Evaluation of camera trap sampling designs used to determine change in occupancy rate and abundance of feral cats

Alan Robley, Andrew Gormley, Luke Woodford, Michael Lindeman, Bernard Whitehead, Ray Albert, Michael Bowd and Aileen Smith

2010

Evaluation of camera trap sampling designs used to determine change in occupancy rate and abundance of feral cats

Alan Robley, Andrew Gormley, Luke Woodford, Michael Lindeman, Bernard Whitehead, Ray Albert, Michael Bowd and Aileen Smith

Front cover photo: Feral cat captured on camera at Anglesea (Alan Robley)

Contents

List of tables and figures

Acknowledgements

Summary

Although many agencies and organisations commit resources to managing feral cats, there remains a high degree of uncertainty about our ability to accurately and precisely estimate the relative or absolute abundance of feral cats, or the kill rates obtained in control operations.

In 2007 the Commonwealth Department of Environment, Water, Heritage and the Arts (DEWHA) commissioned the Arthur Rylah Institute for Environmental Research (ARIER) to undertake research to evaluate detection methods and sampling designs used to determine the abundance of feral cats (*Felis catus*) that have established wild populations in Australia. Stage one of the project, undertaken at Anglesea in Victoria in September 2007, investigated the detection probability of cage traps, leghold traps, a DNA sampling device, heat-in-motion activated digital cameras and sand plots (Robley et al. 2008). This required the capture of feral cats, fitting GPS data-logging collars, mapping their areas of activity and monitoring their encounter with the various devices. Digital cameras provided the most robust detection data for feral cats. Modelling of control efficacy determined that the probability of correctly concluding that a cat population had decreased was highest for the highest level of monitoring (49 cameras for 20 nights, set at one camera per 2 km^2).

An attempt was made to validate the use of heat-in-motion digital cameras in the Grampians National Park in central western Victoria in February 2008, but no feral cats were captured. A second attempt was made to validate the approach at Anglesea in March 2008, but only one feral cat was captured. Further development of the protocol was attempted as part of a research project investigating the interactions between wild dogs (*Canis familiaris*), Red Foxes (*Vulpes vulpes*), feral cats and Spot-tailed Quolls (*Dasyurus maculatus*) in Tuggolo State Forest, New South Wales (Robley 2009a). But although six feral cats were captured, collared and released, none were detected on the digital cameras.

In July 2008 in south-western Victoria we assessed two different spatial configurations of cameras as part of the further development of the camera monitoring protocol. In the first configuration, cameras were set in pairs at selected sites. In the second configuration, one camera was set but at more sites. Using the first configuration, we compared site occupancy rates of feral cats where the number of foxes had been reduced (using FoxOff® baiting over the previous five years) to a paired site where no fox control had taken place (Robley et al. 2009b). We applied the second configuration to assess differences in occupancy of feral cats before and after a control operation on the site with the highest level of feral cat occupancy.

We used a Bayesian modelling approach and a mark–recapture model to assess whether there was a difference in levels of site occupancy and abundance of feral cats between sites where foxes had been reduced compared to a site where they had not. We used only the Bayesian modelling approach to assess differences before and after control as insufficient feral cats were detected to undertake a mark-recapture estimate during this phase.

Both camera configurations provided robust estimates of occupancy (ψ) for feral cats. We were able to assess a difference in occupancy rate between sites with a history of fox control ($\psi = 0.69 \pm 0.10$ SE) and without ($\psi = 0.50 \pm 0.13$ SE), but not before ($\psi = 0.53 \pm 0.129$ SE) and after ($\psi = 0.45 \pm 0.137$ SE) a lethal cat control operation (because only four cats were killed). We estimated that, for the given sampling effort (1 camera at 49 sites for 28 days), we would have only been able to measure a significant change had there been a reduction in occupancy of 44% — much more than was achieved by the control operation (15%).

The site-level daily detection probabilities for the paired configuration ($p = 0.077$) and the single configuration ($p = 0.060$) were similar, but a comparison between two different brands of camera

and other cameras showed that detection probabilities varied markedly. There was also a relationship between camera and bait-holder type: wire cage bait-holders with Reconyx cameras had a high detection rate. For a daily detection probability of 0.05, the post control occupancy rate would not have to exceed 0.3 to detect a change with 80% power.

Uncertainty about the estimated rate of occupancy is related to detection probability, the level of occupancy itself, and the number of sites sampled. We used the coefficient of variation to assess the difference in precision of occupancy rates. While the overall detection probability at 22 sites with two cameras set for 21 days was higher (81%), than at the 49 sites with a single camera set for 28 days (73%), the occupancy estimate was more precise with a greater number of sites.

By comparing the two camera configurations and using different camera and bait holder types, we have shown that the camera protocol is flexible and — importantly — that care needs to be taken in the application of this tool in the field. Camera performance is a critical factor, as well as the physical set-up of the camera in the field. The monitoring protocol is able to detect changes in the abundance of feral cats resulting from control poerations, however the ability to kill cats is limited by the currently available tools, such as cage trapping, which is labour intensive and time consuming.

We have demonstrated that the combination of heat-in-motion activated digital cameras and statistical techniques that have been developed for the unbiased estimation of rates of occupancy under conditions of imperfect detection (MacKenzie et al. 2002, Tyre et al. 2003) can robustly determine differences in feral cat populations in temperate eucalypt woodlands and forests. We have also demonstrated that digital cameras can be used to identify individual cats, so that feral cat abundance can be determined. These methods can be used to assess changes in feral cat populations over time, and to assess the success of feral cat control operations.

1 Introduction

Feral cats (*Felix catus*) probably became established in Australia soon after the arrival of Europeans, and wild populations now occupy most parts of the mainland, Tasmania and some offshore islands (Abbott 2002). Cats kill a wide range of native wildlife (reviewed in Robley et al. 2004), and for this reason are thought to reduce the distribution and abundance of many native species, especially on islands. Predation by feral cats is listed as a known or perceived threatening process for 58 native species under the Australian Government's *Environment Protection and Biodiversity Conservation Act 1999*.

Although many agencies and organisations commit resources to managing feral cats (Reddiex et al. 2004), there remains a high degree of uncertainly about the ability to accurately and precisely estimate the relative or absolute abundance of feral cats, or the kill rates obtained in control operations. This limits the ability of policy-makers and managers to judge the effectiveness of control operations or justify the investment of resources in further operations.

Determining the success of feral cat control operations is problematic. Commonly used monitoring techniques (e.g., spotlighting, scat counts and sand-plot monitoring) for indexing changes in populations of other carnivores such as Red Foxes (*Vulpes vulpes*) or Dingos (*Canis lupus dingo*) are either inaccurate or limited (in their application to feral cats) to islands or sandy environments (Mahon et al. 1998, Edwards et al. 2000, Saunders and McLeod 2007). Feral cats tend not to look towards spotlights and do not preferentially use roads or tracks (Mahon et al. 1998, Edwards et al. 2000). The proportion of toxic baits taken is also commonly used to report the success of fox and wild dog control operations. The cues used by cats to locate food are visual and auditory rather than olfactory, so buried baits are seldom taken by a feral cat, which significantly limits the use of bait take in monitoring feral cats in eastern Australia where these baits must be buried (Seebeck and Clunie 1997, DEWHA 2008).

In 2007 the Commonwealth Department of Environment, Water, Heritage and the Arts (DEWHA) commissioned the Arthur Rylah Institute for Environmental Research (ARIER) to undertake research to evaluate detection methods and sampling designs used to determine the abundance of feral cats that have established wild populations in Australia. Stage one of the project was undertaken at Anglesea, Victoria, in September 2007 and investigated the detection probability of cage traps, leghold traps, a DNA sampling device, heat-in-motion activated digital cameras and sand plots (Robley et al. 2008). This required feral cats to be captured, fitted with GPS data-logger collars and monitor their interactions with the various devices. Digital cameras provided the most robust detection data for feral cats, with a per camera daily detection probability of 0.013. Data collected from digital cameras at Anglesea was used to simulate various monitoring intensities and their ability to correctly identify a 50% decrease in cat abundance, i.e. the efficacy of control. Of the nine monitoring intensities modelled, all indicated a post-control decrease in cat abundance. The potential to correctly conclude that the cat population had decreased was highest for the highest level of monitoring (49 cameras for 20 nights set at one camera per 2 km^2), which was correct 85% of the time. In February 2008 an attempt was made to validate this approach in the Grampians National Park in central-western Victoria, where habitat and local knowledge suggested that feral cat abundance was sufficient to expect the capture of feral cats. However, despite 945 trap nights, no feral cats were caught. A second attempt was made to validate the approach at Anglesea in March 2008 (852 trap nights), but only one feral cat was trapped.

In May 2008 the project continued with the aim of firstly continuing the development of the camera monitoring protocol, and secondly to assess its ability to measure a change in cat abundance following a control operation.

Further development of the protocol was attempted in collaboration with a New South Wales Department of Primary Industries research project investigating the interactions between wild dogs (*Canis lupus familiaris / Canis lupus dingo*), foxes, feral cats and Spotted-tailed Quolls (*Dasyurus maculatus*) in Tuggolo State Forest, approximately 150 km south-east of Armidale, New South Wales. A combination of leghold and cage traps were set continuously for 22 days (755 leghold trap nights and 220 cage trap nights), resulting in six feral cats being caught. Following the capture, collaring (with GPS-data loggers) and release of the feral cats, 39 heat-in-motion cameras were deployed for an average of 28 days at a density of one camera per 1.7 km^2. The variation in this from the protocol resulted from limited access in rugged terrain. No feral cats were detected on the digital cameras in that study. Further details are provided in Robley (2009a).

This report outlines the results of a further study undertaken to assess how detection probabilities vary with different camera types and spatial arrangements' of cameras. In south-western Victoria we assessed two different spatial configurations of cameras: (1) cameras set in pairs over fewer sites, and (2) a single camera set at a greater number of sites. We first compared site occupancy of feral cats at a site where foxes had been reduced using FoxOff® baiting over the previous five years to a paired site where no fox control had taken place (Robley et al. 2009b). We hypothesised that feral cat occupancy rates would be higher on sites with a reduced number of foxes. Having estimated the occupancy rate of feral cats at these sites, we applied the second camera configuration to assess differences in occupancy rates before and after a control operation on the site with the highest rate of feral cat occupancy.

We report on the results of these activities and provide recommendations for applying the camera monitoring protocol.

2 Methods

2.1 Study site

The study site was located in south-western Victoria (141°15′39″E, 38°07′39″S) between the township of Portland and the Victoria – South Australia border, and within the Glenelg Ark Project area (Fig. 1). The Glenelg Ark project is a large-scale, continuous fox baiting initiative using buried FoxOff® baits across 100 000 ha of state forest and national park. Within this area are three monitoring sites that receive continuous fox control (treatment), and three areas that receive no fox control (non-treatment). We conducted this project in one of these paired sites, the Lower Glenelg National Park – south (LGNP-south, 8954 ha; treatment) and north (LGNP-north, 4659 ha; non-treatment) which are physically separated by the Glenelg River.
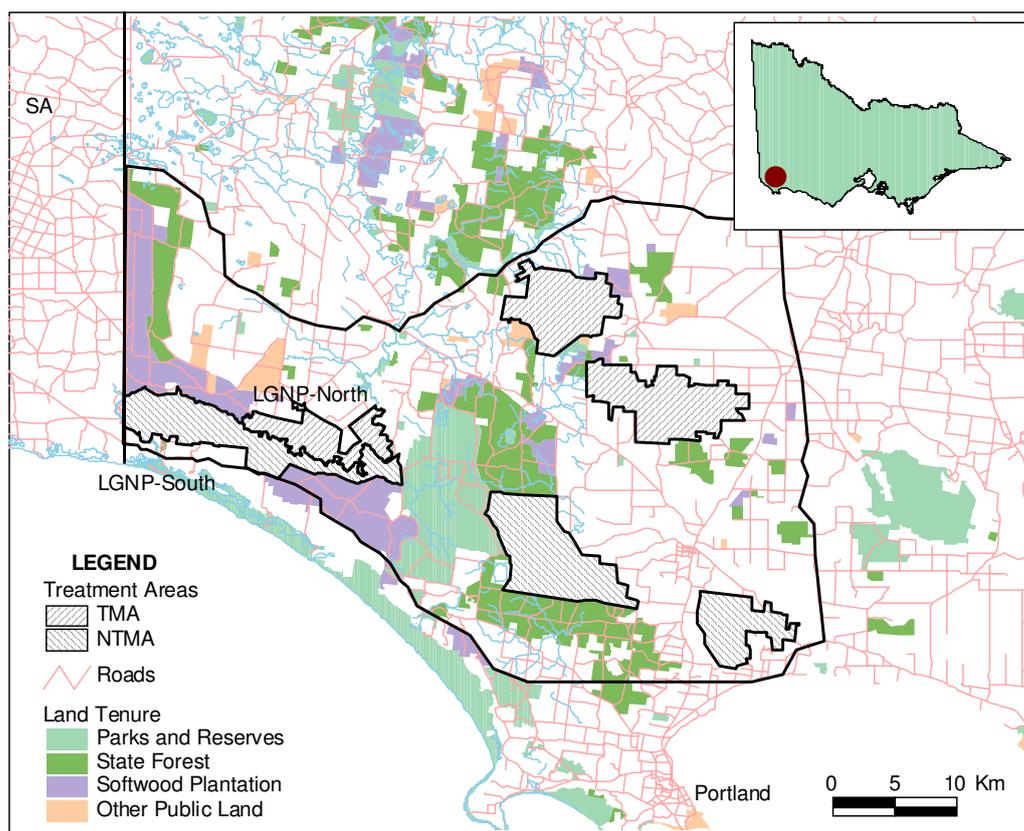
Figure 1. Location of the Lower Glenelg National Park-south and north sites used in this study.
TMA – Treatment (fox control) Monitoring Areas, NTMA – Non-treatment Monitoring Areas.
Solid black line – outline of Glenelg Ark project area.

2.2 Paired camera surveys, 2008

We compared occupancy rates for feral cats at the Lower Glenelg National Park – south and north sites in July 2008 using heat-in-motion digital cameras (Fig. 2). A pool of 72 cameras was available: 35 Reconyx RapidFire ProPC90s[1] and 37 TrailMacs[2]. In LGNP-south we deployed 42 cameras (19 Reconyx and 23 TrailMac) at 22 sites and in LGNP-north we deployed 30 cameras (14 Reconyx and 16 TrailMac) at 15 sites (Fig. 2). Two cameras of the same type were located

---

[1] Reconyx, Holmen, Wisconsin, USA
[2] Trailsense Engineering, Middletown, Delaware, USA

within a randomly selected 1 km^2 grid as insurance against a single camera failure. Cameras were operated for an average of 21 days.

Reconyx cameras record 3.1 mega-pixel colour images during daylight and 3.1 mega-pixel infra-red images at night. These cameras were set to record three images as fast as possible once motion was detected and to keep recording images until motion ceased. TrailMac game trail cameras recorded a six mega-pixel colour image and use a white flash according to lighting conditions. Images are recorded 30 seconds after motion is detected until motion ceases. All images were time and date stamped. A small wire cage (15 × 15 cm) containing a fresh chicken maryland, 1 or 2 pilchards and tuna oil was fixed to the top of a 1.5 m stake and placed 3 m in front of each camera. Where necessary, vegetation was cleared from the field of view of the camera to reduce false triggering of the camera and improve image identification.
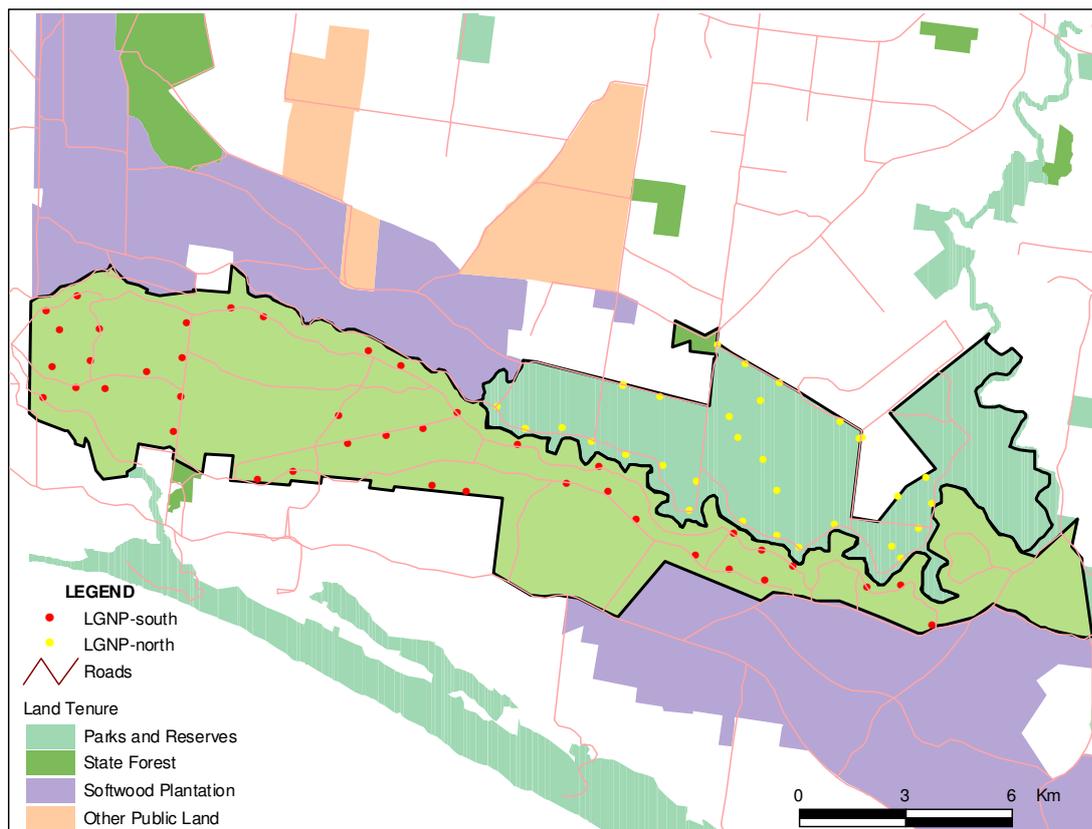
Figure 2. Location of 72 paired cameras at LGNP-south (*n* = 42) and LGNP-north (*n* = 30).

2.3 Single camera surveys, 2009

Before-control occupancy surveys were conducted at LGNP-south in September 2009 and after-control occupancy surveys in November 2009. Either a single Reconyx camera (*n* = 20) or a single Scout Guard SG550V[3] camera (*n* = 29) was placed at a randomly selected location within a 2 km^2 grid (Fig. 3). Cameras were operated on average for 28 days.

---

[3] HCO, Norcross, Georgia, USA
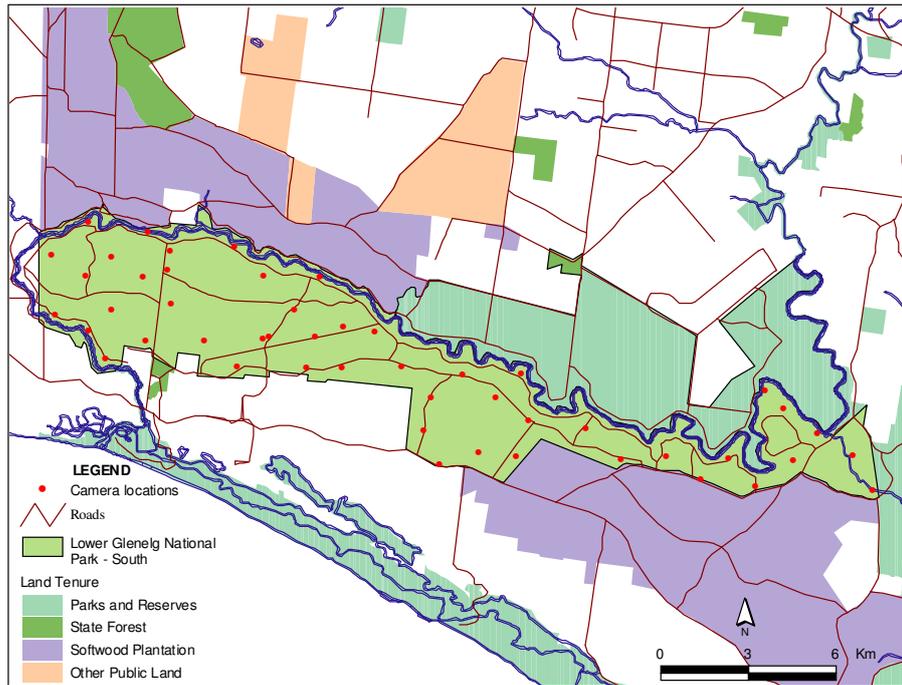
Figure 3. Location of 49 single cameras set up at LGNP-south before and after feral cat control.

Reconyx cameras operated as described above. Scout Guard cameras recorded a 3.1 megapixel colour image during the day and a 3.1 megapixel infra-red image at night. Images were recorded within two seconds of motion being detected, and recording continued until motion ceased. All images were time and date stamped.

Two bait holder types were used in this survey — either a small wire cage as used in 2008, or a PVC pipe (25 cm long × 90 mm diameter) capped at both ends and with numerous holes drilled into the sides (Fig. 4a, b). Each bait holder contained a fresh chicken maryland, 1 or 2 pilchards and some tuna oil, and was fixed to the top of a 1.5 m stake placed 3 m in front of each camera.

Figure 4. Bait holders used in this study: (a) small wire cage, (b) PVC tube.

2.4   Feral cat control, LGNP-south

A professional pest control contractor (Outfoxed Pest Control, Ivanhoe, Victoria) was employed to trap feral cats in October 2009 from LGNP-south. Seventy-two Victor number 1.5 rubber jaw traps were set along roads and tracks throughout LGNP-south (Fig. 5). Traps were operated continuously for 20 days, giving 1368 trap nights. Traps were checked every 24 hours and reset after rain or being set off by target or non-target animals. Captured feral cats were euthenised at the point of capture. Each cat had its weight, sex, and reproductive condition recorded.
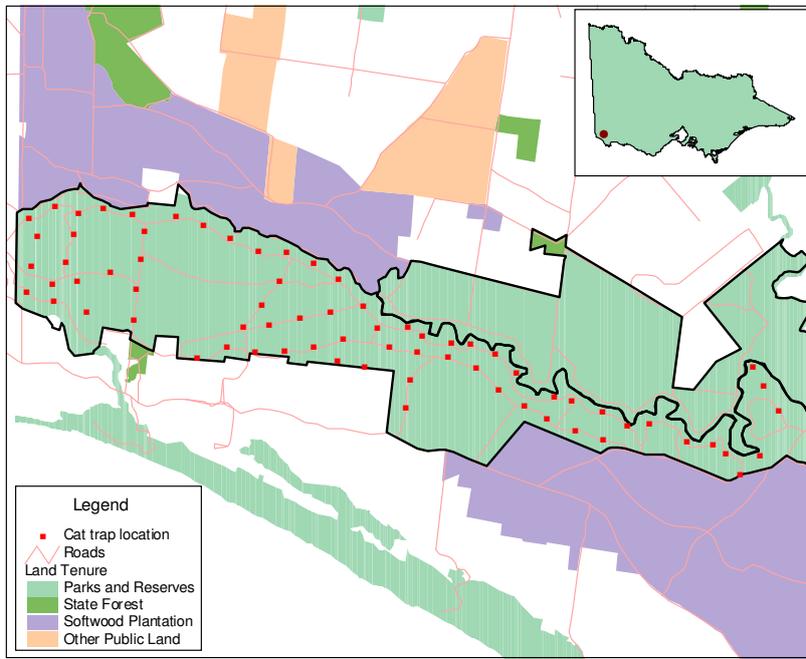
Figure 5. Location of Victor 1.5 soft jaw traps set in LGNP-south in October 2009.

2.5   Occupancy estimation

We used the proportion of camera sites within a monitoring area (i.e. LGNP-south and LGNP-north) that were occupied to assess the differences on occupancy rates. The phrase 'occupancy' is used here to mean the proportion of sampling units that contain feral cats at a given point in time.

Data from the camera surveys were summarised into detection histories for each site, where $y_{ij} = 1$ indicated that the species was detected at site i on day j, and $y_{ij} = 0$ otherwise. For example, for a 5 day sampling program, the entire history for a particular site $y_i = [1,0,0,1,1]$ would indicate that cats were detected on the 1st, 4th and 5th day but not on the 2nd or 3rd day.

It is typical of presence/absence data that a species may be present on a site, yet not be detected on a given day. For example, a detection history such as $y_i = [0,0,0,0,0]$ could indicate that the species was either absent from site i, or it was present but not detected on any of the days. We used the occupancy modelling approach outlined by MacKenzie et al. (2006) to estimate whether non-detections were actually true absences. This approach allows us to remove the confounding between the occupancy rates and the detection probability.

Assuming the daily detection probability (Pr) is constant, the overall probability of each detection history can be calculated as:

$$\Pr\left(y_i \mid z_i^{\mathrm{obs}} = 1\right) = \psi p^n (1-p)^{T-n}$$

$$\Pr\left(y_i \mid z_i^{\mathrm{obs}} = 0\right) = (1-\psi) + \psi(1-p)^T$$

where ψ is the occupancy probability, p is the detection probability, n is the number of days that the species was detected, T is the total number of days cameras were deployed at the site, and $z_i^{\text{obs}} = 1$ indicates that the species of interest was detected at least once on site i, and $z_i^{\text{obs}} = 0$ otherwise.

In order to asses whether the monitoring protocol could detect a difference between LGNP-south and LGNP-north, a model was specified that allowed occupancy to differ between areas. Furthermore, the model was specified so that detection probabilities were constant each day and between areas, but could differ between camera types. Note that occupancy is assumed to be constant in each area over the monitoring period (i.e. 21 days in July 2008).

In analysing the single camera survey, six occupancy models were specified. Occupancy probabilities could differ between LGNP-south and LGNP-north, but were assumed to be constant for the duration of each survey (i.e. 30 days in September 2009 and 21 days in November 2009).

The six models specified were:

- occupancy and detection constant
- occupancy constant and detection varies with camera type
- occupancy constant, and detection varies with camera and bait holder combination
- occupancy varies with survey period, and detection varies with camera type
- occupancy varies with survey period, and detection varies with camera and bait holder
- occupancy varies with survey period, and detection varies with camera and bait holder and period

Occupancy models were fitted using WinBUGS 1.4.3 (Lunn et al. 2000), a program for deriving Bayesian parameter estimates using Monte Carlo Markov chain updaters. A Markov chain was run for 5000 iterations in order to check for convergence and to ensure each parameter had reached a stable distribution. These 'burn-in' samples were discarded and the algorithm was run for a further 40 000 iterations. The values from each iteration comprised the posterior distribution for each of the parameters. The posterior distributions were summarised in terms of parameter means and standard deviations (SD).

Competing models that were used to assess occupancy before and after control in 2009 were assessed using the deviance information criterion (DIC). The model with the lowest DIC is the one that is the most parsimonious model, i.e. the one that gives the best balance between precision and bias. The DIC is given as $\text{DIC} = \overline{D} + pD$, where $\overline{D}$ is the posterior mean of the deviance and pD is the effective number of parameters, given by $pD = \overline{D} - \hat{D}$ (Spiegelhalter et al. 2002). $\hat{D}$ is a point estimate of the deviance evaluated at the posterior means of the parameters. DIC can be rewritten as:

$$\text{DIC} = 2\overline{D} - \hat{D}$$

In general WinBUGS returns the DIC directly, but for models constructed from a mixture of distributions, such as the one used here, this is not the case. However, the deviance is able to be monitored, which allows the calculation of DIC for each model using a two-step process. Firstly, the estimation model is run with the mean deviance giving $\overline{D}$. The model is then re-run with the mean parameter estimates plugged in as data. The mean of the deviance in this case gives $\hat{D}$.

The probability that site occupancy was higher on the LGNP-south site or before feral cat control was determined by calculating occupancy at LGNP-south minus LGNP-north, and occupancy before minus after control, at each iteration of the WinBUGS updating process. The proportion of the posterior density of this difference that is greater than zero gives the probability that site occupancy is higher.

2.6 Power analysis

The power of a statistical test is the probability of not falsely concluding that an outcome has occurred when it has not; in this case, that the cat population has decreased when in fact it has not. As the power of a test increases, the chance of making this error decreases. A power analysis was carried out to determine what level of power we had to detect an actual difference in occupancy rate following a cat control operation. We based the power analysis on the study design in 2009, i.e. cameras at 49 sites with each camera set for 28 days. The pre-control occupancy rate was held constant at 0.53, whilst the post-control occupancy rate was set at various levels between 0.1 and 0.5, representing various levels of control efficacy. For each level we randomly generated the site occupancy and the subsequent detection history based on a daily detection probability of 0.05. The occupancy rate was estimated from the resulting data, along with its corresponding confidence interval. When the confidence interval minus the pre-control occupancy rate excluded zero, this indicated that a significant difference would have been detected. This process was repeated 1000 times, with the proportion of times that a difference was detected used as the power to detect that difference. This was repeated for the different levels of occupancy rate, and at different levels of daily detection probability (0.02 and 0.1).

Uncertainty around the estimated rate of occupancy is related to detection probability, the level of occupancy itself, and the number of sites. We used the co-efficient of variation (the ratio of the standard deviation of the estimate to its mean) to assess the difference in precision of occupancy rates (0.3, 0.5, 0.7 and 0.9) over a range of detection probabilities (0.2 to 1.0) for more and less sites.

2.7 Feral cat abundance

We assessed feral cat abundance in 2008 at LGNP-south and LGNP-north by indentifying individuals based on multiple images of feral cats captured by cameras and noting coat colouration, markings, and banding. Detections of cats at camera sites were included in the analysis only if multiple images were recorded showing both flanks, the head and the back of the cat.

Coat colouration was divided into four basic types, black, ginger, and tabby, with tortoise shell included as a subset of tabby. Tabbies were further divided into five types of coat pattern, each possessing unique markings. Individuals within a coat type, eg classic tabby, could be identified from individual patterns from multiple images.

A 'classic tabby' has bold, swirling patterns on its sides. The pattern of circular blotches (or smudges) on the body closely resembles a bullseye. A 'mackerel tabby' has narrow stripes running down the sides in a vertical pattern. The stripes are unbroken lines; evenly spaced, which branch out from one stripe that runs along the top of the cat's back down the spine, resembling a fish skeleton. A 'spotted tabby' has spots all over its sides. These spots can be large or small, and sometimes appear to be broken mackerel stripes. They can be round, oval or rosettes. A 'ticked tabby' (sometimes called Abyssinian tabby or agouti tabby) does not have the traditional stripes or spots on its body. However, like all tabbies, this coat pattern has tabby markings on the face and agouti hairs on the body. A 'tortoise-shell tabby' has separate patches of brown tabby and red tabby on the same animal. Tortoise-shell tabbies can show any one of the above four distinct tabby patterns. The markings are usually more apparent on the legs and head.

For each site survey a capture history matrix X was constructed where $X_{ij} = 1$ indicates capture of cat i during period j, and $X_{ij} = 0$ denotes non-capture. All individuals in the population were assumed to be present throughout the duration of the survey, i.e. the population was assumed to be closed to births, deaths and migration. A broad class of capture-recapture models, called closed population models, can be used to analyse this type of data (Otis et al. 1978). The simplest model (M_0) assumes that each individual has a constant capture probability across time periods that is independent of whether it has been captured before. General models exist that allow capture probabilities to differ with time (M_t), with a behavioural response to capture (M_b), with the individual cat because of heterogeneous capture probabilities (M_h), or combinations of these factors (M_{th}, M_{tb}, M_{bh}, M_{tbh}). We used the method of finite mixtures that allows for individuals to belong to one of the eight classes of capture probabilities listed above (Pledger 2000).

3   Results

3.1   Difference in occupancy rate — paired cameras, 2008

While primarily focused on the ability of the camera monitoring protocol to assess differences in the occupancy rate of feral cats between LGNP-south and LGNP-north, we also included six native species and the red fox as we had sufficient detections to undertake the analyses.

The monitoring protocol detected a difference in occupancy rate of feral cats between the two sites. The estimated probability of occupancy for feral cats was greater at LGNP-south ($\psi = 0.69 \pm 0.10$ SE) compared to LGNP-north ($\psi = 0.50 \pm 0.13$ SE) during 2008. The probability that occupancy rates were higher at LGNP-south was 0.88 (i.e. 88%; Table 1) suggesting that the cat population was more prevalent at LGNP-south.

The monitoring protocol also estimated the probability of occupancy for foxes at LGNP-south ($\psi = 0.24 \pm 0.17$ SE) as much lower than that for LGNP-north ($\psi = 0.60 \pm 0.21$ SE; Table 1). The probability that site occupancy was higher at LGNP-south was very low for foxes (0.04 or 4%), suggesting that foxes are significantly more common in LGNP-north.

Southern Brown Bandicoots (*Isoodon obesulus*) were significantly more widespread in LGNP-south, whereas Red-necked Wallabies (*Macropus rufogriseus*) and Common Ringtail Possums (*Pseudocheirus peregrinus*) were significantly more common in LGNP-north (Table 1). The results from other species were more equivocal.

Table 1. Estimated probability of site occupancy for species detected at LGNP-south and LGNP-north sites and the probability that site occupancy is higher in the treated site. SE shown in parentheses.

Species	LGNP-south	LGNP-north	Pr(south > north)
Feral Cat	0.69 (0.10)	0.50 (0.13)	0.88
Southern Brown Bandicoot	0.67 (0.18)	0.15 (0.11)	0.99
Swamp Wallaby	0.57 (0.16)	0.50 (0.17)	0.65
Common Brushtail Possum	0.58 (0.08)	0.56 (0.09)	0.59
Western Grey Kangaroo	0.34 (0.11)	0.43 (0.13)	0.27
Red Fox	0.24 (0.17)	0.60 (0.21)	0.04
Red-necked Wallaby	0.38 (0.11)	0.71 (0.13)	0.02
Common Ringtail Possum	0.13 (0.11)	0.52 (0.21)	0.01

3.2   Difference in occupancy rate — single camera, 2009

In October 2009 we removed two female and two male feral cats from the LGNP–south after 1368 trap nights - acapture rate of 0.3 cats per 100 trap nights (Table 2).

Of the six occupancy models tested for the 2009 data, the most parsimonious model was for constant occupancy ($\psi = 0.336$) and constant detection probability ($p = 0.045$) before and after cat control (Table 3).

Table 2. Details of feral cats removed during the control operation.

Date	Sex	Weight (kg)
11/10/09	F	5.0
14/10/09	F	3.0
15/10/09	M	3.5
25/10/09	M	5.0

Table 3. Details of the model outcomes ranked according to lowest DIC.

Model	Description	Parameters	DIC
$\psi(.)$, $p(.)$	Occupancy and detection constant	2	320.5
$\psi(.)$, p(camera)	Detection varies with camera type	3	331.5
$\psi(.)$, p(camera+bait holder)	Detection varies with camera and bait holder	4	332.6
ψ(pd), p(camera)	Occupancy varies with period and detection varies with camera	4	333.1
ψ(pd), p(camera+bait holder)	Occupancy varies with period and detection varies with camera and bait holder	5	337.8
ψ(pd), p(camera+bait holder +pd)	Occupancy varies with period and detection varies with camera and bait holder and period	6	341.2

The highest ranked model that allowed for a difference between before and after control occupancy rates was model ψ(pd), p(camera), with estimates of occupancy before of $\psi = 0.53 \pm 0.129$ SE and after of $\psi = 0.45 \pm 0.137$ SE. Based on this model, there was a 68% chance of a decrease in occupancy by feral cats following control (Fig. 6). However, the 95% credible limits of the treatment effect overlap zero indicating that a significant effect of treatment was not detectable; .e. removing four individuals was not enough to show a significance difference in occupancy rates.
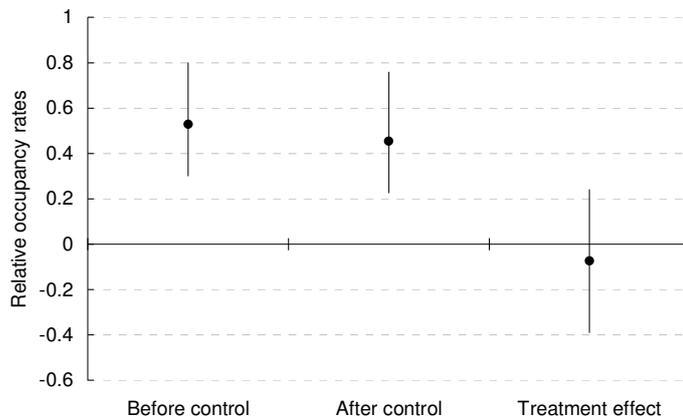
Figure 6. Occupancy rate for feral cats before and after control, and the treatment effect of control on occupancy from model ψ(pd)p(camera). Bars are 95% credible intervals.

3.3 Detection probabilities for occupancy modelling

3.3.1 Paired cameras, 2008

The site level daily detection probabilities for feral cats per site in 2008 differed between Reconyx cameras ($p = 0.077$) and TrailMac cameras ($p = 0.054$). These daily detection probabilities translate into an 81% and 69% chance of at least one feral cat at an occupied site being detected over 21 days for Reconyx and TrailMac cameras respectively (Fig. 7).
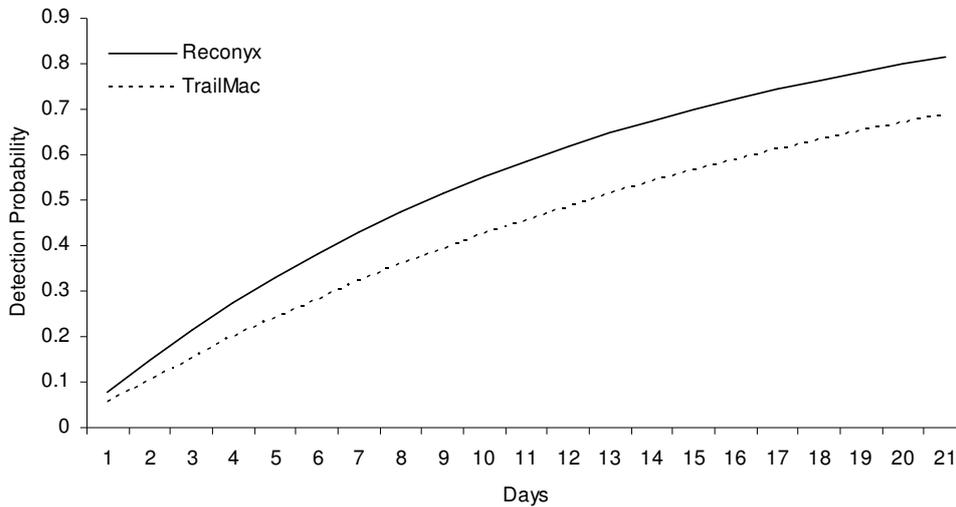
Figure 7. Cumulative mean detection probabilities for feral cats in 2008 using Reconyx (solid line) and TrailMac (dashed line) cameras.

3.3.2 Single camera, 2009

The site-level daily detection probabilities for feral cats per site in 2009 differed significantly between Reconyx cameras ($p = 0.060 \pm 0.018$ SE) and Scout Guard cameras ($p = 0.017 \pm 0.029$ SE). After 28 days these daily probabilities correspond to cumulative detection probabilities of 82% and 37% for Reconyx and Scout Guard respectively (Fig. 8).
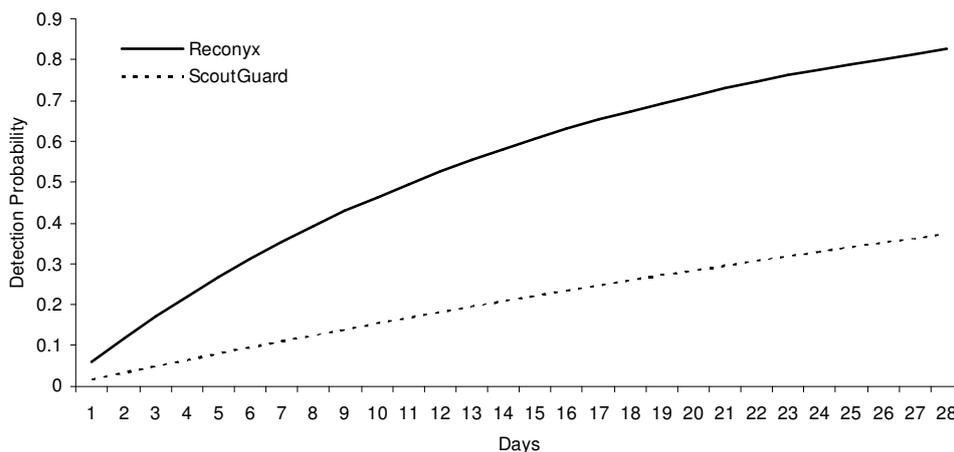
Figure 8. Cumulative mean detection probabilities for feral cats at LGNP-south in 2009 using Reconyx (solid line) and Scout Guard (dashed line) cameras

The highest ranked model that considered bait-holder was model *psi*(.)*p*(camera+bait holder) (Table 4). From this model, Reconyx cameras had a higher detection probability than Scout Guard cameras. There was an effect of bait-holder type, with wire cages having a higher detection probability than PVC pipe. The highest combination of daily detection probability was from a combination of Reconyx cameras and wire cage (Table 4).

Table 4. Detection probabilities in 2009 by combinations of camera and bait-holder type from model *psi*(.)*p*(camera+bait holder) SE shown in parenthesis.

Camera	Bait holder	Detection probability
Reconyx	wire cage	0.071 (0.018)
Reconyx	PVC	0.043 (0.016)
Scout Guard	wire cage	0.019 (0.010)
Scout Guard	PVC	0.011 (0.005)

3.4   Power analysis

For each level of daily detection probability (0.10, 0.05, 0.01) the power to detect a reduction in occupancy increases as the difference between pre-control and post-control increased. Higher detection probabilities had an increasing power to detect a difference (Fig. 9).

In the 2009 study at LGNP-south the initial pre-control occupancy rate was 0.53, and following control it was 0.45, although this difference was not statistically significant (Fig. 6). Based on the power analysis, there was only a 20% chance of detecting a significant decrease in occupancy as measured after the cat control operation.
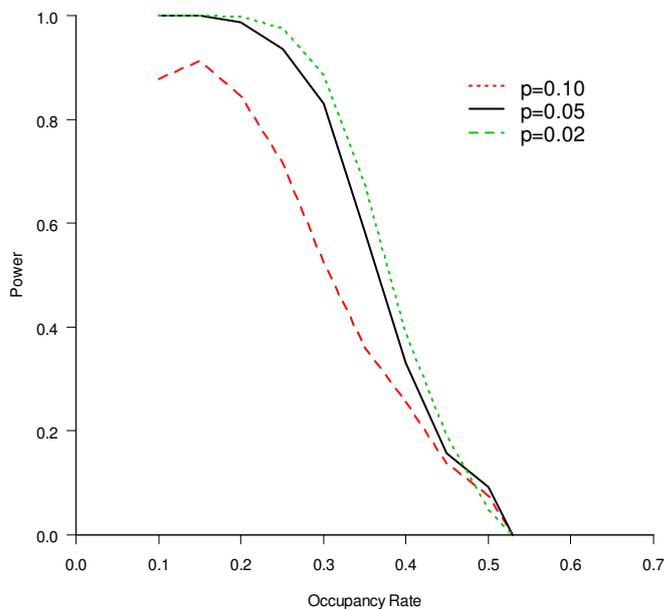
Figure 9. Results of the power analysis for detecting change in occupancy rate for feral cat control.

The results of the power analysis indicate that, for an average daily detection probability of 0.05, post-control occupancy rates needed to be reduced to (at most) 0.3 in order to have an 80% power of detecting a significant difference.

The uncertainty around the estimated rate of occupancy is related to detection probability, the level of occupancy itself, and the number of sites. An exploration of the precision of occupancy estimates (i.e. lower CV) showed that precision improves with higher detection probability, as well as with higher occupancy rate. However, the precision markedly improves with an increase in the number of sites from 22 to 49 (Fig. 10), regardless of the number of cameras set at a site. For example, for an overall detection probability of 0.8 and an occupancy rate of 0.5 the CV of occupancy for 22 sites is 0.26 and for 49 sites 0.17. This represents an increase in precision of 33%.
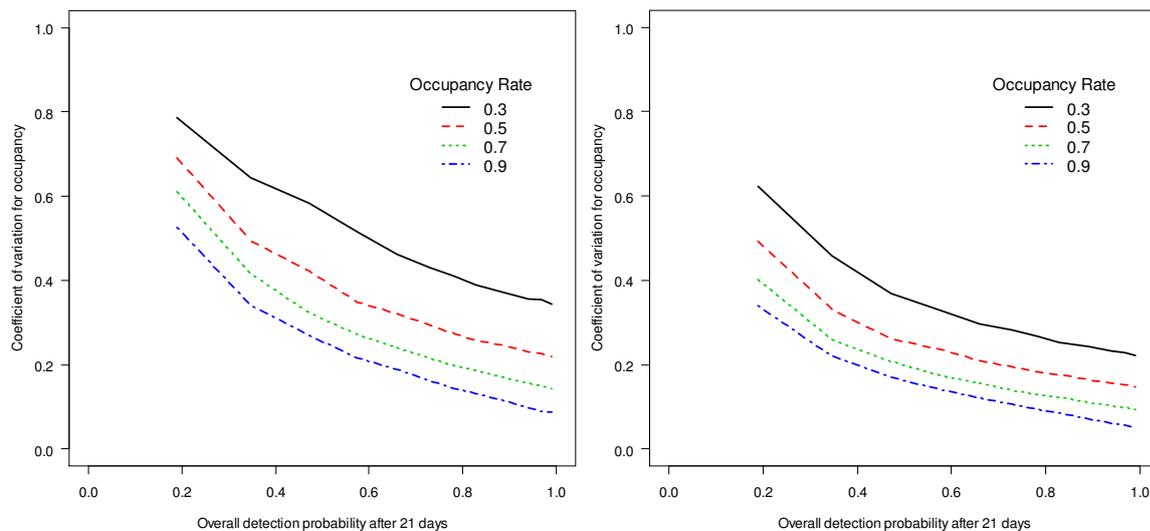
Figure 10. Relationship between the coefficient of variation (CV) and cumulative detection probability for different levels of occupancy rate, for 22 sites (left) and 49 sites (right).

3.5 Feral cat abundance and control

In 2008, a minimum of 15 individual cats were identified from photographs in LGNP-south and 6 in LGNP-north. The most appropriate model for abundance at LGNP-south was M_h (individuals may have different photo-capture probabilities from one another, but these remain the same across time periods). For LGNP-north the selected model was M_0 (all individuals have the same probability of capture in all time periods). Details of model outcomes are provided in Appendix 1.

Based on the model selection outcomes we assessed abundance of feral cats in LGNP-south and LGNP-north (Figure 11; Table 5). There were significantly more feral cats on the south site ($n = 19 \pm 4.82$ SD) compared to the north ($n = 6.7 \pm 1.08$ SD).
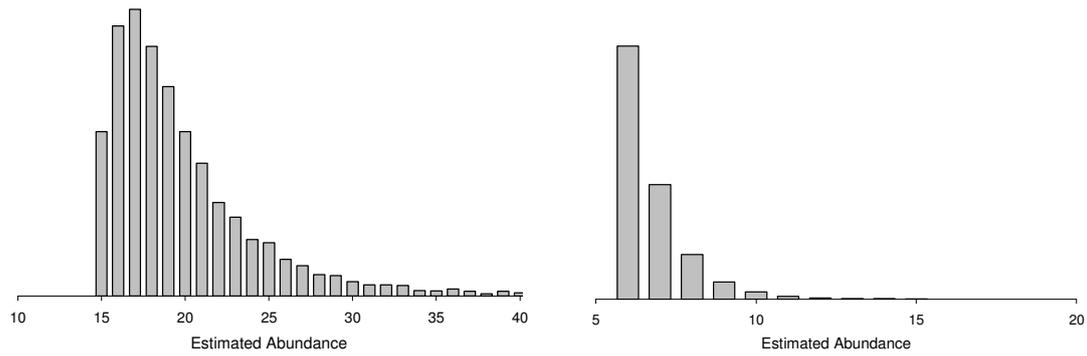
Figure 11. Estimated probability of abundance of feral cats and LGNP-south and LGNP-north.

In these plots, the values less than 15 and 6 from the south and north plot are not sampled in the Winbugs mark-recapture model. These values are less than the minimum number known, so they would be an impossible value.

Table 5. Estimates of abundance for each area from the best model, along with standard deviation.

Area	Model	Mean	SD
South	M_h	19.87	4.82
North	M_0	6.7	1.08

4 Discussion

Our aim was to further develop the camera monitoring protocol established in earlier work (Robley et al. 2008) and to assess its ability to measure change in a feral cat population before and after lethal control.

The monitoring protocol developed earlier indicated 49 cameras set for 20 days at one camera per site would be sufficient to detect a 50% decrease in feral cats 15% of the time. In the first part of this current work we set 44 cameras at 22 sites (two cameras per site) for an average of 21 days to investigate if dual cameras at a lower number of sites would detect and assess differences in site occupancy rates of feral cats. Dual cameras have been used to successfully detect Spotted-tailed Quoll (Nelson and Scroggie 2009). In addition to increasing detection probability, having two cameras per site allows for camera failure at a site. In the second part of this current work we attempted to validate the initial monitoring protocol by assessing changes in occupancy before and after a feral cat control operation. We used an occupancy modelling approach and a mark-recapture model in the first part, and an occupancy modelling approach only in the second part to assess if there was a difference in levels of site occupancy and abundance of feral cats between sites.

The monitoring protocol using paired cameras was successful at assessing a difference in site occupancy rate between a site that had been subject to sustained fox control for the previous five years (LGNP-south: $\psi = 0.69 \pm 0.10$ SE) and one that had not (LGNP-north: $\psi = 0.50 \pm 0.13$ SE), and did so with a high degree of confidence (88%). We were also able to assess a significant difference in occupancy rates for foxes between sites, with lower levels of occupancy on the site subject to fox control. We have no pre-fox control data on feral cat occupancy rate to indicate if the difference in occupancy is associated with a decline in foxes. It is possible the removal of a dominant or apex predator can result in the release of a subdominant or mesopredator, which in turn can lead to increased rates of predation on shared food resources (reviewed by Richie and Johnson 2009). However, these authors do not cite evidence for mesopredator release between foxes and feral cats, but do note that mesopredator release is widespread geographically and taxonomically.

The monitoring protocol using a single camera at 49 sites was able to successfully estimate occupancy rates for feral cats before ($\psi = 0.53 \pm 0.129$ SE) and after ($\psi = 0.45 \pm 0.137$ SE) cat control. However, this was not the most parsimonious outcome from the modelling, which was that occupancy was constant between periods. The fact that we were unable to detect a significant difference in feral cat occupancy after the feral cat control operation was probably because of the small number of feral cats killed ($n = 4$), despite 1368 trap nights. Power analysis indicated that we had little power to detect the small change in occupancy that was measured. However, the analysis also indicated that a reduction in feral cat occupancy from 0.53 to 0.3 would have had greater than 80% power to detect a significant reduction.

In pest control it is generally assumed that a reduction in abundance leads to a reduction in the damage caused by the pest (Braysher 1993, Hone 1994). Where eradication is not feasible, the reduction in abundance is achieved by stopping population growth following initial control. However, after a population reduction a higher rate of population growth generally occurs. Thus, this higher growth rate needs to be prevented after control in order to maintain the lower abundance (Hone 1994). Management operations to stop subsequent growth following the initial control would aim to keep the population at or below the lower level (Hone 1994). It is possible to estimate the point at which the population would be kept at or below this threshold level using the equation

$$p = 1 - e^{-r}$$

where r is the instantaneous rate of annual increase.[4] As an example, Short and Turner (2005) estimated the annual rate of increase for two populations of feral cats in semi-arid coastal Western Australia. Using the average of their estimates (0.985), the annual proportional threshold reduction in feral cats needed would be $1 - (e^{-0.985}) = 0.63$, or 63%. The monitoring protocol used in this study would be capable of measuring such a level of change in occupancy rate. Caution is required however, as a 63% reduction in occupancy is not the same as a 63% reduction in abundance.

Site-level daily detection probabilities differed between camera set ups (two per site versus one per site) and camera types. The detection probability for paired Reconyx cameras was slightly higher ($p = 0.077$) than for a single Reconyx ($p = 0.060$). There was also an interaction between camera and bait holder; the combination of Reconyx cameras and cage bait-holders having a higher detection probability than any other camera – bait-holder combination. A range of factors can contribute to successful detection of a species, relating to camera function and operation (Nelson and Scroggie 2009). In this study, care was taken to ensure all camera types were set up in the same manner each time, so the differences in detection rates of feral cats were more likely caused by differences in camera performance and not the spatial arrangement of cameras or underlying behavioural difference in cats. In particular, the speed at which a camera can power up and capture an image once motion is detected is a critical factor.

The cost and subsequent performance of digital cameras activated by heat-in-motion varies greatly between brands. Careful consideration needs to be given to the performance parameters of cameras used for estimating changes in occupancy rates when assessing the effectiveness of management actions.

We examined the uncertainty around the estimated occupancy rates when running paired cameras for 21 days at 22 sites compared to single cameras at 49 sites for the same length of time. Despite there being a slightly better chance of detecting a feral cat at an occupied site with paired cameras, the precision of the occupancy estimate was higher with a single camera at more sites. While there would be some additional cost for operating more sites, this would be offset by the increased precision in the estimate of occupancy and the improved power to detect any change that has occurred.

By comparing the two spatial configurations of cameras and using different camera and bait-holder types, we have shown that the camera protocol is flexible and that care needs to be taken in applying this tool in the field. Factors other than the spatial arrangement of cameras can have a significant impact on detection probability. The ability of the monitoring protocol to detect change in a cat population is not limited by the protocol, as shown by the clear difference between the LGNP-south and LGNP-north sites and the results of the power analysis, but either by the ability to kill substantial numbers of feral cats with the current methods, or by the resources available. However, a reduction of 50% in a feral cat population is probably the lower limit of what could be deemed a successful control operation.

The analytical approach used in this study allows for the degree of change to be assessed and the confidence with which this has occurred, enabling managers to assess the effectiveness of their control program.

The choice of which camera configuration to use would in part be determined by the resources available; we recommend one camera over at least 50 sites for areas of tis size operated over at least 21 days. The spatial separation of cameras is a subject that requires further investigation.

---

4 For more details and a full discussion on this topic, see Caughley (1980) and Hone (1994).

However, from this study it would appear that a randomly selected camera site within a 1 to 2 km^2 block would be sufficient.

There appeared to be no difference in the attractiveness of the lure we used in these trials. Detection probabilities before and after the control operation were similar, suggesting that feral cats were equally attracted to the lure after previous exposure several weeks earlier. However, the selection of bait-holder type and camera had a significant influence on detection. We recommend that small wire cages be used, rather than closed PVC tubes. It was found that Australian Ravens (*Corvus coronoides*) visited the cages almost continuously until all the chicken was removed. Although the PVC tubes eliminated this interference, they appear to have reduced the detection rates of feral cats. Modification of the cages, including using a smaller mesh size and placing stiff black plastic at the bottom and a small way up the side of the cage, is likely to prevent birds from removing the chicken.

We have demonstrated that the combination of digital cameras and statistical techniques that have been developed for the unbiased estimation of rates of occupancy under conditions of imperfect detection (MacKenzie et al. 2002, Tyre et al. 2003) can robustly determine differences in feral cat populations in temperate eucalypt woodland and forest. We have also demonstrated that digital cameras can be used to identify individual cats, enabling cat abundance to be assessed. These approaches can be used as monitoring tools for assessing changes in the status of feral cat populations over time and to assess the success of feral cat control operations.

If assessing actual abundance is of critical importance when using camera surveys, we recommend that two cameras be placed at right angles to the lure station. This should improve the level of individual identification and greatly improve the ability to determine the success of both fox and feral cat control operations.

References

Abbott, I. (2002) Origin and spread of the cat, *Felis catus*, on mainland Australia, with a discussion of the magnitude of its early impact on native fauna. *Wildlife Research* **29**, 51–74

Braysher, M. (1993) Managing vertebrate pests: principles and strategies. AGPS, Canberra

Caughley, G. (1980). Analysis of vertebrate populations. John Wiley & Sons, London

DEWHA (2008) Background document for the threat abatement plan for predation by feral cats. Department of the Environment, Water, Heritage and the Arts, Canberra

Edwards, G.P., de Preu, A. N. D., Shakeshaft, B. J. and Crealy, I. V. (2000) An evaluation of two methods of assessing feral cat and dingo abundance in central Australia. *Wildlife Research* **27**, 143–149

Forsyth, D. M., Robley, A. J. and Reddiex, B. (2005) Review of methods used to estimate the abundance of feral cats. Arthur Rylah Institute for Environmental Research, Department of Sustainability and Environment, Melbourne

Hone, J. (1994) Analysis of vertebrate pest control. Cambridge University Press, South Melbourne

Lunn, D. J., Thomas, A., Best, N. and Spiegelhalter, D. (2000) WinBUGS — a Bayesian modelling framework: concepts, structure, and extensibility. *Statistics and Computing* **10,** 325–337

MacKenzie, D. and Kendall, W. (2002) How should detection probabilities be incorporated into estimates of relative abundance? *Ecology* **83**, 2387–2393

MacKenzie, D. I., Nichols, J. D. , Royle, J. A., Pollock, K. H. , Bailey, L. L. and Hines, J. E. (2006) Occupancy estimation and modelling: inferring patterns and dynamics of species occurrence. Elsevier, San Diego, USA

Mahon, P. S., Banks, P. B. and Dickman, C.R. (1998) Population indices for feral carnivores: a critical study in sand-dune habitat, south-western Queensland. *Wildlife Research* **25**, 11–22

Nelson, J. L. and Scroggie, M. P. (2009) Remote cameras as a mammal survey tool — survey design and practical considerations. Arthur Rylah Institute for Environmental Research Unpublished Report. Department of Sustainability and Environment, Heidelberg

Otis, D. L., Burnham, K. P., White, G. C. and Anderson, D. R. (1978) Statistical inference from capture data on closed animal populations. *Wildlife Monographs* **62,** 1–135

Pledger, S. (2000) Unified maximum likelihood estimates for closed capture-recapture models for mixtures. *Biometrics* **56**, 434–442

Reddiex, B., Forsyth, D. M., McDonald-Madden, E., Einoder, L. D., Griffioen, P. A., Chick, R. R. and Robley, A. J. (2004) Review of existing feral goat, feral cat, feral rabbit, feral pig, red fox, and wild dog control in Australia. I. Audit and evaluation of control programs. Unpublished report to the Department of the Environment and Heritage. Arthur Rylah Institute for Environmental Research, Department of Sustainability and Environment, Melbourne

Richie, E. and Johnson, C. (2009) Predator interactions, mesopredator release and biodiversity conservation. *Ecology Letters* **12**, 1–18

Robley, A. (2009) Experimental research to evaluate detection methods and sampling designs used to determine the abundance of feral cats that have established wild populations in Australia – Stage 2 Progress Report. Unpublished report. Arthur Rylah Institute for Environmental Research, Department of Sustainability and Environment, Heidelberg

Robley, A., Gormley, A., Albert, R., Bowd, M., Smith, A. and Main, M. (2009) Assessing site occupancy of introduced predators in the Lower Glenelg National Park. Arthur Rylah Institute for Environmental Research Technical Report No. 185. Department of Sustainability and Environment, Heidelberg

Robley, A., Ramsey, D., Woodford, L., Lindeman, M., Johnston, M. and Forsyth, D. (2008) Evaluation of detection methods and sampling designs used to determine the abundance of feral cats. Arthur Rylah Institute for Environmental Research Technical Report Series No. 181. Department of Sustainability and Environment, Heidelberg, Victoria

Robley, A., Reddiex, B. Arthur, T. Pech, R. and Forsyth, D. (2004) Interactions between feral cats, foxes, native carnivores, and feral rabbits in Australia. Unpublished report to the Department of the Environment and Heritage. Arthur Rylah Institute for Environmental Research, Department of Sustainability and Environment, Melbourne

Saunders, G. and McLeod, L. (2007) Improving fox management strategies in Australia. Bureau of Rural Sciences, Canberra

Seebeck, J. and Clunie P. (1997) Predation of native wildlife by the domestic cat *Felis catus*. Flora and Fauna Guarantee Action Statement No. 80. Department of Natural Resources and Environment, Melbourne

Short, J. and Turner, B. (2005) Control of feral cats for nature conservation. IV. Population dynamics and morphological attributes of feral cats at Shark Bay, Western Australia. *Wildlife Research* **32**, 489–501

Soule, M. E., Bolger, D. T., Alberts, A. C., Wright, J., Sorice, M. and Hill, S. (1988) Reconstructed dynamics of rapid extinctions of chaparral-requiring birds in urban habitat islands. *Conservation Biology* **2**, 75–92

Spiegelhalter, D. J., Best, N. G., Carlin, B. P. and Van der Linde, A. (2002) Bayesian measures of model complexity and fit (with discussion). *Journal of the Royal Statistical Society,* series B, **64**, 583–616

Tyre, A., Tenhumberg, B., Field, S. A., Niejalke, D., Parris, K. and Possingham, H. (2003) Improving precision and reducing bias in biological surveys by estimating false negative error rates in presence-absence data. *Ecological Applications* **13**, 179–1801

Appendix 1

Model assessment for feral cat abundance

Table A1. Model scores for each model of feral cat abundance in Lower Glenelg National Park – south.

Model	Deviance	p	AIC
M_0	157.2	2	161.2
M_h	157.0	4	165.0
M_b	164.6	3	170.6
M_{bh}	163.5	5	173.5
M_t	159.7	28	215.7
M_{tb}	163.6	29	221.6
M_{tbh}	156.1	57	270.1
M_{th}	160.2	56	272.2

Table A2. Model scores for each model of feral cat abundance in Lower Glenelg National Park – north.

Model	Deviance	p	AIC
M_h	402.4	4	410.4
M_{bh}	402.9	5	412.9
M_0	429.5	2	433.5
M_b	432.6	3	438.6
M_t	405.6	28	461.6
M_{tb}	408.9	29	466.9
M_{tbh}	364.3	57	478.3
M_{th}	373.3	56	485.3