Biodiversity and Conservation Science

Dirk Hartog Island National Park Ecological Restoration Project:

Stage Two – Year Four

Translocation and Monitoring Report

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Front page image: Greater stick-nest rat (*Leporillus conditor*) © Steve Reynolds/DBCA

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Summary

The fourth year of Stage Two of the Dirk Hartog Island National Park Ecological Restoration Project saw supplementation translocations of three mammal species to the island: dibbler (*Parantechinus apicalis*), Shark Bay mouse (*Pseudomys gouldii*) and greater stick-nest rat (*Leporillus conditor*). Dibblers bred at Perth Zoo's Native Species Breeding Program were released in October 2021. Shark Bay mice from Bernier Island in Shark Bay were translocated in April 2022. Greater stick-nest rats from East and West Franklin Islands in the Nuyts Archipelago in South Australia were translocated in May 2022. With 36 dibblers, 50 Shark Bay mice and 60 greater stick-nest rats released in 2021-22, this brings the total number of animals translocated to Dirk Hartog Island since 2017 to 658.

Recently translocated cohorts, as well as established populations of translocated and extant fauna, were monitored using a range of different methods. Translocations continue to be assessed against success criteria, prescribed in approved Translocation Proposals, with more progress made towards achieving these goals in 2021-22. The monitoring of species such as dibblers has proved to be challenging, but innovations in release strategies and monitoring techniques have been trialled and have proved to be more effective in promoting release site fidelity and the number of detections.

Here we present the results of the three supplementation translocations and monitoring undertaken between July 2021 and July 2022 on Dirk Hartog Island. We also report on the ongoing monitoring of extant mammals and reptiles on the island.

1 Background

The vision for the ecological restoration of Dirk Hartog Island National Park (DHI) is ‘to create a special place with healthy vegetation and ecosystem processes that support the full suite of terrestrial native mammal species that occurred there at the time of Dirk Hartog’s landing in 1616, and that this is highly valued and appreciated by the community’. By June 2021, the ecological restoration project had achieved eradications of sheep (*Ovis aries*), goats (*Capra hircus*) and feral cats (*Felis catus*) and translocations of six mammal species had been completed or commenced. A strategic framework for the reconstruction of the former fauna assemblage on DHI has been prepared (Morris *et al.* 2017) and outlines a further seven species to be translocated to the island.

1.1 Site Description

Dirk Hartog Island is located in the Shire of Shark Bay in Western Australia (WA) at approximately -26° S and 113° E, and forms part of the Shark Bay UNESCO World Heritage Area. It falls within the DBCA Parks and Wildlife Service’s Gascoyne District in the Midwest Region. The island is approximately 80km long and up to 12km wide with a total area of 63,300 ha, making it the largest island in WA. The island contains a range of terrestrial habitats, including *Acacia*-dominated shrubland communities, *Triodia*-dominated grasslands, *Thryptomene dampieri* heath, consolidated and mobile dune-systems with large areas of *Spinifex longifolius* and many small ‘birrida’ clay-pans vegetated by chenopods (Beard 1976).

Shark Bay bandicoots, dibblers and greater stick-nest rats were released in the area around Herald Bay (Figure 1), approximately half-way along the east coast of DHI. In 2021 Shark Bay mice were released in *Spinifex longifolius*-dominated dune systems between Tetradon Loop and Herald Heights (Figure 1). Previously, banded and rufous hare-wallabies had been released between Notch Point and Cape Ransonnet, with an additional release of rufous hare-wallabies around Herald Bay (Figure 1).

1.2 Rainfall

Dirk Hartog Island has a semi-arid climate, typically receiving most rain over the winter months but with occasional heavy falls in the summer and autumn due to cyclonic events. Annual rainfall for the reporting period (1 July 2021 to 30 June 2022) was 230mm, which is very close to the annual mean. The largest falls were in mid-October 2021 and early April 2022 (Figure 2).

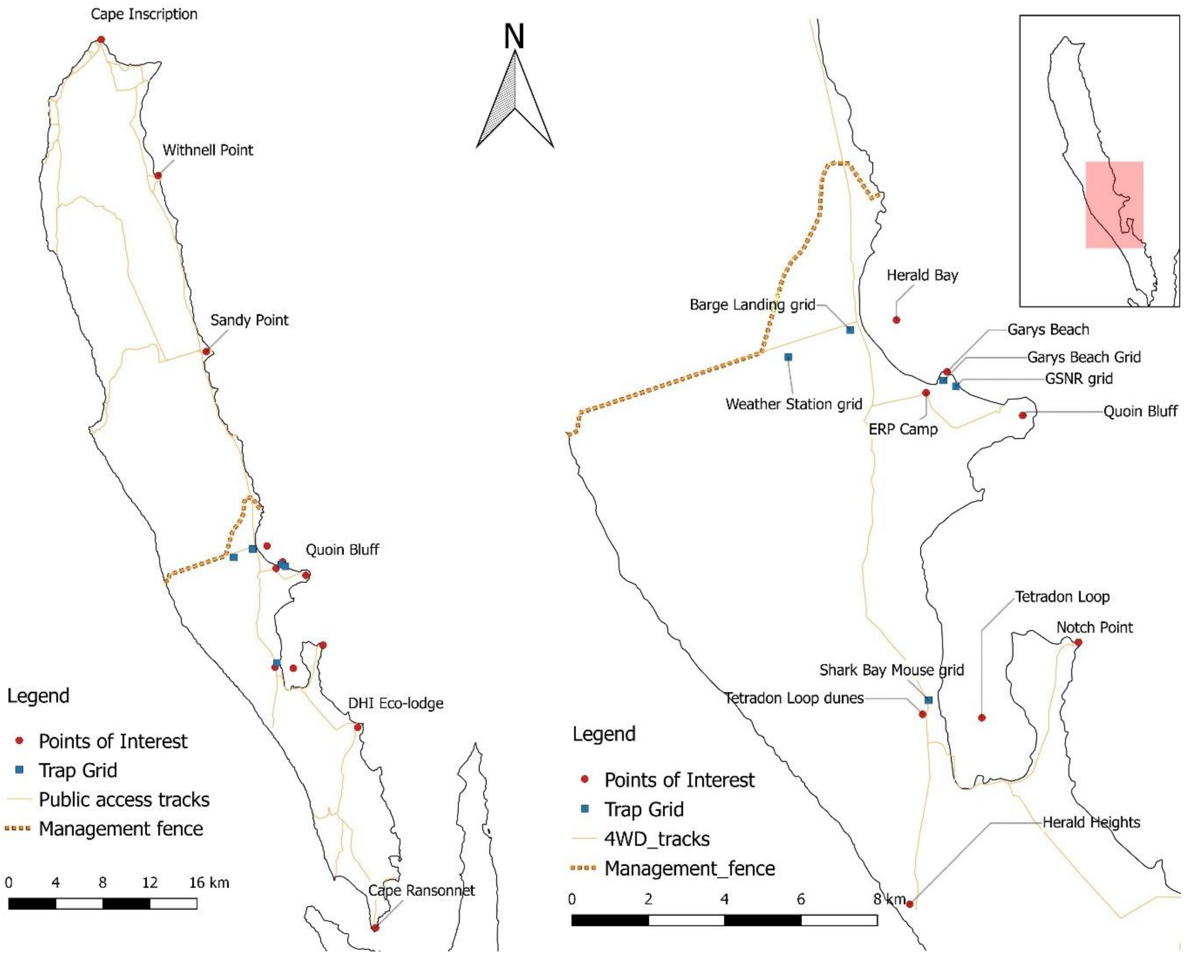


Figure 1. Overview (left) and close-up (right) of DHINPERP areas of operation in 2021-22.

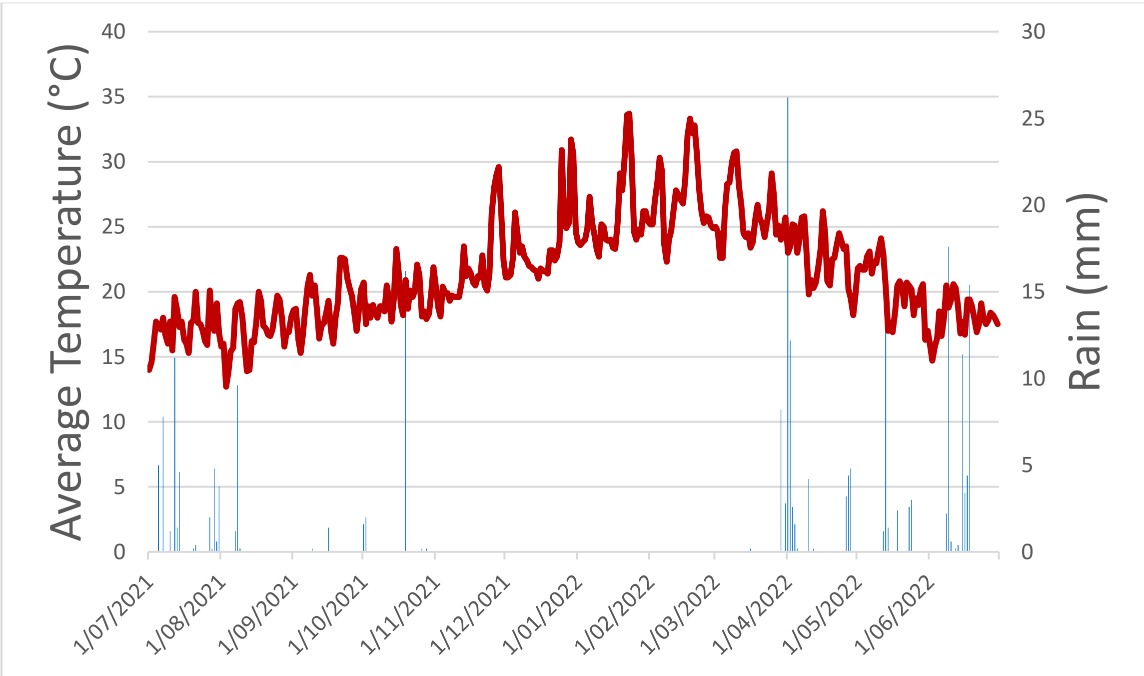


Figure 2. Climate data from DHI weather station (Herald Bay) between 1 July 2021 and 2 June 2022.

2 Dibblers

Dibblers (*Parantechinus apicalis*) were first translocated to Dirk Hartog Island in October 2019 from a captive breeding program at Perth Zoo (Cowen *et al.* 2020). The original founders of this population were from the islands of Boullanger, Whitlock and Escape in Jurien Bay. Further translocations to reinforce this initial founder cohort were undertaken in October 2020 and 2021 under DBCA Animal Ethics Committee Approval AEC 2020-20A.

2.1 Methods

2.1.1 Translocation

As per methods described in Cowen *et al.* (2020), 36 dibblers from Perth Zoo were translocated and released on DHI on 5 October 2021. This cohort consisted of 28 captive-bred subadults and 8 adults (Table 3). The sex ratio was 17 males and 19 females. Animals were released within or close to the same area as those translocated to DHI in 2019 and 2020 around the Barge Landing trapping grid at Herald Bay (Figure 1).

Table 3. Numbers of dibblers translocated and released on Dirk Hartog Island on 5 October 2021.

Source	Age	Female	Male	Total
Boullanger Island (Wild-born)	Adult	1	2	3
Perth Zoo (Captive-bred)	Adult	3	2	5
Perth Zoo (Captive-bred)	Sub-adult	15	13	28
Total		**19**	**17**	**36**

2.1.2 Soft-release and nest-boxes

The 2021 translocation of dibblers to DHI trialled the use of 'soft' release strategies with the aim of encouraging release site fidelity and improving post-release monitoring efficacy. 'Soft' or delayed releases have been shown to achieve these aims in other mammals (Resende *et al.* 2021), including a recent study of another dasyurid species, the chuditch (*Dasyurus geoffroii*) (Jensen *et al.* 2021).

Release sites were selected close to the Barge Landing grid at Herald Bay (Figure 1) in similar habitat to where previous detections of dibblers have occurred. The first strategy trialled involved soft-release pens 2.4 x 2.4m in size (Figure 3) where individual dibblers were held for up to 10 days with food, water and refuge provided, before the walls of the pens were removed and the dibblers could roam freely. Dibblers were recaptured (using Elliott traps baited with universal bait) at least twice within these pens to check their weight and condition.

The walls of the pens were constructed from panels of 6mm white acrylic plastic sheets, 1220 x 2440mm in size. Each pen was located around a dense, spreading shrub (mainly *Acacia ligulata*) and panels were dug into the soil to a depth of 100-150mm. Panels were fastened using L-shaped brackets and stainless-steel bolts and wingnuts. The brown paper covering the panels was left on the outside face to make the pens less obvious. Signage was displayed to discourage visitors from approaching or interfering with the pens.

The food supplied was in line with dibbler diet at Perth Zoo, with pinky (1-4 day-old) rats mixed with a minced meat mix developed for dibblers (Lambert 2012). Initially, each animal was provided with 6g of pinkies and 6g of mince mix but this increased to 8g of each by the end of the soft-release period in line with increased consumption. Food was supplied each evening around sunset and food bowls were collected before sunrise to avoid attracting ants and predators. Water was supplied in non-metal bowls and replenished as required. Food dishes were placed on a small concrete paver which was sprayed with permethrin (Coopex™, Bayer AG, Leverkusen, Germany) at 25g/5L dilution. A camera trap (Reconyx™ HF2X, Holmen WI, USA) was deployed facing the paver and SD cards switched each day to review images from previous day. A second camera was deployed outside the pen when the walls were removed. Cameras remained in place until the end of the reporting period.

Figure 3. Example of dibbler soft-release pen on DHI (credit: S. Cowen/DBCA)

In addition, a separate trial involved the release of dibblers from nest-boxes (originally designed for greater stick-nest rats (see Cowen *et al.* (2021))). The nest-boxes were also provided to the dibblers held in the soft-release pens. Each nest-box remained in-situ following the release and was monitored with a camera trap.

2.1.3 Cameras

In addition to the cameras deployed at soft-release and nest-box release sites, the grid of 25 lured camera traps were also used in 2021-22. The methodology for these cameras was the same as reported in Cowen *et al.* (2020). Cameras were serviced in September and November 2021 and March and June 2022.

2.1.4 Trapping

The Barge Landing and Weather Station trapping grids (Figure 1) were used to monitor dibblers in November 2021 and May 2022. Soft-release and nest-box release sites were also targeted with traps. Elliott traps were deployed at 60 points and baited with universal bait (peanut butter, oats and sardines).

2.2 Results

2.2.1 Soft-release

Nine dibblers were held in soft-release pens, nine were released from nest-boxes and the remaining 18 were 'hard' released from their transport boxes (Elliott traps filled with shredded paper and taped shut).

Soft-released dibblers were held for 10 days and were released fully on 15 October 2021. After five days, all nine dibblers were weighed and percentage weight losses since departing Perth Zoo varied between 4.3% and 21.4%. One individual gained 3.8%. Five individuals that lost more than 10% of their original weight were weighed again 3 days later and all had regained weight, including one individual that was 16.3% heavier than when it left Perth Zoo. All dibblers were observed feeding and visiting the water bowl. Although it could not be confirmed, most individuals appeared to be using the nest-box for refuge.

One unforeseen issue was visitation of the pens by grey butcherbirds (*Cracticus torquatus*). While apparently attracted to the pens by the provision of water, these birds are curious and potential predators of dibblers. While dibblers were observed easily evading the butcherbirds, this was still an undesirable outcome. Butcherbirds were also recorded approaching nest-boxes when dibblers were present and again, although the dibblers were apparently in no danger, this represents a potential risk that should be mitigated in future.

After the soft-release period ended, dibblers continued to be detected on cameras at pen sites for between 5 days and 152 days, with five sites recording dibbler activity more than 100 days post-release (Table 1). At nest-box release sites, dibblers were recorded between 0 and 163 post-release and again five sites recorded dibblers more than 100 days post-release (Table 1).

Table 1. Number of days when dibblers were detected on camera traps monitoring soft-release and nest-box release sites (cameras serviced on 23 March 2022).*

Site	**Days post-release***		**Site**	**Days post-release***
Pen 1	8		Nest-box 1	0
Pen 2	152		Nest-box 2	104
Pen 3	140		Nest-box 3	66
Pen 4	96		Nest-box 4	61
Pen 5	105		Nest-box 5	35
Pen 6	142		Nest-box 6	160
Pen 7	26		Nest-box 7	135
Pen 8	5		Nest-box 8	135
Pen 9	142		Nest-box 9	163

Figure 4. Dibbler visiting nest-box release site – November 2021 (credit: DBCA)

2.2.2 Cameras

There were three detections of dibblers by lured camera traps, with two detections at one site in the north-east of the grid on 13 and 27 September 2021 (i.e. before the 2021 translocation) and one detection at another site in the south-east of the grid on 10 November 2021. Dibblers had not been detected at either site previously.

2.2.3 Trapping

Trapping in November 2021 resulted in the capture of one individual dibbler, the same female (1238) that was captured in May 2021 with eight pouch young (Cowen *et al.* 2021). This individual had gained weight since May and the level of regression of her nipples indicated that she had weaned young in the past 50-60 days (Lambert 2012). Rodent abundance was particularly high in November 2021, with 298 individual ash-grey mice (*Pseudomys albocinereus*), 366 sandy inland mice (*P. hermannsburgensis*) and 251 house mice (*Mus musculus*) captured.

Trapping in May 2022 again resulted in the capture of the female 1238, which by this time had six pouch young. No other dibblers were captured. Rodent abundance was again high with a total trap success of 94% for rodents alone (602 captures in 624 trap nights. This reduced the availability of traps for dibblers which are relatively trap-shy compared to most rodent species.

3 Shark Bay mouse

Shark Bay mice (djoongari) (*Pseudomys gouldii*) were first translocated to DHI in April 2021, with an initial founder cohort of 80 individuals from Northwest Island in the Montebello Islands off the Pilbara coast (Cowen *et al.* 2021). After meeting all but one short-term success criteria and all medium-term success criteria, a supplementation of 50 individuals from the original natural population on Bernier Island took place in April-May 2022. The Translocation Proposal for this reintroduction was approved in April 2021 and the translocation was undertaken under DBCA Animal Ethics Committee Approval AEC 2021-03A.

3.1 Methods

3.1.1 Translocation

Shark Bay mice were released on DHI between 28 April and 3 May 2022. A total of 50 mice were translocated, with a sex ratio of 27M:23F.

Table 4. Numbers of Shark Bay mice translocated and released on Dirk Hartog Island between April-May 2022.

Release date	Source	Female	Male	Total
28 April	Bernier Island	11	14	25
1 May		6	7	13
2 May		5	4	9
3 May		1	2	3
Total		23	27	50

Ten release points were used in the release area to the south-west of Tetradon Loop close to Herald Heights (Figure 1). At all of these points, two artificial refuges (‘pseud-home-ys’) were established to provide additional refuges for the released animals, as per the 2021 release (Cowen *et al.* 2021). One Shark Bay mouse was released directly into each of the artificial refuges and the remaining mice were ‘hard’ released (i.e. released directly from transport container (medium Elliott trap) into cover).

3.1.2 Radio-tracking

Twelve Shark Bay mice were collared under isofluorane general anaesthesia following the same methods used on dibblers and similar to those described in Sims *et al.* (2020), using Holohil BD-2C transmitters and a weak link made from cotton sewing thread. Transmitters had an expected battery life of 28 days (0.95g). These transmitters did not have a mortality function, so an attempt to track animals to their refuges was made daily and nocturnal tracking was undertaken when required (i.e. the same daily refuge used on three or more consecutive days) to confirm movement.

One passive VHF logger tower was also erected to record signal strength from any collars that were in range of the VHF antenna (mounted on a 6.5m high pole) 24 hours a day. As well as providing another method to confirm the presence of animals in the release area and that those animals were moving, some additional information was also collected on activity periods and patterns. Data were collected and analysed using software from radio-tracking.eu (Gottwald *et al.* 2019).

Some animals were recaptured by partially fencing off refuges using a drift fence created from plastic shower curtain (‘pseudo-no-roamys’). Several Elliott traps baited with universal bait were then placed inside the fence and in the vegetation surrounding the refuge site.

3.1.3 Cameras

Cameras were deployed at the 10 artificial refuge release sites from the 2021 translocation and were maintained until March 2022.

For the 2022 translocation, one remote camera was set on a single artificial refuge site at each release point on the day prior to the first night of releases. These cameras were deployed for the immediate post release period (~60 days) before being serviced. Monitoring of the 2021 release sites found that Shark Bay mice are readily identified from the three other rodent species present, based on their size compared to the aperture of the ‘pseud-home-ys’.

3.1.4 Trapping

Post-release trapping for Shark Bay mice utilised two transects running parallel through the release area in the Tetradon Loop dunes. Each transect consisted of 30 trap points, each with two traps baited with universal bait These sites incorporated the 17 release points used in April 2021.

Trapping took place from 21 to 25 September 2021 and 8 to 11 May 2022. Analysis of trapping data was conducted using the Spatially Explicit Capture Recapture (SECR) package (secr 4.5.3) in ‘R’ version 3.6.3 (R Core Team 2020) to provide density estimates.

3.2 Results

3.2.1 Radio-tracking

A summary of the outcomes of radio-tracking 12 Shark Bay mice collared during the 2022 release is shown in Table 2. One mortality was recorded after nine days, most likely relating to predation by a grey butcherbird, based on the condition of the carcass and the way it had been hung in a Tamala rose (*Diplolaena grandiflora*) shrub. The outcomes of four other collared animals were unknown, with three collars either found chewed, broken or slipped-off between eight- and 17-days post-release. Another collar was not relocated after five-days post-release and after extensive searching (including 90mins tracking in an aircraft) could not be found. It is presumed that the collar had malfunctioned, possibly due to chewing on the antenna as several other collars were found to have been damaged in this way.

The remaining seven collars were removed by recapturing the collared individual. Mean weight gain of these individuals was 3.9%, with one individual gaining 25.5% on its initial body weight in the 20 days following release (Table 2). This contrasts with weight loss observed in translocated Shark Bay mice in 2021 (Cowen *et al.* 2021).

Table 2. Results of post-release monitoring of collared cohort of Shark Bay mice on DHI (April-May 2022) (collar not relocated after this date despite extensive searching – presumed collar malfunction).*

Animal ID	Sex	Release date	Collar retrieval	Outcome	Days elapsed	No. refuges	Weight change
PG2215	M	29/04/22	08/05/22	Mortality (bird)	9	3	n/a
PG2208	M	29/04/22	19/05/22	Live	20	6	+25%
PG2204	M	28/04/22	06/05/22	Chewed	8	2	n/a
PG2220	F	29/04/22	18/05/22	Live	19	8	-12%
PG2225	F	29/04/22	18/05/22	Live	19	6	+5%
PG2203	M	28/04/22	19/05/22	Live	19	4	-7%
PG2209	M	28/04/22	02/05/22*	Unknown	5	0	n/a
PG2243	F	02/05/22	22/05/22	Live	20	6	+16%
PG2229	F	01/05/22	09/05/22	Collar found	8	4	n/a
PG2228	F	01/05/22	22/05/22	Live	21	4	-6%
PG2238	M	01/05/22	18/05/22	Broke	17	6	n/a
PG2233	M	01/05/22	22/05/22	Live	21	5	+6%

3.2.2 Cameras

Shark Bay mice were noted at all ten 2021 artificial refuge sites for the duration of the 11-month deployment, including regularly entering the refuges, confirming ongoing presence in the release area between trapping surveys. Four other species of rodent were also detected by these cameras, including a greater stick-nest rat.

3.2.3 Trapping

Trapping in September 2021 resulted in the capture of 33 individual Shark Bay mice (13M:20F), including 25 new individuals that had been born on DHI since April. Four of the 16 new females were noted as being pregnant, with three of the four founder females captured also pregnant at this time. A SECR analysis on this trapping data revealed an estimated density of 0.51/ha (SE 0.12).

Trapping at the same sites in May 2022 resulted in the capture of 46 individuals (25M:21F), including 38 new individuals. Of the remaining eight recaptures, four were founders from the 2021 translocation and four were island-born individuals first captured in September 2021. SECR analysis revealed that density had doubled over this period for that area to 1.18/ha (SE 0.26).

4 Greater stick-nest rat

Greater stick-nest rats (wopilkara) (*Leporillus conditor*) were translocated to DHI for the first time in May 2021. The first founder cohort was taken from Salutation Island in Shark Bay, with a supplementation planned for 2022 from the original natural population on East and West Franklin Islands in South Australia. The Translocation

Proposal for this supplementation was approved in November 2021 and the translocation was undertaken under DBCA Animal Ethics Committee Approval AEC 2021-50A.

4.1 Methods

4.1.1 Translocation

Greater stick-nest rats were translocated to DHI on 27 and 28 May 2022, with a total of 30 individuals translocated from each of East and West Franklin Islands. The sex ratio was 36M:24F. Since greater stick-nest rats on the Franklin Islands do not build nests, it was impossible to target family groups. Therefore, all translocated animals were treated as individuals, unlike in 2021 when some family groups were translocated and released together (Cowen *et al.* 2021).

As in 2021, prior to release artificial refuges (or 'protonests') were constructed in the release area, using a pre-existing scaffold such as a dead shrub and consisting mostly of *A. ligulata* branches arranged in a tepee-like fashion, with a collection of smaller woody sticks making up the shelter underneath. Animals were transported in purpose-built boxes, with sliding doors at either end, secured with a screw. Animals were released directly into protonests as calmly and quietly as possible by placing transport boxes into/under the protonests and the doors gently slid open, allowing animals to quietly emerge in their own time. The transport boxes were left as an extra refuge that would retain the animal's scent and encourage fidelity to the release site. Animals were released at 60 protonests.

Table 6. Numbers of greater stick-nest rats translocated and released on Dirk Hartog Island on between in May 2022.

Release date	Source	Female	Male	Total
27 May	East Franklin Island	11	19	30
28 May	West Franklin Island	13	17	30
Total				**60**

4.1.2 Radio-tracking

A total of 13 individuals were fitted collars with Holohil RI-2DM transmitters and a weak link made from multi-strand embroidery thread. Collars were planned for deployment for four to five weeks post-release. Collared animals were recaptured using a combination of Sheffield traps, hand capture and hand-netting.

As per 4.1.2, a total of nine passive VHF logger towers were erected in a ring around the release area between Garys beach and Quoin Bluff. Modelling showed that these towers would provide coverage of much of the eastern part of the island between the management fence and the Tetradon Loop dunes. Data from these towers could be used to ascertain which collared greater stick-nest rats had been in the vicinity, providing a first step to locating any 'missing' collars of dispersing individuals. This array included towers at Garys Beach and near Quoin Bluff to provide information on behaviour of less mobile collared individuals. Data were collected and analysed using software from radio-tracking.eu (Gottwald *et al.* 2019).

4.1.3 Cameras

Cameras deployed in 2021 remained in use until May 2022, when they were redeployed to the 2022 release area. Cameras were set up on 20 protonests prior to the second release of animals and were serviced in early July 2022.

4.1.4 Trapping

Post-release trapping for greater stick-nest rats targeted occupied nests, refuges and other areas of activity in 2021, as well as a grid set over the 2021 release area in June 2022. Both cage traps and large Elliott traps were used and were baited with universal bait and other food items thought to be attractive to this species (e.g. fresh corn). Some traps were pre-baited for up to two days in an attempt to promote capture success.

Trapping in spring took place from 27 to 30 September 2021, with some additional traps set between 8 and 12 October. Trapping in winter took place between 24 and 26 June 2022.

In addition to conventional trapping methods, in 2021 the use of sensors to remotely monitor Passive Integrated Transponder (PIT) tags (i.e. microchips implanted in animals) was trialled in collaboration with researchers at La Trobe University (WildTrack). A total of 40 WildTrack modules were deployed in the vicinity of Garys Beach and Quoin Bluff, with the aim of detecting any tagged animals that came close enough for the modules to read their PIT tags. However, given the difficulty in trapping greater stick-nest rats, this was the main target species for the trial. The modules record PIT tag numbers using Radio-frequency Identification (RFID) technology and connect wirelessly to a base station (LoRa) to upload the data remotely via the Telstra Next-G network. All units are powered by solar panels so battery life is not a limitation.

4.1.5 Scat and track searches

Given the apparently trap-shy nature of greater stick-nest rats, to confirm ongoing presence in the release area, opportunistic scat and track searches were undertaken in and around the 2021 release area. Fresh scats were collected and frozen for use in a trial to evaluate faecal DNA as a monitoring method for this species.

4.2 Results

4.2.1 Radio-tracking

A summary of the outcomes of radio-tracking 13 greater stick-nest rats collared during the 2022 release is shown in 3. One mortality was recorded after eight days, most likely relating to predation by a raptor or owl, and a second was recorded 19 days post release, most likely as a result of predation from Gould’s monitor (*Varanus gouldii*). One collar broke away from an animal and was found hanging off a bush, however the transmitter had been in ‘live’ mode earlier that same day.

The remaining ten collars were removed by recapturing the collared individuals. Mean weight gain of these individuals was 9.1%, with one individual gaining 19.7%

on its initial body weight in the 35 days following release (however, this weight did include two attached, suckling young; Table 23). This contrasts with weight loss observed in translocated greater stick-nest rats in 2021 (Cowen *et al.* 2021).

Table 3. Results of post-release monitoring of collared cohort of greater stick-nest rats on DHI (May-July 2022).

Animal ID	Sex	Release date	Collar retrieval	Outcome	Days elapsed	Distance from release site (m)	Weight change
LC2211	M	27/05/22	15/06/22	Mortality (monitor)	19	1500	n/a
LC2218	M	27/05/22	04/06/22	Mortality (bird)	8	2100	n/a
LC2220	M	27/05/22	01/07/22	Collar found	35	47	n/a
LC2202	M	27/05/22	05/07/22	Live	39	6500	+4.8%
LC2206	F	27/05/22	01/07/22	Live	35	945	+3.4%
LC2214	F	27/05/22	01/07/22	Live	35	160	+19.7%
LC2245	F	28/05/22	05/07/22	Live	38	300	+10.9%
LC2246	F	28/05/22	01/07/22	Live	34	235	+9.8%
LC2253	F	28/05/22	02/07/22	Live	35	3700	+18.7%
LC2251	M	28/05/22	30/06/22	Live	33	2000	+5.7%
LC2247	M	28/05/22	02/07/22	Live	35	2600	+6.1%
LC2250	M	28/05/22	29/06/22	Live	32	2900	+10.1%
LC2257	M	28/05/22	03/07/22	Live	36	10500	+1.6%

In contrast to movements observed in 2021 (Cowen *et al*. 2021), females released in 2022 generally stayed within close proximity to the release area, with collared individuals recaptured on average 1.06km from the point of release (range 160m to 3.7km). While males had moved further on average (3.5km) between the point of recapture and their initial release location (range 47m to 10.5km).

4.2.2 Cameras

Much of the camera image data collected up to May 2022 are still to be analysed, with over 300,000 images required species IDs to be assigned. However, in July 2021 there were 176 independent detections of greater stick-nest rats at 22 camera sites in the Garys Beach release area, which had declined to 88 by October. This coincided with an observed increase in activity away from the immediate release area during this period.

4.2.3 Trapping

Trapping in September and October 2021 resulted in the capture of six individual greater stick-nest rats (3M:3F), of which five were founders and one was a new subadult female. Changes in weight varied widely, with percentage changes of -23%, -2%, +8%, +11% and +40% among the five recaptured founders. Interestingly, three of the five recaptures were individuals that had been collared during the 2021 release.

Trapping in June 2022 failed to capture any individuals, despite ongoing presence in the release area being confirmed (prior to the 2022 release nearby) through camera traps, scats and tracks and incidental observations.

As of 26 July 2022 (approximately two months post-release), WildTrack modules recorded 1033 detections, including 11 individual greater stick-nest rats. In addition, 29 Shark Bay bandicoots and two rufous hare-wallabies were also detected.

4.2.4 Scat and track searches

Given the difficulties in capturing greater stick-nest rats, monitoring the presence of animals in areas of previous activity was best achieved by opportunistic searches for scats and tracks. These searches confirmed the presence of greater stick-nest rats in the immediate vicinity of the release area at Garys Beach as well as other nearby sites such as south of Quoin Bluff and west of Herald Bay Camp.

Prior to the 2022 supplementation translocation, searches were undertaken in areas of previous activity in around 22-24 March 2022. Scats and/or tracks were again found in the main areas of activity around Herald Bay (Figure 5), providing evidence of ongoing persistence to justify the supplementation.

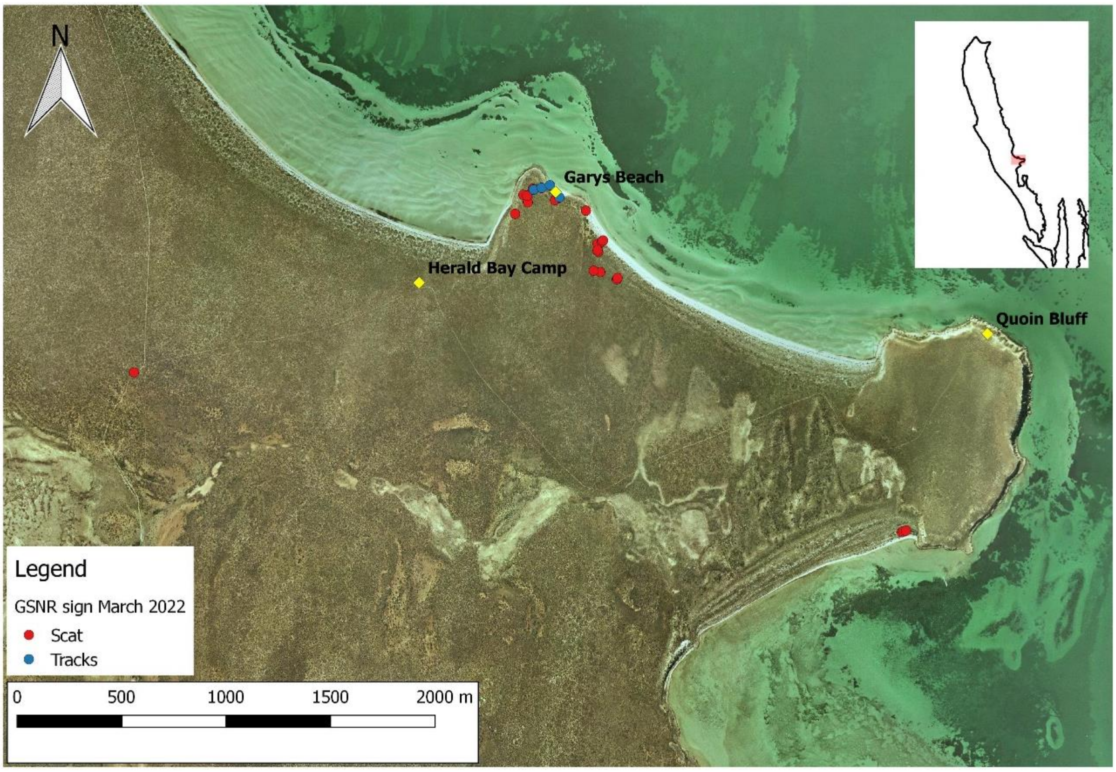


Figure 5. Map of Herald Bay-Quoin Bluff area showing locations of greater stick-nest rat scats and tracks observed in March 2022.

5 Shark Bay bandicoot

Shark Bay bandicoots (marl) (*Perameles bougainville*) were first translocated to Dirk Hartog Island in September 2019 from Bernier and Dorre Islands (Cowen *et al.* 2020) with an additional translocation to reinforce these initial founder cohorts in September 2020.

5.1 Methods

5.1.1 Cameras and incidental records

Shark Bay bandicoots were not targeted with camera traps in 2021-22 but were recorded incidentally on cameras deployed for other species.

As this species continues to establish on DHI, the extent of occurrence has increased. Changes in the extent of occurrence has been partly monitored by incidental records, such as observations of bandicoot tracks.

5.1.2 Trapping

Trapping to monitor Shark Bay bandicoots took place between 17 and 21 May 2022. To streamline the survey and to ensure that traps could be easily cleared within three hours of sunrise, only the Weather Station grid was used. This grid consisted of 60 trapping points, but due to time constraints in 2022 only 44 points were used. A Sheffield (cage) and Elliott trap were set at each point for four nights.

Analysis of trapping data was conducted using the Spatially Explicit Capture Recapture (SECR) package (secr 4.5.3) in 'R' version 3.6.3 (R Core Team 2020) to provide density estimates.

5.2 Results

5.2.1 Cameras and incidental records

Shark Bay bandicoots were regularly detected on camera traps deployed for other translocated species on DHI. For example, on the cameras deployed for greater stick-nest rats, there were 1,048 independent detections of bandicoots between 1 July 2021 and 24 May 2022 at 33 locations in the Garys Beach area. On the grid of 25 lured cameras for dibblers, there were 2,475 independent detections of bandicoots between 1 July 2021 and 24 March 2022. On multiple occasions, females have been detected with up to three young-at-foot, compared to the mean of 1.8 (Short *et al.* 1998).

Shark Bay bandicoot tracks were regularly observed in the central-south area of the island, north of Tetradon Loop and south of Louisa Bay. However, tracks were also recorded south of Notch Point in the south of the island and near Withnell Point in the north. The current approximate extent of occurrence (EOO) for this species on DHI is around 24,000ha, or 38% of the island. However, given how infrequently the northern and western areas of the island have been surveyed, it is likely to be much bigger than this. The IUCN Red List (Burbidge and Woinarski 2016) listed the global

EOO for this species as 15,000ha, prior to translocations to DHI (and Mt Gibson Sanctuary), so this represents a 160% increase.

5.2.2 Trapping

Trapping in May 2022 resulted in 34 captures of 19 bandicoots (11M:8F) from 176 trap nights, including six females with pouch young and nine new individuals. Among the recaptures were four individuals (1M:3F) that were founders released in 2019 from Bernier and Dorre Islands. Two of these females had pouch young. Unfortunately, one new female with two pouch young ejected both of them and although they were returned to the pouch which was taped and the female 'soft-released', one pouch young was found dead in the handling bag later on.

SECR analysis of the 2021 trapping data (Cowen *et al.* 2021) resulted in a density estimate of 0.42/ha (SE 0.08) across both the Weather Station and Garys Beach trapping grids. When just the Weather Station grid was included in the analysis, this estimate increased to 0.47/ha (SE 0.12). In comparison, on Bernier and Dorre Islands densities of Shark Bay bandicoots vary between 1.03/ha and 1.34/ha.

Density estimates for the 2022 data by comparison were somewhat higher, with 0.76/ha (SE 0.21) indicating an ongoing increase in the population at the Weather Station grid site, one of main release areas in 2019.

During trapping for greater stick-nest rats in the Garys Beach area in June 2022, high numbers of Shark Bay bandicoots were caught as bycatch, with 130 captures of 49 individuals in 306 trap nights. In contrast to the 19% trap success in May, this represents 42% trap success.

Further density estimates were obtained for the Shark Bay bandicoots from the trapping of greater stick-nest rats due to the high capture rates of bandicoots in these areas. Density around the immediate greater stick-nest rat release area in September 2021 was estimated at 1.28/ha (SE 0.53), whereas estimates over a larger trapping area around this same area in May 2022 were 0.69/ha (SE 0.12). These results suggest that the expansion of Shark Bay Bandicoot is showing a consistent if not increasing density at a larger scale.

6 Extant vertebrate fauna

Since 2017, the extant vertebrate fauna has been monitored on DHI, to evaluate the impact of the eradications on populations of these species, as well as the potential effect of restoring populations of locally extinct fauna. These data can then be compared with baseline monitoring data, obtained using identical methods at the same sites prior to the commencement of the eradication programs. This monitoring was undertaken under DBCA Animal Ethics Committee Approval AEC 2020-12B.

Incidental observation and camera trap data (obtained through monitoring for other translocated species) for vertebrate fauna on DHI were also collected.

6.1 Methods

6.1.1 Trapping

The trapping methodology used was a combination of Elliott traps and pitfalls at eight sites in the centre of the island for seven nights, as per Cowen *et al.* (2020). Traps were closed for one night in the middle of the trapping period due to a forecast thunderstorm but the full seven night's trapping was completed. The total number of trap nights were 658 for pitfalls (not including one pitfall that was not opened due to the presence of large numbers of ants) and 672 trap nights for Elliotts.

6.1.2 Cameras and incidental observations

Captures of 'non-target' incidental species on camera traps for surveys of translocated fauna were recorded and entered in the CPW Photo Warehouse database. Some taxa (e.g. rodents and hare-wallabies) were often not able to be identified to species level and entered as e.g. 'small mammal' or 'hare-wallaby'.

Incidental observations were recorded by personnel on a weekly basis on a communal list and entered into an Access database at the end of each week.

6.2 Results

6.2.1 Trapping

In October 2021, a total of 892 individual animals were captured, representing a 17% increase from 2020. Individual species totals are shown in Appendix 1, but one new species for the trapping program (since 2017) was recorded: yellow-faced whipsnake (*Demansia psammophis*). Overall, 29 species of reptile and mammal were recorded. Sandy inland mice represented 440 (49%) of the overall captures, with ash-grey mice making up 146 (16%). These represent increases of 52% and 51% respectively from 2020. However, house mouse captures increased markedly with 77 in 2021, compared to 19 in 2020, representing a 300% increase in capture success. In contrast, captures of little long-tailed dunnarts (*Sminthopsis dolichura*) decreased by 46%, although the 28 individuals captured was comparable to surveys prior to 2020.

6.2.2 Cameras and incidental observations

A total of 122 species were either observed, captured on remote camera or captured in traps in the reporting period. These species have been collated in Appendix 5. Diamond dove (*Geopelia cuneata)*, rock dove (*Columba livia*) and galah (*Cacatua roseicapilla*) were detected for the first time since observations began in spring 2017, with a single individual diamond dove recorded on remote camera in the 2021 greater stick-nest rat release area; whilst single individuals of both rock doves and galah were observed at the Herald Bay fauna camp.

7 Banded and rufous hare-wallabies

Translocations of banded hare-wallabies (*Lagostrophus fasciatus*) and rufous hare-wallabies (*Lagorchestes hirsutus*) appear to have been highly successful, with rufous hare-wallabies in particular being encountered regularly, either on roads at night, around the Herald Bay camp or on camera traps. A banded hare-wallaby was sighted at night on the road near Notch Point on 21 May 2022, the first sighting of this species by the DHINPERP team since May 2019. However, DNA extracted from faecal pellets (scats) and detections on camera traps have confirmed this species is also doing well and expanded its extent of occurrence on the island.

7.1 Scat surveys

Faecal DNA has proved to be a promising tool for monitoring both of these species, which can be difficult to survey for effectively given they rarely enter live-capture traps. After a feasibility study in 2018 (Cowen *et al.* 2022) a trial survey was conducted in November 2019. November is thought to be the ideal month, as rainfall and humidity are usually lower than during the middle of the year, but solar exposure and temperatures are not as high as they usually are in summer and early autumn.

Analysis of scat collection data was conducted using the Spatially Explicit Capture Recapture (SECR) package (secr 4.5.3) in ‘R’ version 3.6.3 (R Core Team 2020) to provide density and abundance estimates. The best fitted models for each species provided density estimates of 0.058/ha (SE 0.014) for banded hare-wallabies and 0.036/ha (SE 0.010) for rufous hare-wallabies. This resulted in estimates of 24 (15-38) banded and 15 (9-25) rufous in the vicinity of the trial study area, which might indicate a small increase based on the number of hare-wallabies released in the vicinity of the survey area in 2017 and 2018. However, we were concerned that the number of recaptures of some individuals was apparently high and this may have been due to the discriminatory power of the microsatellite markers that were used.

Based on these results, a full-scale survey took place in November 2020 at three locations across the southern half of DHI. This resulted in the collection of 449 samples, of which 136 were genotyped as banded, 188 as rufous and 125 were unable to be determined. However, due to some of the potential issues with using microsatellite markers, an array of single-nucleotide polymorphism (SNP) markers was developed. The extra time required to develop this SNP array has delayed the analysis of the DNA obtained from this survey and it is still ongoing. Density and abundance estimates are expected by the end of 2022.

7.2 Camera and incidental observations

During monitoring for other translocated fauna using camera traps, images of both species of hare-wallaby were also recorded. Observations of animals and tracks and/or scats were recorded on an ad-hoc basis.

Rufous hare-wallabies continue to be regularly detected at nearly all camera trap sites in the Herald Bay area, with occasional detections of banded hare-wallabies in this area as well.

8 Discussion

8.1 Dibbler

Dibblers continue to be a difficult species to monitor on DHI, with few detections on lured cameras and only one individual captured during trapping sessions in November 2021 and May 2022. However, this female has bred again in 2022 and indications are that it had successfully weaned its eight offspring from the 2021 breeding season. Furthermore, the trial of soft-release pens and nest-box releases have dramatically increased the number of detections of dibblers at these sites, validating the use of these techniques.

However, while these methods have helped establish ongoing survivorship in the release area, it is important to confirm how many individuals are being detected and in 2022, without being able to capture more animals, other methods are necessary to achieve this. The use of WildTrack modules that can pick up PIT tags remotely was trialled in 2021 and showed promise for monitoring greater stick-nest rats and is intended to be incorporated into dibbler monitoring in 2022.

Further translocations of dibblers are planned in 2022 and possibly 2023 to bolster the 93 individuals that have been released so far. The original approved Translocation Proposal provided for up to 150 being released over three years, but so far this has fallen well short. While it is hoped that detections of this species will increase over time as the population increases, the development of more innovative release and monitoring strategies with assist.

8.2 Shark Bay mouse

Monitoring of Shark Bay mice in the 2021 release area in September 2021 and May 2022 indicates this population is establishing well, with survival of founders and substantial recruitment. To date, at least three of the four medium-term success criteria have been met for the first cohort, with ongoing presence in the release area, F1 progeny recorded and dispersal of new recruits into the Herald Heights area. While the condition of captured animals has been good, weights have not returned to the levels at the time of release. Shark Bay mice on Northwest Island are typically 10-20% heavier than animals from Bernier Island and the weights on DHI are more typical of the latter so this is not believed to be cause for concern.

The 2022 supplementation translocation of 50 Shark Bay mice from Bernier represents the completion of the translocation of this species to DHI. As in 2021, collared animals quickly established regular refuges, often with resident conspecifics. The single predation of a collared animal by a grey butcherbird in the three weeks of radio-tracking was an improvement on the four mortalities in 2021 (92% survival compared to 67%), three of which were the result of snake predation. Mean temperatures during the early post-release period were unusually high in 2021 (Cowen *et al.* 2021), but temperatures in 2022 were more typical for the time of year. Also, as was observed by Cowen *et al.* (2021), snakes are present on Bernier Island but not on Northwest, which may also have been a factor in these outcomes.

Camera trap detections have confirmed that the artificial refuges (‘pseud-home-ys’) deployed for Shark Bay mice in the release areas have continued to be used by this species (and other rodents). While it is unclear whether the refuges have played a role in the success of this translocation so far, they are a valuable innovation as they provide a focal-point to assist with detections and the aperture of the refuge entrance also allows more accurate species identification between the four smaller rodent species that now occur on DHI.

8.3 Greater stick-nest rat

Greater stick-nest rats have proved somewhat difficult to monitor using live-capture trapping. Anecdotally, trap-shy behaviour has been observed at other sites where this species occurs such as Arid Recovery (South Australia) and Mallee Cliffs Sanctuary (New South Wales), and at Mt Gibson Sanctuary in WA, recent trapping success resulted only after a week of pre-baiting (Australian Wildlife Conservancy *in litt*.). The presence of greater stick-nest rats in and around the release area has been confirmed through detections on camera traps, scat and track searches and incidental observations, but captures are important to confirm survivorship, recruitment and maintenance of condition and ultimately to assess progress against success criteria. More work is required to improve trapping success for this species and will likely require longer periods of pre-baiting and experimentation with bait types. The high trap rate of Shark Bay bandicoots in the same release area also interferes with trapping of greater stick-nest rats and may prove difficult to counter-act. However, the evidence of ongoing presence of this species in and around the release area 12 months post-release provides some confidence that the translocation is progressing. The reasons for the apparently delayed dispersal between July and October are unclear, but could relate to changes in the social structure between occupied protonests, or increased predator activity (i.e reptiles) as temperatures began to increase in spring.

The 2022 supplementation translocation of 60 greater stick-nest rats from East and West Franklin Islands represents the completion of the translocation of this species to DHI. Survivorship of collared individuals was comparable with 2021 (85% compared with 87%), with two mortalities resulting from predation events, most likely by an unknown bird (raptor or owl) and a Gould’s monitor. As in 2022, long-distance dispersal was limited to a few individuals, with most remaining in or close to the release area. There were no adverse outcomes associated with these movements. However, unlike 2021 it was males that dispersed the furthest, whereas females generally chose to settle in the immediate release area. As in 2021, augmentation of protonests was noted within a few days of release, which is especially remarkable given that the populations on the Franklin Island do not build nests, but rather refuge in shearwater burrows (Short *et al.* 2019).

8.4 Shark Bay bandicoot

The translocations of Shark Bay bandicoots in 2019 and 2020 appear to have been highly successful, and with the identification of F2 (and longer) generations present, the continuing expansion of the EOO and the maintenance of body weight and condition, three of the five long-term success criteria have now been achieved. Shark Bay bandicoots currently occur across approximately 50% of the island's area (probably more) and density in the release area has doubled between 2021 and 2022 (although it is less than recently recorded densities on Bernier and Dorre Island (Sims *et al.* 2022)). Recruitment has continued to be high, with litters of two pouch young frequently observed and records of three young-at-foot noted. Several individuals from the original founder cohort in 2019, continue to be captured and are still reproductive at four or more years old. No evidence of Bandicoot Papillomatosis Carcinomatosis Virus 1 has been recorded on DHI to date, despite conducting inspections on all captured individuals and screening of any suspicious lesions. However, there is an ongoing need for vigilance, particularly if the population is subject to any stressors in the future (e.g. drought). Monitoring of Shark Bay bandicoots will continue on an annual basis and genetic analysis of tissue samples conducted to confirm successful admixture between founders from Bernier and Dorre. As the Shark Bay bandicoot population increases, there will be a concomitantly increased risk of vehicle strikes and ongoing management of tourism on the island will be necessary to mitigate this.

8.5 Extant vertebrate fauna

As in previous years, extant rodents continue to be highly abundant on DHI, particularly the two native species of *Pseudomys*, ash-grey and sandy inland mice (Cowen *et al.* 2019, Cowen *et al.* 2020, Cowen *et al.* 2021). During the seven-night trapping session in October 2021, house mice numbers were much higher than in previous years but this was not mirrored in other trapping surveys where house mice continued to be substantially less abundant than the native species. In some areas, such as the Weather Station grid (Figure 1), used to monitor Shark Bay bandicoots, ash-grey mice capture success was higher than for sandy inland mice, but at the Barge Landing grid (Figure 1), the opposite was true. While capture success for native rodents on DHI appears to fluctuate somewhat, these species are consistently the most abundant native vertebrates on the island and it seems increasingly unlikely that this is 'boom' associated with good environmental conditions, but rather a population increase associated with the recovery of vegetation and reduced predation pressure after the eradications of sheep, goats and feral cats, that has been sustained over a period of several years. Captures of little long-tailed dunnarts were lower in 2021, but were still comparable to 2020 and this species continues to be more abundant now than pre-eradication or immediately post-eradication.

Further analysis is required to evaluate any trends in extant mammal and reptile abundance from the trapping data between 2017-2021 and compare this with data collected prior to the commencement of DHINPERP.

Raptors that may prey on mammals such as white-bellied sea-eagle (*Haliaeetus leucogaster)* and nankeen kestrel (*Falco cenchroides*) continue to be observed regularly on DHI, but there is no evidence of any significant predation on translocated fauna by these species. Other raptors and owls are rarely observed, including wedge-tailed eagles (*Aquila audax*) which still appear to be scarce or infrequent visitors to DHI. Predatory reptiles such as Gould's monitor, mulga snake

(*Pseudechis australis*), gwardar (*Pseudonaja mengdeni*) and Children's python (*Antaresia childreni*) continue to be observed relatively frequently and are likely to prey upon translocated fauna.

8.6 Hare-wallabies

Rufous hare-wallabies continue to be observed regularly across the southern portion of the island and their EOO (based on sighting and track records) is at least 50,000ha (or 80% of the land area of DHI). Banded hare-wallabies are not as widespread as rufous but are now recorded with increasing frequency in the Herald Bay area. A sighting in May 2022 was the first since 2019, but their ongoing presence and expansion of EOO has been confirmed by faecal DNA surveys and camera trap surveys for other species.

The results of the 2020 scat survey (Cowen *et al.* 2021) are anticipated by spring 2022 which will allow evaluation and augmentation of the methodology prior to the planned 2022 survey in November. The use of faecal DNA is likely to be the most effective monitoring method for these species and SECR analysis of the pilot study dataset from 2019 indicates robust population estimates can be obtained using this method. As a non-invasive technique, it will also assist with streamlining the broader monitoring program on DHI as more species continued to be reintroduced.

8.7 Planning for 2022-23

Three translocations to DHI are planned for 2022-23, representing two new species and one supplementation. The first translocation of western grasswren (*Amytornis textilis*) is planned for October 2022, harvesting founders from the two subpopulations in Shark Bay, focusing on Hamelin Station Reserve (managed by Bush Heritage Australia) and conservation estate on Peron Peninsula. In addition, the first translocation of brush-tailed mulgara (*Dasycercus blythi*) is tentatively proposed for May-June 2023, pending approvals and availability of source populations that will support a sustainable harvest. Finally, a further supplementation translocation of dibblers is timetabled for October-November 2022, depending on when weaning at Perth Zoo occurs.

A provisional schedule for translocation and monitoring work is outlined in Table 4.

Table 4. Provisional program for translocations and monitoring on DHI in 2022-23 (SBM, Shark Bay mouse; GSNR, greater stick-nest rat; WGW, western grasswren; BTM, brush-tailed mulgara; SBB, Shark Bay bandicoot).

Year	Month	Activity on Dirk Hartog Island
2022	Jul	
	Aug	
	Sep	monitoring of SBM and GSNR
	Oct	translocation of WGW and Dibblers
	Nov	monitoring of WGW, Dibblers and hare-wallabies
	Dec	
2023	Jan	

Year	Month	Activity on Dirk Hartog Island
	Feb	
	Mar	
	Apr	monitoring of SBM and WGW
	May	translocation of BTM; monitoring of SBB and Dibblers
	Jun	monitoring of BTM and GSNR

Appendices

Appendix 1

List of small vertebrate captures during trapping surveys conducted in October 2021.

Family	Species	Common name	Individual captures
Agamidae	*Ctenophorus maculatus*	Spotted military dragon	24
	Pogona minor minor	Western bearded dragon	1
Carphodactylidae	*Nephurus levis*	Smooth knob-tailed gecko	12
Diplodactylidae	*Diplodactylus ornatus*	Ornate gecko	9
	Strophurus spinigerus	Soft spiny-tailed gecko	10
Gekkonidae	*Gehyra variegata*	Variegated gehyra	10
	Heteronotia binoei	Bynoe's gecko	2
Pygopodidae	*Delma butleri*	Spinifex delma	1
	Lialis burtoni	Burton's legless lizard	4
	Pletholax edelensis	Shark Bay keeled legless gecko	6
Scincidae	*Ctenotus australis*	Western limestone ctenotus	7
	Ctenotus fallens	West coast laterite ctenotus	10
	Cyclodomorphus celatus	Western slender bluetongue	1
	Lerista elegans	Elegant slider	10
	Lerista lineopunctulata	Line-spotted robust slider	2
	Lerista planiventralis	Keeled slider	9
	Lerista praepedita	West coast worm-slider	9
	Lerista varia	Variable-striped robust slider	2
	Menetia greyii	Common dwarf skink	2
	Morethia lineoocellata	West coast morethia skink	26
Varanidae	*Varanus gouldii*	Gould’s monitor	6
Elapidae	*Demansia psammophis*	Yellow-faced whipsnake	1
	Neelaps bimaculatus	Black-naped snake	1
	Pseudechis australis	Mulga snake	1
	Simoselaps littoralis	West coast banded snake	14
Dasyuridae	*Sminthopsis dolichura*	Little long-tailed dunnart	28
Muridae	*Pseudomys albocinereus*	Ash-grey mouse	146
	Pseudomys hermannsburgensis	Sandy inland mouse	440
	Mus musculus	House mouse	77

Appendix 2

List of incidental sightings between July 2021 and June 2022, in addition to those collated through remote cameras and trapping.

Common name	Scientific name	Observation	Live capture	Remote camera
Sandhill Frog	*Arenophryne rotunda*	X		
Loggerhead Turtle	*Caretta caretta*	X		
Green Turtle	*Chelonia mydas*	X		
Shark Bay Heath Dragon	*Ctenophorus butlerorum*	X	X	
Spotted Military Dragon	*Ctenophorus maculatus*	X	X	
Western Netted Dragon	*Ctenophorus reticulatus*	X	X	
Western Bearded Dragon	*Pogona minor minor*	X	X	
Smooth Knob-tailed Gecko	*Nephrurus levis*	X	X	X
Barking Gecko	*Underwoodisaurus milii*	X		
South-western Clawless Gecko	*Crenadactylus ocellatus*	X	X	
Ornate Gecko	*Diplodactylus ornatus*	X	X	
Soft Spiny-tailed Gecko	*Strophurus spinigerus*	X	X	
Variegated Gehyra	*Gehyra variegata*	X	X	
Bynoe's Gecko	*Heteronotia binoei*	X	X	
Shark Bay Worm-lizard	*Aprasia haroldi*		X	
Spinifex Delma	*Delma butleri*	X	X	
Burton's Legless Lizard	*Lialis burtonis*	X	X	
Shark Bay Keeled Legless Gecko	*Pletholax edelensis*		X	
Peron's Snake-eyed Skink	*Cryptoblepharus plagiocephalus*	X		
Western Limestone Ctenotus	*Ctenotus australis*	X	X	
West Coast Laterite Ctenotus	*Ctenotus fallens*	X	X	
Western Slender Bluetongue	*Cyclodomorphus celatus*		X	

Common name	Scientific name	Observation	Live capture	Remote camera
Western Spiny-tailed Skink	*Egernia stokesii badia*	X	X	
Elegant Slider	*Lerista elegans*	X	X	
Line-spotted Robust Slider	*Lerista miopus*		X	
Keeled Slider	*Lerista planiventralis*		X	
West Coast Worm-slider	*Lerista praepedita*	X	X	
Variable-striped Robust Slider	*Lerista varia*		X	
Common Dwarf Skink	*Menetia greyii*	X	X	
West Coast Morethia Skink	*Morethia lineoocellata*	X	X	
Shark Bay Bobtail	*Tiliqua rugosa palarra*	X		X
Gould’s Monitor	*Varanus gouldii*	X	X	X
Yellow-faced Whipsnake	*Demansia psammophis*	X	X	
Black-naped snake	*Neelaps bimaculatus*		X	
Mulga Snake	*Pseudechis australis*	X	X	
Gwardar	*Pseudonaja mengdeni*	X		
West Coast Banded Snake	*Simoselaps littoralis*		X	
Children's Python	*Antaresia childreni*	X		
Southern Blind Snake	*Anilios australis*		X	
Brown Quail	*Coturnix ypsilophora*	X		
Stubble Quail	*Coturnix pectoralis*	X		
Australian Shelduck	*Tadorna tadornoides*	X		
Wilson's Storm Petrel	*Oceanites oceanicus*	X		
Little Pied Cormorant	*Phalacrocorax melanoleucos*	X		
Pied Cormorant	*Phalacrocorax varius*	X		
Australian Pelican	*Pelecanus conspicillatus*	X		
White-faced Heron	*Ardea novaehollandiae*	X		
Nankeen Night Heron	*Nycticorax caledonicus*	X		
Great Egret	*Ardea alba*	X		

Common name	Scientific name	Observation	Live capture	Remote camera
Little Egret	*Egretta garzetta*	X		
Eastern Reef Egret	*Egretta sacra*	X		
Osprey	*Pandion haliaetus*	X		
Wedge-tailed Eagle	*Aquila audax*	X		
White-bellied Sea-Eagle	*Haliaeetus leucogaster*	X		
Brown Falcon	*Falco berigora*	X		
Australian Kestrel	*Falco cenchroides*	X		X
Australian Boobook	*Ninox boobook*	X		
Buff-banded rail	*Gallirallus philippensis*	X		
Australian Bustard	*Ardeotis australis*	X		X
Painted Button-quail	*Turnix varia*	X		X
Whimbrel	*Numenius phaeopus*	X		
Common Sandpiper	*Actitis hypoleucos*	X		
Sanderling	*Calidris alba*	X		
Red-necked Stint	*Calidris ruficollis*	X		
Grey-tailed Tattler	*Tringa brevipes*	X		
Common Greenshank	*Tringa nebularia*	X		
Ruddy Turnstone	*Arenaria interpres*	X		
Pied Oystercatcher	*Haematopus longirostris*	X		
Sooty Oystercatcher	*Haematopus fuliginosus*	X		
Banded Stilt	*Cladorhynchus leucocephalus*	X		
Black-winged Stilt	*Himantopus himantopus*	X		
Red-necked Avocet	*Recurvirostra novaehollandiae*	X		
Greater Sand Plover	*Charadrius leschenaultii*	X		
Red-capped Plover	*Charadrius ruficapillus*	X		
Grey Plover	*Pluvialis squatarola*	X		
Banded Lapwing	*Vanellus tricolor*	X		

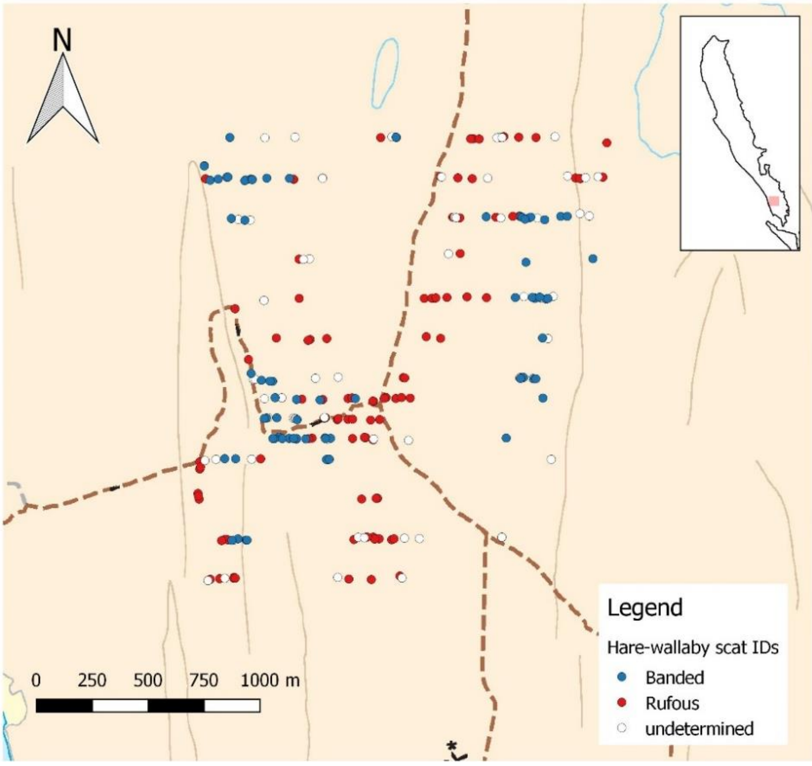
Common name	**Scientific name**	**Observation**	**Live capture**	**Remote camera**
Silver Gull	*Larus novaehollandiae*	X		
Pacific Gull	*Larus pacificus*	X		
Lesser Crested Tern	*Thalasseus bengalensis*	X		
Crested Tern	*Thalasseus bergii*	X		
Caspian Tern	*Hydroprogne caspia*	X		
Bridled Tern	*Onychoprion anaethetus*	X		
Galah	*Cacatua roseicapilla*	X		
Rock Dove	*Columba livia*	X		
Laughing Turtle Dove	*Spilopelia senegalensis*	X		X
Diamond Dove	*Geopelia cuneata*			X
Horsfield's Bronze Cuckoo	*Chrysococcyx basalis*	X		
Shark Bay Purple-backed Fairy-wren	*Malurus assimilis bernieri*	X		X
Dirk Hartog Island Black and White Fairywren	*Malurus leucopterus leucopterus*	X		
Dirk Hartog Island Emu-wren	*Stipiturus malachurus hartogi*	X		X
Dirk Hartog Island Rufous Fieldwren	*Calamanthus campestris hartogi*	X		X
Spotted Scrubwren	*Sericornis maculatus*	X		X
Singing Honeyeater	*Gavicalis virescens*	X		X
White-fronted Chat	*Epthianura albifrons*	X		
Crested Bellbird	*Oreoica gutturalis*	X		
Willie Wagtail	*Rhipidura leucophrys*	X		
Black-faced Cuckoo-shrike	*Coracina novaehollandiae*	X		
Little Woodswallow	*Artamus minor*	X		
Black-faced Woodswallow	*Artamus cinereus*	X		
Grey Butcherbird	*Cracticus torquatus*	X		X
Little Crow	*Corvus bennetti*	X		X
Australian Pipit	*Anthus australis*	X		X
Zebra Finch	*Taeniopygia guttata*	X		

Common name	Scientific name	Observation	Live capture	Remote camera
Welcome Swallow	*Hirundo neoxena*	X		
White-backed Swallow	*Cheramoeca leucosternus*	X		
Fairy Martin	*Hirundo ariel*	X		
Tree Martin	*Hirundo nigricans*	X		
Brown Songlark	*Cincloramphus cruralis*	X		
Silvereye	*Zosterops lateralis*	X		
Rufous Hare-wallaby	*Lagorchestes hirsutus*	X		X
Banded Hare-wallaby	*Lagostrophus fasciatus*	X		X
Shark Bay Bandicoot	*Perameles bougainville*	X	X	X
Little Long-tailed Dunnart	*Sminthopsis dolichura*		X	X
Dibbler	*Parantechinus apicalis*	X	X	X
Greater Stick-nest Rat	*Leporillus conditor*	X	X	X
Ash-grey Mouse	*Pseudomys albocinereus*	X	X	
Shark Bay Mouse	*Pseudomys gouldii*	X	X	X
Sandy Inland Mouse	*Pseudomys hermannsburgensis*	X	X	
House Mouse	*Mus musculus*	X	X	
Lesser Long-eared Bat	*Nyctophilus geoffroyi*	X		
Indo-Pacific Bottlenose Dolphin	*Tursiops aduncus*	X		
Humpback Whale	*Megaptera novaeangliae*	X		

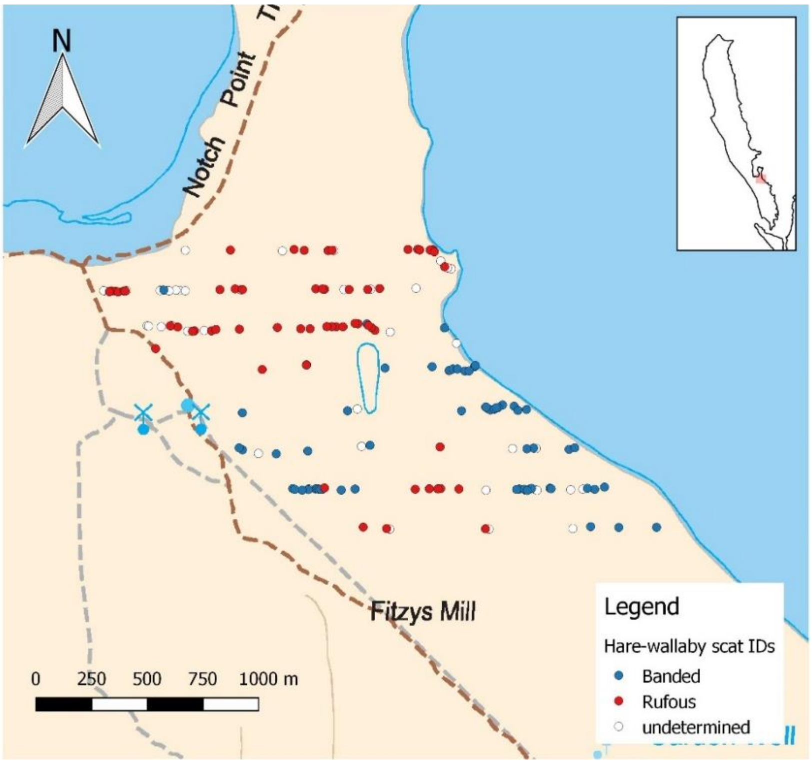
Appendix 3

Maps showing areas where hare-wallaby scat collections were made in November 2020. (a) Blowholes area; (b) Notch Point area; (c) Herald Bay area.

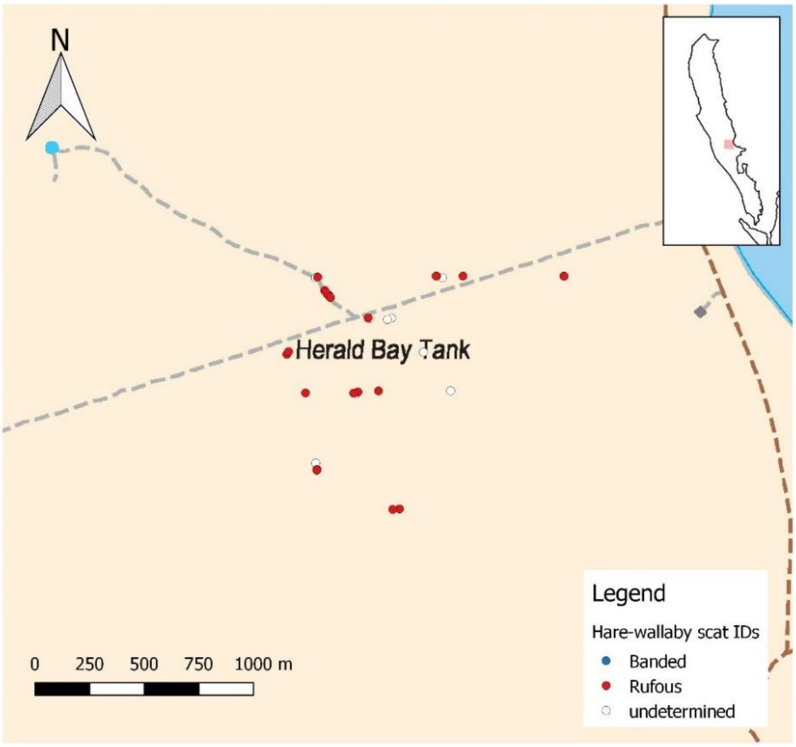
a)



b)



c)


N
Herald Bay Tank
0 250 500 750 1000 m
Legend
Hare-wallaby scat IDs
Banded
Rufous
undetermined

Appendix 4

Saunders, D., H. Nguyen, **S. Cowen**, M. Magrath, K. Marsh, S. Bell and J. Bobruk (2022). Radio-tracking wildlife with drones: a viewshed analysis quantifying survey coverage across diverse landscapes. *Wildlife Research* **49**, 1-10. doi.org/10.1071/WR21033

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Viewpoint

Radio-tracking wildlife with drones: a viewshed analysis quantifying survey coverage across diverse landscapes

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Abstract. Radio-tracking tagged wildlife remains a critical research technique for understanding the movements, behaviours and survival of many species. However, traditional hand-held tracking techniques on the ground are labour intensive and time consuming. Therefore, researchers are increasingly seeking new technologies to address these challenges, including drone radio-tracking receivers. Following the implementation of drone radio-tracking techniques for five different threatened species projects within different habitat and landscape types, we identified the need to quantify the relative spatial extent of surveys using both drone and hand-held techniques for each project. This was undertaken using viewshed analyses. These analyses demonstrated that survey coverage with drone-based radio-tracking was substantially greater than that of hand-held radio-tracking for all species and landscapes examined. Within mountainous landscapes, drone radio-tracking covered up to four times the area of hand-held tracking, whereas in flat to undulating landscapes, drone surveys covered up to 11.3 times the area that could be surveyed using hand-held techniques from the same locations on the ground. The viewshed analyses were also found to be a valuable visualisation tool for identifying areas for targeted surveys to reduce the risk of 'losing' tagged animals, which has traditionally been one of the biggest radio-tracking challenges.

Keywords: movement, drone, wildlife, radio-telemetry, localisation, unmanned aerial vehicle, VHF tag, threatened species.

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Introduction

Radio-tracking wildlife using very high frequency (VHF) radio-tags has enabled critical insights into the movement, behaviour and survival of many species for over 50 years (Slater 1962; Mech 1979). Despite advances in satellite and GPS tag technologies (Thomas *et al.* 2011), radio-tracking continues to play an integral role in wildlife research for a diversity of species, including both threatened (Finlayson *et al.* 2008; Gitzen *et al.* 2013; Robins *et al.* 2019) and invasive taxa (Ward-Fear *et al.* 2016; Burstal *et al.* 2020). Radio-tags remain the most appropriate or only feasible option for tracking small, highly mobile species, including 70% of birds and 65% of mammals (Bridge *et al.* 2011; Kays *et al.* 2015), given tags are typically limited to less than 5% of the animal's bodyweight (Murray and Fuller 2000; Barron *et al.* 2010). Radio-tracking is also valuable for studying larger species, including those with VHF implant tags (Robins *et al.* 2019) or GPS/satellite tags where real-time location information is required in addition to the data collected remotely. Such real-time information enables direct observations, health checks, animal recapture for tag replacement/removal, and retrieval of tags when the GPS/satellite component has failed (Matthews *et al.* 2013).

Traditional hand-held radio-tracking techniques are limited to tracking one individual animal at a time, and are labour intensive, given the requirement to hold a directional VHF antenna high in the air while traversing the study site (Mech 1979). Signals are often attenuated or reflected by landscape features (e.g. hills, rocks, gullies), or environmental factors

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Appendix 5

Cowen, S., M. Smith, S. McArthur, **K. Rayner**, C. Jackson, G. Anderson and **K. Ottewell** (2022). Novel Microsatellites and Investigation of Faecal DNA as a Non-Invasive Population Monitoring Tool for the Banded Hare-Wallaby. *Australian Journal of Zoology* **69**, 55-66. doi.org/10.1071/ZO21015

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Novel microsatellites and investigation of faecal DNA as a non-invasive population monitoring tool for the banded hare-wallaby (*Lagostrophus fasciatus*)

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ABSTRACT

Monitoring programs for populations of small or medium-sized animals often use live-capture or photo-monitoring trapping methods to estimate population size. The banded hare-wallaby (*Lagostrophus fasciatus*), a small macropodiform marsupial, does not readily enter traps or have individually unique distinguishing physical features and is consequently difficult to monitor using these methods. Isolating DNA from faecal material to obtain individual genotypes is a promising monitoring technique and may present an alternative approach for this species. We developed novel species-specific microsatellite markers and undertook trials to assess faecal DNA degradation in ambient environmental conditions at two locations where this species has been translocated. The quality of DNA yielded from faecal pellets was evaluated through amplification failure and genotyping error rates of microsatellite markers. Error rates were compared for different treatments and exposure duration across multiple individuals. DNA was successfully obtained from all samples and error rates increased with exposure duration, peaking after 14–30 days depending on the site and treatment. The level of solar exposure was the most significant factor affecting degradation rate but both this and exposure duration had significant effects on amplification failure. Analysing DNA obtained from faecal pellets may represent a practical non-invasive method of deriving population estimates for this species and warrants further development.

Keywords: conservation, faeces, hare-wallaby, *Lagostrophus*, minimally invasive, molecular genetics, monitoring, threatened species, wildlife management

Introduction

When it comes to managing threatened species, population size is a fundamental metric (Williams *et al.* 2002; Lindenmayer *et al.* 2012). Appropriate monitoring methods will vary according to the ecology and behaviour of the target species, as well as consideration of detection and cost-efficiencies (Garden *et al.* 2007). By monitoring the size of a population, conservation managers can assess the species' status relative to specific targets and management goals. Popular and robust methods of obtaining population estimates are capture–recapture models (Nichols 1992; Chao 2001; Efford and Fewster 2013). These methods rely on the ability to identify individual animals in a population across survey periods, after physical or non-physical (i.e. photographs from camera traps) capture (Jones *et al.* 1996; Chao 2001). For example, in Australia, live-capture surveys have been an effective method of monitoring a suite of taxa, including reptiles, frogs and small and medium-sized mammals and are identified as such in survey guidelines for threatened taxa drawn up by the Australian government (DEWHA 2010; DSEWPaC 2011*a*, 2011*b*). However, some species may not be readily lured to live-capture or remote camera traps and live-capture can sometimes pose serious risk to captured animals (Cole *et al.* 1994; Soulsbury *et al.* 2020).

Whilst camera traps may offer a practical, non-invasive alternative for some taxa (De Bondi *et al.* 2010), they have inherent limitations (Ballard *et al.* 2014; Meek *et al.* 2015)

Appendix 6

Wilson, L., **R. van Dongen**, **S. Cowen** and T. Robinson (2022). Mapping Restoration Activities on Dirk Hartog Island Using Remotely Piloted Aircraft Imagery. *Remote Sensing* **14**. doi.org/10.3390/rs14061402

remote sensing MDPI

Article

Mapping Restoration Activities on Dirk Hartog Island Using Remotely Piloted Aircraft Imagery

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Abstract: Conservation practitioners require cost-effective and repeatable remotely sensed data for assistive monitoring. This paper tests the ability of standard remotely piloted aircraft (DJI Phantom 4 Pro) imagery to discriminate between plant species in a rangeland environment. Flights were performed over two 0.3–0.4 ha exclusion plot sites, established as controls to protect vegetation from translocated animal disturbance on Dirk Hartog Island, Western Australia. Comparisons of discriminatory variables, classification potential, and optimal flight height were made between plot sites with different plant species diversity. We found reflectance bands and height variables to have high differentiation potential, whilst measures of texture were less useful for multisegmented plant canopies. Discrimination between species varied with omission errors ranging from 13 to 93%. Purposely resampling *c.* 5 mm imagery as captured at 20–25 m above terrain identified that a flight height of 120 m would improve capture efficiency in future surveys without hindering accuracy. Overall accuracy at a site with low species diversity ($n = 4$) was 70%, which is an encouraging result given the imagery is limited to visible spectral bands. With higher species diversity ($n = 10$), the accuracy reduced to 53%, although it is expected to improve with additional bands or grouping like species. Findings suggest that in rangeland environments with low species diversity, monitoring using a standard RPA is viable.

Keywords: species differentiation; remote sensing; UAV; monitoring and evaluation; rangelands

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1. Introduction

1.1. *Remotely Piloted Aircraft Environmental Monitoring*

Australian conservation monitoring is often conducted manually at the plot-scale [1]. Satellite remote sensing as an assistive dataset for testing plot-scale conservation efficacy has been largely untapped due to previous impediments, such as prohibitive costs or acquisition of capture with insufficient resolution. Remotely Piloted Aircrafts (RPAs) have begun to fill the void between open access satellite imagery with moderate resolution and very high resolution satellite imagery, too costly for repeat monitoring of vegetation changes at the plot-scale [2]. There is an urgent need for guidelines to assist conservation practitioners with best-practice RPA capture and subsequent processing [3].

Modern RPAs can capture imagery with subcentimetre spatial resolution [2,4] and have been used to monitor individual plants and grasses [5]. Differentiation of a single target species by timing imagery acquisitions with some distinguishing feature (e.g., flowers, leaf colour, defoliation) relative to coexisting species is important for tracking the trajectories of individual species (e.g., [6]), though it is less useful for studies that require

Appendix 7

Onley, I. R., L. C. White, K. E. Moseby, P. Copley and **S. Cowen** (2022). Disproportionate admixture improves reintroduction outcomes despite the use of low-diversity source populations: population viability analysis for a translocation of the greater stick-nest rat. *Animal Conservation*. doi.org/10.1111/acv.12812

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Disproportionate admixture improves reintroduction outcomes despite the use of low-diversity source populations: population viability analysis for a translocation of the greater stick-nest rat

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Keywords
conservation genetics; population viability analysis; reintroduction biology; admixture; translocation; *Leporillus conditor*; reintroduction; genetic diversity.

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Abstract

Translocation is becoming an increasingly important approach to threatened species conservation. Coupled with the knowledge that maximizing genetic diversity aids population establishment, the growing use of translocations can place unsustainable harvesting pressure on critical and vulnerable source populations. However, adaptive, genetically informed modelling tools such as Population Viability Analysis (PVA) can be used to predict translocation outcomes and optimize harvesting strategies. In this study, we use PVAs for the frequently translocated greater stick-nest rat (*Leporillus conditor*) to demonstrate the value of admixing founder populations for translocation, even when one source population is deemed genetically depauperate. This approach not only maximizes genetic diversity in the translocated population but reduces harvesting pressure on critical populations. Further, we show that admixed harvesting ratios can be skewed significantly towards the genetically depauperate population in order to further protect the critical population while still producing favourable outcomes, providing adequate founder numbers are used. As many threatened species are limited to fragmented and bottlenecked populations, these results are broadly applicable to the science of reintroduction biology, and demonstrate the value of PVAs for preliminary translocation planning and species management.

Introduction

Australia's biodiversity faces a growing number of threats associated with land use changes, habitat loss and climate change, and many conservation managers have employed the practice of translocation, the facilitated movement of a species from one area to another, to combat extinctions and secure populations (Seddon, 2010; IUCN, 2013). Translocation programs face a number of practical challenges both pre- and post-release, including funding shortages, monitoring difficulties, predation, poor habitat quality and lack of baseline knowledge (Clayton *et al.*, 2014; Short *et al.*, 2019; Berger-Tal, Blumstein, & Swaisgood, 2020). Translocation success may often rely on sufficient numbers of genetically diverse individuals. Low founder numbers are associated with high failure rates due to the increased likelihood of inbreeding and founder effects (Weeks *et al.*, 2011; McCoy *et al.*, 2014; Pacioni, Wayne, & Page, 2019). Similarly, low genetic diversity (either from founders or due to founder effect/post-release bottlenecks) also places translocations at risk of inbreeding depression or a lack of adaptive potential (Jamieson, 2011; Biebach & Keller, 2012; Ramstad *et al.*, 2013; Murphy *et al.*, 2019).

One of the guiding principles of translocations is to ensure that the source population is not negatively impacted by harvesting (IUCN, 2013). The increasing use of translocation programs combined with the importance of maximizing genetic diversity for population establishment and persistence means that source populations are under more pressure for conservation reintroductions (Armstrong & Seddon, 2008; Jamieson & Lacy, 2012; IUCN, 2013; Schäfer *et al.*, 2020). As many threatened species have already suffered genetic

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