorporate ecological, economic, and social considerations to prioritise options for control of invasive grazing species on Bonaire, Caribbean Netherlands. We estimate impacts of control of terrestrial invasive species on the dry-forest, and on the coral reef, linked by changes in terrestrial sedimentation rates. To address absolute shortfalls in funding, we estimate willingness of SCUBA divers to pay for terrestrial invasive species control. We find significant negative relationships of donkey density with vegetation ground cover; and a significant positive relationship of ground cover on the watershed with coral cover at depths below 10m. Using these models we estimate the impacts on coral cover of strategies to control grazing, including fencing and eradication. Cost curves for each strategy indicated that fencing of watersheds to exclude grazers presented the most cost effective solution within a 50-year time frame. We conducted choice experiments with SCUBA divers to estimate willingness to pay for control of terrestrial invasive species, where this would improve reef health. Willingness to pay exceeded the total costs of both fencing and eradication. We illustrate that control of terrestrial invasive species can lead to benefits in both terrestrial and marine ecosystems, and that funding for such projects may be possible via marine stakeholders. The combination of both terrestrial and marine considerations into invasive species control can greatly improve efficiency, while ensuring funding is allocated to address all threats to ecosystems under direct use.

Incorporating interaction networks into conservation: Tasmania as a case study

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Quantifying the direct impacts of invasive species requires time and resources which are not always available. In systems with limited information, qualitative interaction networks provide a method in which to explore the potential interactions between species at the community level. In Tasmania, hollow-breeding bird communities have been invaded by five hollow-nesting birds and one hollow-using, predatory marsupial, contributing to the decline in populations of several threatened species. While some interactions between native and invasive species on the island have been well documented, little information exists on the impact of most invasive species across the island. The aim of this research is to develop a model which quantifies the likely competitive interactions between hollow-breeding species across Tasmania in order to determine the potential impacts of unstudied invasive species. Hollow-breeding communities are an ideal community in which to study competitive interactions because there is direct competition between species over shared resources and it is possible to include all species in the community. Here we use species traits to model individual species breeding niche space, and use a metric of niche overlap between species to build qualitative networks representing potential competitive interactions for entire hollow-breeding communities. This method highlights known impacts of established invasive species and can be used to model the potential interactions of alien species present but not yet established.

Invasive plants of the Caribbean: application of herbarium collections to protect a regional biodiversity hotspot

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The Caribbean Islands represents a biodiversity hotspot with over 650 Critically Endangered or Endangered species. Collation and dissemination of knowledge is a requisite to address the problem of invasive species, a major driving force of species extinction with many other serious socio-economic impacts. This poster describes how, building on the keystone work, 'Catalogue of seed plants of the West Indies' and analysis of over 14,300 georeferenced herbarium accessions at the Smithsonian Institution, a project has collated data on over 570 invasive plant species prioritised from 1,879 plants identified as exotic to the region. Expert authors were selected to compile datasheets on each species from records in the herbarium and in scientific journal articles and authoritative databases. The datasheets were peer reviewed and submitted to CABI for final style edits and publication in the Invasive Species Compendium (ISC), a scientific knowledgebase with global coverage and reach. As of February 2017, 253 of 417 completed datasheets have been published. Inclusion in the ISC provides an Open Access platform for comparison with other taxa and geographic regions within a sustainable programme where information will be updated. Data are collated and presented with particular focus on risk assessment, management of pathways, public awareness, policy development, identification, detection, and options for control. Future work will include in-country gap analyses through consultation and comparison with locally compiled invasive species lists.

Can large database mining inform invasive non-native species management on islands?

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Global databases, including the IUCN Red List database, the Global Invasive Species Database, and the Threatened Island Biodiversity Database represent invaluable assets for investigating global patterns of extinction risk in insular vertebrates and target priority islands and species for conservation or eradication management. In view of the growing number of studies mining these databases to inform global conservation priorities, we ask two key questions: 1) what questions can these data most effectively address or not address; and 2) are the recommendations issued useful to practitioners and policy makers? Here, we critically assess the quality of the evidence used for quantifying global impacts of invasive non-native species on island vertebrates, and the methodology used in analyses of large publicly-available datasets. We provide recommendations on how to overcome limitations identified in the data, their processing and reporting, and suggest perspectives to address critical knowledge gaps.

Management of numerous introduced plants on Matiu (Somes Island), Wellington, New Zealand

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Matiu (Somes Island) is a 24.9 ha island in Wellington Harbour, New Zealand. The island has been free of introduced wild mammals since the late 1980s and provides a valuable opportunity to restore coastal forest ecosystems, including biota now rare or extinct on the mainland. Despite being only c. 2.5 km from the mainland, experience to date suggests invasion by invasive plants from the mainland is generally unlikely, although this situation may change in future. Restoration planting began in 1981. Major efforts to manage numerous plants known to threaten the restoration and protection of the island's native biodiversity ("weeds") began in 1998 and were initially somewhat ad hoc. Due partly to the retention of skilled personnel the island's weed management strategy has been refined greatly since 2007, including: enhancing biosecurity procedures pertaining to weeds; developing a thorough, systematic and regular approach to surveying; considering *all* introduced plants and implementing a precautionary approach (erring on the side of controlling plants that *may* be a threat, especially if rare and easy to kill); upskilling personnel; more strategically dividing volunteer, staff and contract labour and prioritising control work (including placing greater emphasis on early detection and nascent foci); and increasing the diversity of the island's native vegetation to enhance its resistance to weed invasion. Of 129 plants of concern to date, 73 (57%) are rated as posing a very high, high or moderate threat and 53 (73%) of those are now considered rare, possibly eradicated or probably eradicated. Major progress has also been made in most other areas of the weed management strategy, although some tenacious weeds remain a challenge. Lessons learnt on Matiu during the last 20 years may be applicable to other sites, including larger ones; sites with multiple land uses, owners and management regimes; and sites with greater chances of weed invasion.

Managing *Vespula* wasp invasion in New Zealand

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Introduced *Vespula* wasps cause severe problems to New Zealand ecosystems. Though vespulid wasps have successfully invaded most of New Zealand's offshore islands, little is known about their abundance and population development on those islands. Anecdotal observations suggest three offshore islands in the Hauraki Gulf and on the coast of the Coromandel (Little Barrier Island, Korapuki and Tiritiri Matangi) have become vespula wasp-free following successful mammal eradication. This study aims to investigate the drivers of successful wasp suppression and the prevention of reinvasion. Wasp monitoring will be conducted on different offshore islands along the northern east coast of New Zealand's North Island to measure the relative abundance of wasps and to collect a database on the island's environmental parameters. The combination of wasp trapping and a molecular analysis of paternity levels will allow us to estimate nest densities on offshore islands. The proposed study is novel because it will use a combination of methods (field based and molecular) to assess the density of vespula wasps in low-density areas (not beech forest). This database will also serve as a baseline for future investigations on pest dispersal and colonisation processes. It is crucial that we improve understanding of how different factors influence the development of wasp colonies to elaborate efficient pest control plans. The efficiency of five novel control methods will be forecasted using population modelling on colony and landscape scales.

Invasive rat colonisation history and movement dynamics in Haida Gwaii

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The brown rat (*Rattus norvegicus*) and black rat (*R. rattus*) are among the most invasive species worldwide with distributions encompassing every inhabited continent. Through predation and competition, invasions from both species have caused range contractions, local extirpation, and extinctions, resulting in reduced biodiversity. On Haida Gwaii, invasive rats have been implicated in population declines of six seabird species. Eradications were conducted on several islands where important nesting sites for sea-birds exist. On the Bischof Islands, reappearance of rats post-eradication has been observed. The objectives of this research are to investigate population history and movement dynamics of invasive rats in Haida Gwaii. Presently, 551 brown and black rats have been sampled from eighteen islands, collected from 2008 to 2016. Pre- and post-eradication samples were collected from the Bischofs allowing for an explicit evaluation of re-emergence versus re-colonisation in these locations. Genomic DNA was extracted from ear samples and used to construct double digest restriction site associated DNA sequencing libraries and sequenced using the Illumina HiSeq2500 PE125 platform. Single nucleotide polymorphisms (SNPs) were identified, genotyped, and used to assign individuals to species using a Bayesian clustering approach. The two species were then separated, and SNPs were re-identified and genotyped for further analysis. Resulting SNP data will be analysed using a series of population genetic and spatially-explicit analyses to determine the source of re-established populations and quantify the extent and direction of gene flow throughout the system. Genotypic data are being collected such that they offer full connectibility to a global SNP database of brown rats to infer potential sources of the populations in Haida Gwaii. Results of these analyses will help facilitate future eradications and provide useful insights to prevent the spread of rats elsewhere within the system.

B. Skei

Garden cans and river rafts – equipped to approach invasive freshwater fish

B. Skei

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How can environmental and fishery managers benefit from a natural toxin when aiming to maintain healthy native aquatic ecosystems? Rotenone is the only substance on the EU Biocides Regulation 528/2012 product-type 17 (piscicides) list and considered one of the most environmentally benign toxicants available for eradication of invasive fish. The substance is distributed in the formulated product CFT Legumine (CFT-L). In the wake of a CFT-L treatment, rotenone persistence in natural waters differs from a few days to several weeks depending on the season. Unless all parts of a large water body or catchment can be treated more or less simultaneously, the breakdown of rotenone may allow fish migration back into previously treated areas, i.e. undermining a successful treatment operation. When aiming for treatment of invasive alien species against a complex hydrogeological backdrop, standard tools are often pushed towards customised equipment. This poster presents equipment and techniques used in CFT-L treatment of diverse habitats such as groundwater entries, ponds and lakes, streams and rivers, tarns and marshlands, opening a toolbox containing garden cans, peristaltic pumps, backpack pumps and river rafts.

What happens after the helicopters have gone – assessing post-eradication changes on Macquarie Island

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In 2014 an eradication operation targeting house mice (*Mus musculus*), black rats (*Rattus rattus*) and European rabbits (*Oryctolagus cuniculus*) on sub-Antarctic Macquarie Island was successfully concluded. Monitoring of outcomes since that time has been sporadic and some is partly anecdotal, however the changes apparent on Macquarie Island in the absence of pest species are nonetheless considerable and are significant indicators of the progressive recovery of an island ecosystem. Vegetation changes following cessation of rabbit grazing are the most visually dramatic and widespread and are demonstrated partly by use of photo-points. Censuses of some seabird and invertebrate species have documented changing trends in island populations. Further changes can be expected for decades to come although some changes will be influenced by changing climatic conditions.

Predicting the distribution of island invader bird species under climate change

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The red-vented bulbul (*Pycnonotus cafer*), the common myna (*Acridotheres tristis*) and the red-whiskered bulbul (*P. jocosus*) are three passerine bird species native to the Indian subcontinent that were transported to islands from the early 1900s. Nowadays, the common myna is considered established in 20 island territories, the red-vented bulbul in 11 (32 islands) and the red-whiskered bulbul established in four island territories. Considering that perturbations associated with human activities will continue to increase during the 21st Century, leading to unprecedented species transportation rates, understanding potential climatic ranges of these species could be crucial. Moreover, predicting future range shifts under various climate change scenarios could be very useful in order to better inform management strategies. This is particularly true for birds as climate is often assumed to be one of the main drivers of the distribution of this taxon at large spatial scale. Here, we used eight species distribution models, five global circulation models and four representative concentration pathways using presence data from both the native and alien ranges of the three species. The objectives of this study were to i) assess the potential invasion risk of the red-vented bulbul, common myna and red-whiskered bulbul; ii) highlight priority locations for the management of these species and prevention of their introduction; and iii) explore the likely influence of climate change on the future climatic range of each. Our world climate suitability maps for each species predict a latitudinal expansion of climatic range. Then, our projections highlight three major potential climatic pathways for the establishment of the three species around the coasts of Northern Brazil, Guinea Gulf and North-West of the United States.

Biosecurity Plan for invasive ants in the Pacific

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Invasive ants are very adept hitchhikers and invaders of novel ecosystems. They have the ability to move through a wide range of international trade pathways. Once established, invasive ants are very difficult to eradicate or control. Their impacts are felt across many sectors including, agriculture, horticulture, trade, tourism, human health and the environment. Pacific islands are un-adapted to the presence of invasive ants and largely devoid of native ant species. Their impacts are often more far-reaching than at other locations, threatening not only delicate and complex island ecosystems, but the livelihoods and wellbeing of island communities. In the face of climate change, invasive ants will further reduce the climate resilience and food security of subsistence economies. A best practice integrated biosecurity system is needed to prevent the entry and establishment of these species as well as mitigate impacts caused by priority invasive ants currently present in the region. We recommend a regional approach to invasive ant biosecurity be established, which includes the essential elements of prevention, early detection, rapid response, ongoing management, capacity building, outreach and research. This system should operate at island, national and regional scales.

Rat control to protect the Turks and Caicos rock iguana: monitoring and responding to rat activity on a Caribbean island Nature Reserve

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A significant proportion of the global population of the Critically Endangered Turks & Caicos rock iguana (*Cyclura carinata*) is found on the small 43 ha island of Little Water Cay, which is managed as a nature reserve by the Turks & Caicos National Trust (TCNT). Black rats (*Rattus rattus*) and feral cats (*Felis catus*) are also found on the island, which is connected by sand bars to two larger islands and within black rat swimming distance of the large inhabited island of Providenciales. While rat eradication is not currently thought sustainable, a control programme began in 2015 aiming to control rats to zero/low density, reducing predation pressure on young iguanas. The baiting programme uses the first-generation anticoagulant rodenticide diphacinone set in bait boxes on a 50 m grid across the island. Following three weeks of baiting in November 2015 anecdotal changes were observed; increased sightings of young iguanas, nesting least terns (*Sternula antillarum*) on the sand bar and presence of a juvenile Antillean nighthawk (*Chordeiles gundlachii*). However, a second scheduled baiting round in November 2016 showed that rats were once more found across the entire island. We therefore devised a monitoring system to observe the speed and distribution of the influx of rats, predicted to walk across from the adjoining cays and/or swim from Providenciales. After the second baiting season non-toxic chocolate wax monitoring blocks were set in 20 bait stations across the island and checked weekly for signs of rat activity. These data will inform the timing and duration of future rat control undertaken on the island, allowing us to maximise the conservation benefits to iguanas while minimising the amount of rodenticide used, and thus non-target impacts. TCNT staff have been trained in rat control and monitoring techniques and now lead this project to reduce the impact of invasive rats on this important species.

An innovative programme to protect the UK's seabird islands

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The UK supports globally significant populations of seabirds, including 80% of the world's Manx shearwaters (*Puffinus puffinus*) and almost 60% of northern gannets (*Morus bassanus*), with breeding populations mostly restricted to offshore islands. However, many of these islands have one or more invasive non-native mammals present which negatively impact seabirds as well as many other native species. In 2013 the RSPB's innovative Seabird Island Restoration Project was established to protect these important islands using three key approaches. Firstly, we have developed a best practice toolkit for UK ground-based rat eradication projects, to be launched in 2017. This toolkit is based on international standards but tailored to the UK environmental, legal and social situation, consisting of technical advice documents on planning and carrying out eradication, biosecurity and incursion response work, as well as templates and series of worked examples. We have also collaborated on a prioritisation exercise to identify the UK islands where the greatest conservation gains can be made through eradication of invasive non-native mammals (eradication priorities) and where the greatest losses would be expected to occur were brown rats (*Rattus norvegicus*) to arrive on currently rat-free islands (biosecurity priorities). Finally, we are building UK capacity in island restoration through supporting UK-based conservation organisations, offering training in biosecurity, safe and effective rodenticide use, and incursion response planning, as well as writing, and supporting others to write, biosecurity plans, feasibility studies and operational plans. We have supported and trained two incursion response teams, one in south-west England and one in north Scotland, and plan to extend this network UK-wide. We believe this combination of working at the sites where the greatest conservation gains can be made, with well-trained people following tailored best practice guidelines offers the best chance to protect the UK's iconic seabird island heritage.

Prioritising islands for the eradication of invasive vertebrates in the Arctic

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As human activity increases and climate warms, invasive alien species pose a serious, growing threat in the Arctic to native biodiversity, ecosystems, and inhabitants, particularly those of Arctic island ecosystems. Consequently, Arctic states have recognised the need to eradicate invasive alien species from Arctic island ecosystems. The Arctic Council – an intergovernmental forum comprised of eight countries and six Permanent Participants that represent Arctic indigenous peoples – has defined a collective priority for upcoming action in the Arctic Invasive Alien Species (ARIAS) Strategy and Action Plan: "actively facilitate the eradication of invasive alien species from island ecosystems throughout the Arctic, as well as the recovery of native island species and habitats that have been impacted by invasive alien species". Prioritising islands for eradication activity is both necessary and strategically important in order to achieve this goal with limited resources across multiple jurisdictional authorities. This paper will explore the application of a study published in Conservation Biology ("Prioritising islands for the eradication of invasive vertebrates in the United Kingdom overseas territories") to the development of a prioritisation schema for the eradication of terrestrial invasive vertebrates on Arctic islands. The paper will provide a summary of key findings, including the identification of relevant data gaps; a proposed Arctic island prioritisation schema for the eradication of terrestrial invasive vertebrates; and a summary of further needs for input from scientific and policy perspectives. These findings will be applied to the ARIAS Strategy and Action Plan Steering Committee's efforts to develop a plan for the eradication of invasive alien species from Arctic island ecosystems.

Changes in forest passerine numbers on Hauturu following rat eradication

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Passerines were monitored on Hauturu (Little Barrier Island) over 15 years (1975–89) spanning the period (1976–80) when feral cats were eradicated from the island and again for the period 2013–2017. All birds seen and heard were recorded while walking three transects representing an altitudinal range from near sea level to approximately 550 m above sea level. Analysis of variance statistics were used to test for differences in bird numbers between transects and between years. Bird species were examined by transect to test for changes in numbers over time. Following cat eradication three species had increased on some transects, and two species had decreased on some transects, but it was difficult to attribute changes in bird numbers to the one cause which we were able to study: reduced cat numbers. Following rat (*Rattus exulans*) eradication in 2004 there have been significant increases and decreases of forest dwelling passerines. Field work for this study was completed in March 2017 and the data have yet to be analysed in detail.

Habitat features that influence predation of endangered Hawaiian common gallinule nests by invasive vertebrates in Hanalei and Huleia National Wildlife Refuges

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Hanalei and Huleia National Wildlife Refuges (NWRs) on the island of Kauai in Hawaii are designated as core wetland areas essential to the recovery of five endangered Hawaiian waterbird species. These two sites support approximately 50% of the endangered Hawaiian common gallinule (Alae ula, *Gallinula galeata sandvicensis*) population state-wide. On Hanalei NWR, taro (*Colocasia esculenta*) farming provides dense emergent aquatic vegetation needed for breeding gallinules, but previous research suggests that dikes, water drawdowns, and harvested fields may increase access by introduced mammalian predators. Although these studies documented egg predation, researchers were unable to determine nest fates for 25% of the nests using observer-based methods. In this pilot study, we evaluated the use of remote motion detection cameras as a method to determine gallinule nest fates and elucidated factors related to predation events through the early brooding phase. We predicted that taro farming practices influence predation by invasive vertebrates (e.g. feral cats (*Felis catus*), rats (*Rattus* spp.)) and negatively affect gallinule nest success in taro fields, when compared to managed wetland units that have fewer dikes, suitable vegetative cover, and stable water levels. Higher gallinule nest success in wetland units, coupled with reliable data regarding drawdowns and predation of nests in taro fields, allows managers to implement more specific management and monitoring methods to control and reduce access of invasive vertebrates that prey on endangered gallinule nests in these critically-important wetland, riverine, and agricultural landscapes.

Using sUAS to direct trap placement in support of feral cat eradication on islands

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Trap location is one the most important elements in a successful trapping programme and requires specialists that can analyse microhabitats across a landscape and identify areas of likely cat (*Felis catus*) presence and key travel routes. This is particularly true when determining the location of walkthrough trap sets. Existing remote sensing data can help specialists identify macrohabitats where cat activity is suspected but is not collected at a fine enough resolution to resolve microhabitats or topographical features where cat activity is likely. Using a case study on Kaho'olawe, Hawaii we evaluate how placing very high resolution sUAS-derived data in the hands of trapping specialists can be used to direct trap placement reducing the need for time intensive exploration of the landscape. On Kaho'olawe (11,550 ha), there is considerable need to direct trap placement because the presence of unexploded ordnance (only 10% of the island is cleared to a depth of four feet and 69% of the island surface-cleared) poses a significant risk to staff safety and greatly increases project risk and cost. In this case study, we use traditional remote sensing techniques to select three representative study areas that have limited UXO concerns and estimated high cat habitat suitability. Each study area is mapped at a resolution of less than 5 cm and resulting products are reviewed in 2D and 3D by trapping specialist to select suitable trap locations. Trapping specialists evaluate each study area on foot using their normal protocols to determine trap locations. Finally, we evaluate the efficacy of sUAS direct trap placement by comparing the sUAS derived trap locations with the ground-truthed locations. The workflows for collecting, processing and analysing sUAS data that we describe should enable managers to determine if integrating sUAS into trapping programmes is a cost-effective and efficient way to improve project success.

Removal of invasive, black rats increases activity levels and population density of Christmas Island's last remaining endemic reptile

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Invasive black rats (*Rattus rattus*) have been implicated in the extinctions of native species across the globe, particularly where native fauna are predator-naïve and are within insular island systems. Through the process of introduced disease and predation, Christmas Island in the Indian Ocean has suffered catastrophic extinctions of four endemic mammals and four reptiles since the early 1900s. Up until now, the endangered Christmas Island giant gecko (*Cyrtodactylus sadleiri*) has resisted extinction, but the interactions of this rainforest-dwelling endemic species with invasive and abundant black rats are unclear. With the recent onset of feral cat eradication by the Australian Government's *Christmas Island National Park*, a greater understanding of the potential for rats to impact on threatened reptile species is critical. Here we will present novel findings from a large-scale manipulation experiment to determine the impacts of the removal (using poison bait) of black rats from primary rainforest areas on Christmas Island, and the consequential behavioural and population responses of giant geckos. Giant gecko activity levels were found to increase as rat activity dropped, and gecko population density doubled, from 27 to 62 geckos per hectare, when rats were no longer present in high densities in the rainforest, with the greatest effect occurring in the dry season, eight weeks after initial baiting. Interestingly, insect and forest bird activity was also observed to increase with the reduction of rat activity, suggesting the role of the black rat as a predator of other native forest species. This research will assist in predicting the consequences of increased rat predation on Christmas Island's last remaining endemic reptile, helping to guide future management of invasive black rats, and suggests the urgent need for further research on complex interactions between invasive species and native prey on Christmas Island, and a multi-species approach to any further predator eradication.

Effect of *Spartina alterniflora* invasion on benthic macro-invertebrate communities in Guangxi Zhuang Autonomous Region

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In order to assess the ecological impacts of *Spartina alterniflora* invasion in Guangxi Zhuang Autonomous Region, we analysed communities of macro-invertebrates in different habitats and with different invasion times of *Spartina alterniflora*. Results showed that Shannon-Wiener index and Simpson diversity indices differed between the *S. alterniflora* wetlands and a mangrove wetland, and macro-invertebrate communities in *S. alterniflora* habitat mainly differed from those of mangrove habitats based on the non-metric multidimensional scaling used in this study. Perhaps due to the invasion of *S. alterniflora*, the bivalve *Glauconome chinensis* became the predominant species, leading to a greater macro-invertebrate biomass in *S. alterniflora* wetlands than in mangrove wetlands. Species composition, biomass and diversity of macro-invertebrates were assessed between the different invasive years of *Spartina alterniflora* including 20 years, five years and one year. Results showed that the community structures of macro-invertebrates were distinctly different between the 20-year *Spartina alterniflora* communities and the other two communities. The biomass of macro-invertebrates decreased with the length of time *Spartina alterniflora* communities were established. No significant differences of richness of macro-invertebrates were found among different invasive years ($p<0.05$). The results also showed all of these changes of macro-invertebrates at different communities or different invasion time were related to the density of *Spartina alterniflora* based on multiple linear regressions.

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