Glenelg Ark—benefits to biodiversity from long-term fox control 2015 update

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Front cover photo: (a) Members of the Glenelg Ark field team: Tom McKinnon, Fed Warton, Megan Bull, (b) Aileen Smith, (c) Justin Cook, (d) Michael Bowd, (e) Chris Hatfield, (f) Dwayne Hauser and (g) Richard Wilson (photographer: DEWLP). (h) Red Fox, (i) Common Brushtail Possum, (j) Southern Brown Bandicoot and (k) Long-nosed Potoroo (photographer: Alan Robley). (l) Feral Cat (photographer: Marc Perri).

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Summary

The Glenelg Ark project was established in 2005 to facilitate the recovery of selected native mammal species considered at risk from Red Fox (‘fox’; *Vulpes vulpes*) predation. The project established continuous landscape-scale fox baiting across 100,000 ha of State Forest and National Park in south-western Victoria. Three native mammal species that are present in the Glenelg Ark project area in low numbers, which have patchy distributions and are thought to be at risk from fox predation were selected for monitoring. These were the Southern Brown Bandicoot (*Isoodon obesulus*), the Long-nosed Potoroo (*Potorous tridactylus*) and the Common Brushtail Possum (*Trichosurus vulpecula*).

This report updates the previous 2012 monitoring and evaluation report (Robley et al. 2014) by adding new data on the outcome of the fox control operation and the response of targeted native species from 2013 to 2015. During this period, an assessment of the monitoring method for detecting native mammal species (hair-tubes) was compared to a new approach (digital cameras). This report also contains recommendations for future management options and suggests areas of further research aimed at improving land managers' knowledge and practices in order to attain better conservation outcomes.

Differences between the level of fox activity at locations with and without fox control (i.e., treatment and non-treatment locations) were assessed from the number of independent images captured on camera traps from 2013 to 2015. Activity at locations with fox control was significantly lower compared with activity at locations without fox control. There was no significant difference in feral Cat activity between sites with and without fox control, although the point estimates suggest higher levels of Cat activity in treated areas.

There was no significant difference in the detection rates of native mammals between the hair-tubes and the digital cameras; however, digital cameras captured a broader range of species, were less expensive to operate in the field, and the camera data had smaller confidence limits.

We used data gathered in 2013 and 2014 from hair-tubes, digital cameras, and both methods combined to assess differences in the number of sites occupied by native mammals between locations with and without fox control. In 2013, all three methods showed strong evidence of a positive effect from fox control on the number of sites occupied by Common Brushtail Possums; cameras, and hair-tubes and cameras combined showed strong evidence of a positive effect for Long-nosed Potoroos and inconclusive evidence of a positive effect for Southern Brown Bandicoots.

In 2013, cameras and hair-tubes alone, and cameras combined with hair-tubes showed strong evidence of a positive effect on site occupancy from fox control on Common Brushtail Possums and Long-nosed Potoroos, and inconclusive evidence of a positive effect on Southern Brown Bandicoots. In 2014, cameras-only showed no evidence of an effect for Common Brushtail Possums, Southern Brown Bandicoots and Long-nosed Potoroos. Hair-tubes alone failed to detect any difference in the number of sites occupied for any of the three species in either year.

We used the camera-trap data to update the long-term dataset (2005–2015). Since the previous Glenelg Ark update in 2012, the number of sites occupied by Common Brushtail Possums remained higher in treatment locations compared with the number in non-treatment locations; little change was observed for Long-nosed Potoroos, and there is no indication of a further increase in number of sites occupied since the initial spike in 2008; the site occupancy of Southern Brown Bandicoots remained unchanged until 2015, at which point the number of sites occupied was greater at locations with fox control.

Key recommendations

The following recommendations are made to improve the outcomes of Glenelg Ark.

Item	Recommendation	Detail
Native species' response	Move to using digital cameras as the main monitoring tool for native species.	There was no overall significant difference in detection rates between cameras and hair-tubes; however, cameras are less costly to operate and are able to 'capture' a wider range of species in all weather conditions.
	Develop bandicoot and potoroo habitat suitability surfaces for the Glenelg Ark project area using presence/absence data to aid in setting species response targets and potential new control and / or monitoring sites.	The limited response of bandicoots and potoroos may be due to a lack of suitable habitat for these species. We propose that the site occupancy information be used to explore the possible limitation of suitable habitat. This data combined with freely available remotely sensed habitat data (e.g., vegetation type, topography, fire history, distance to drainage lines, forest edge) can be combined with information on detection and non-detection of species at sites to develop a species habitat suitability surface across the project area. This information will be useful in understanding the expected increase in species occurrence and also identify potential new locations for monitoring and or fox control actions.
	Using species distribution models of the benefits of fox control for the Heath Mouse, select sites for targeted monitoring on treatment and non-treatment locations.	Current monitoring sites were placed in locations based on Ecological Vegetation Divisions mapping and the best understanding of 'suitable' habitat at that time. Predictive species distribution models that incorporate the likely benefit of fox control have been developed in recent years. These could be used to select sites more likely to have the Heath Mouse present. If fox control has delivered a positive benefit, there should be a detectable difference between treated sites and non-treated sites.
Fox control	Review the predator control program and investigate options for improving where needed.	Bait density and the frequency of bait replacement, as well as bait type and placement can affect the outcome of fox control. A general review of the program with consideration of the items above is warranted.
	Use spatially explicit individual-based population models of the reduction in foxes from control operations to develop strategies for increased reduction in fox populations.	Use existing empirical data and expert elicitation to develop models testing a range of baiting scenarios in order to assess their impact on fox abundance. Despite decades of fox control, we have little understanding of what the best strategy is for reducing and maintaining lower fox abundances.
Differences in fox and feral Cat abundances across treatment/non-treatment areas	Undertake camera monitoring specifically to assess the effectiveness of the control operation, and use the information to assist in the development of an integrated feral Cat and fox control strategy.	Determine the number of camera sites required through a power analysis to assess differences (if they exist) in fox and feral Cat activity on treated and non-treated sites.
Alternative survey methods for foxes and feral Cats	Assess and cost the feasibility of genotyping DNA from fox scats collected using scat detector dogs.	Scat detector dogs and genotyping DNA from scats have both been used successfully to enumerate fox populations before and after fox control. A similar approach could be used in Glenelg Ark to assess differences between baited and comparable unbaited areas.
Scientific support	Continue to source scientific support and advice concerning the ongoing implementation and development of Glenelg Ark.	Evaluation and interpretation of monitoring data, development of new projects addressing emerging issues, and general guidance to the project has been essential to its success.

Item	Recommendation	Detail
Monitoring and reporting	Continue annual monitoring, evaluation and reporting.	Continue annual monitoring and reporting in order to closely track changes in predators and prey, thus allowing more responsive management of emerging issues, e.g. a decline in Southern Brown Bandicoots; a change in feral Cat abundance.
Filling specific knowledge gaps	Develop a set of potential student projects to fill identified knowledge gaps.	The current monitoring program does not assess changes in small native mammals (e.g. Heath Mouse and White-footed Dunnarts), or unintended consequences (e.g. the possible negative impacts on biodiversity of overabundant medium- and small-sized herbivores, e.g. wallabies and Common Brushtail Possums). A series of student projects aimed at filling these knowledge gaps and taking advantage of the infrastructure that Glenelg Ark provides would be possible.

1 Introduction

The Glenelg Ark project was established in July 2005 to facilitate the recovery of selected native mammal populations considered at risk from Red Fox ('fox'; *Vulpes vulpes*) predation. The project established continuous landscape-scale fox baiting across 100,000 ha of State Forest and National Park in south-western Victoria. To justify ongoing government commitment and community support for Glenelg Ark, its benefits to Victoria's biodiversity must be demonstrated. The monitoring and evaluation component of Glenelg Ark measures: (i) the response of foxes to control activities, and (ii) the response to a reduced abundance of foxes of native species that are at risk from fox predation. Without such a program, management will have no capacity to justify reinvestment of scarce public conservation funds, improve management actions based on scientific information, and maintain community support. Thus, monitoring and evaluation forms an essential part of management and is not an imposition or adjunct to it.

Three native mammal species that are present in the Glenelg Ark project area in low numbers (Robley et al. 2011), have patchy distributions (Menkhorst 1995) and are thought to be at risk from fox predation were selected for monitoring. These are the Southern Brown Bandicoot (*Isoodon obesulus*), the Long-nosed Potoroo (*Potorous tridactylus*) and the Common Brushtail Possum (*Trichosurus vulpecula*). The bandicoot and potoroo are medium-sized ground-dwelling mammals (c. 1.0 kg and c. 1.2 kg, respectively) with high and moderate fecundity, respectively (Lobert and Lee 1990). Both species are known to be preyed upon by foxes (Seebeck 1978) and have been reported to positively respond to a reduction in foxes (Kinnear et al. 2002; Arthur et al. 2012). The Common Brushtail Possum is a semi-arboreal species weighing c. 3.0 kg, has a low rate of fecundity (Kerle and How 2008) and is known to occur in the diet of foxes (Triggs et al. 1984) and to respond to fox control (Kinnear et al. 2002).

Given the role that foxes have played in the decline and extinction of Australian mammals (Short and Smith 1994; Salo et al. 2007), the examples of mammal recovery following sustained reduction in fox abundance (Saunders et al. 2008), and considering our knowledge of the initial status of the targeted prey species, it was reasoned that once fox numbers had been reduced, the prey species would be able to escape limitation and the number of sites occupied by the targeted prey species should increase. We assessed changes in foxes and feral Cats (*Felix catus*) by comparing their activity (number of independent images captured by digital cameras at a monitoring site) at locations with an ongoing history of continuous fox control (fortnightly replacement baiting) with that at locations with no history of fox control. We assessed the response of native species to the reduction in foxes by comparing the number of monitoring sites occupied by the native species at locations with and without ongoing fox control.

The response of native species to the reduction in fox abundance at sites in Glenelg Ark was assessed using detections resulting from species contact with hair-tubes each spring from 2005 to 2012. While hair-tubes have been widely used for detecting and assessing the status of ground-dwelling mammals throughout Australia (Lindenmayer et al. 1999), a newer approach using digital cameras to 'trap' animals has been developed in recent years (O'Connell et al. 2011). As part of the continuous improvement process for the Glenelg Ark project, an investigation into the use of digital cameras was undertaken from 2013 to 2014. We examined the differences (if any) in the occupancy and detection estimates of Common Brushtail Possums, Long-nosed Potoroos and Southern Brown Bandicoots obtained using camera trapping compared with using hair-tubes at the six monitoring locations within the Glenelg Ark project area. The aims were:

(i) to determine whether there was any increase in efficacy in changing the monitoring tool used from hair-tubes to digital cameras,

(ii) to determine whether in moving to camera based monitoring it would be possible to maintain continuity with the 9-year hair-tube dataset, and

(iii) to compare the relative costs of each method of data collection.

This report updates the previous monitoring and evaluation report covering 2005–2012 (Robley et al. 2014), by incorporating new data on the outcome of the fox control operation and the response of the targeted native species from 2013 to 2015. This report also contains recommendations on future management options and suggested areas of further research. The outcome is that land managers, policy-makers, and the community can now make informed, evidence-based assessment of the success of broad-scale mainland fox control operations, and decisions about future directions.

2 Methods

2.1 Glenelg Ark operations area

The Glenelg Ark operations area is located in far south-west Victoria, near the township of Heywood (38° 07' 50'' S, 147° 37' 45'' E), and includes six locations in State Forests and National Parks. The main ecological vegetation communities across all six locations are heathy woodland, lowland forest, herb-rich woodland, and wet heathland. The area receives an average annual rainfall of 700 mm, and an average minimum and maximum temperature of 8.1°C and 17.6°C, respectively.

2.2 Monitoring and evaluation design

Three monitoring areas, known as Treatment Monitoring Locations (TMLs, i.e. locations that are subject to fox control) and three Non-Treatment Monitoring Locations (NTMLs, i.e. locations not subject to fox control) (Fig. 1) were used to assess the benefits of fox control. In the TMLs and NTMLs there had been little fox control prior to 2005. In order to achieve a broad-scale reduction in foxes across the public land areas, fox control was consolidated in the southern half of the overall project area (Fig. 1). This meant that random allocation of treatment and non-treatment sites was not feasible. The six monitoring locations are:

1. Lower Glenelg National Park – south (LGNP-south; TML; 8954 ha)
2. Lower Glenelg National Park – north (LGNP-north; NTML; 4659 ha) (separated from ‘1’ by the Glenelg River)
3. Cobboboonee National Park (TML; 9750 ha)
4. Annya State Forest (NTML; 8520 ha)
5. Mount Clay State Forest (TML; 4703 ha)
6. Hotspur State Forest (NTML; 6940 ha).

This strategy was designed to enable the identification of any patterns of association between a reduction in foxes and an increase in targeted native species, but does not allow any statistical interpretation of causality (Lande et al. 1994).

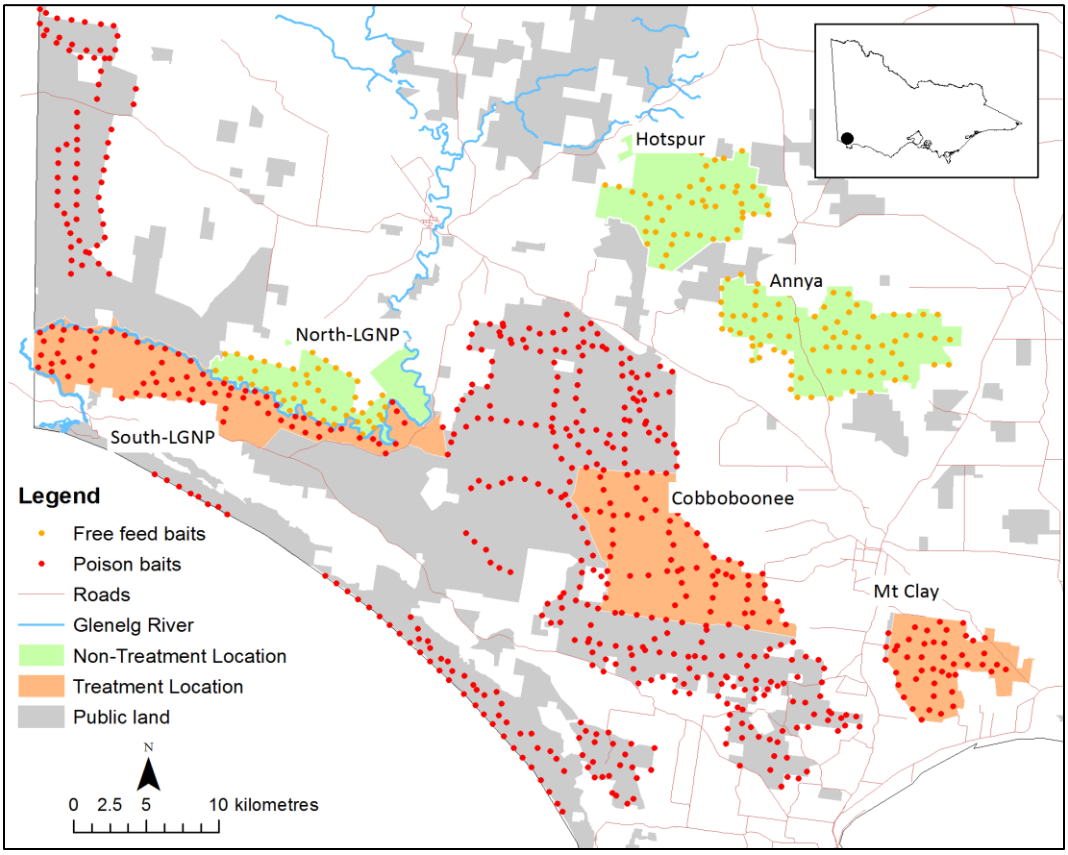


Figure 1. Glenelg Ark operations area. Tan polygons = treatment monitoring locations; green polygons = non-treatment monitoring Locations; red dots = poison bait stations, orange dots = free feed bait.

2.3 Measuring changes in fox and feral Cat activity

In this update, we examined the difference in fox and feral Cat activity between treatment and non-treatment locations from 2013 to 2015 using data generated from camera traps (see section 2.4 for details of when and where camera traps were set). We used the number of independent images (separated by 1 hr) captured per day at each camera site to generate an index of activity for foxes and feral Cats. Fox and feral Cat activity was assessed using a Bayesian non-linear mixed model with treatment set as a fixed effect and year set as random effect in the fox and feral Cat model; the presence of foxes was included in the Cat model as a fixed effect to test the influence foxes might have on Cat activity. The (log)number of cameras that operated on any given day was used as an offset in the model to allow for differing numbers of camera days per sampling period.

2.4 Measuring site occupancy changes in mammal species

Site occupancy of the three target-species (Long-nosed Potoroo, Southern Brown Bandicoot and Common Brushtail Possum) was monitored annually at 40 sites established within each TML and NTML (Fig. 2). The positioning of monitoring sites was based on descriptions of the habitat preferred by the target native mammal species (Menkhorst 1995) and stratified according to the proportion of preferred habitat within each TML and NTML.

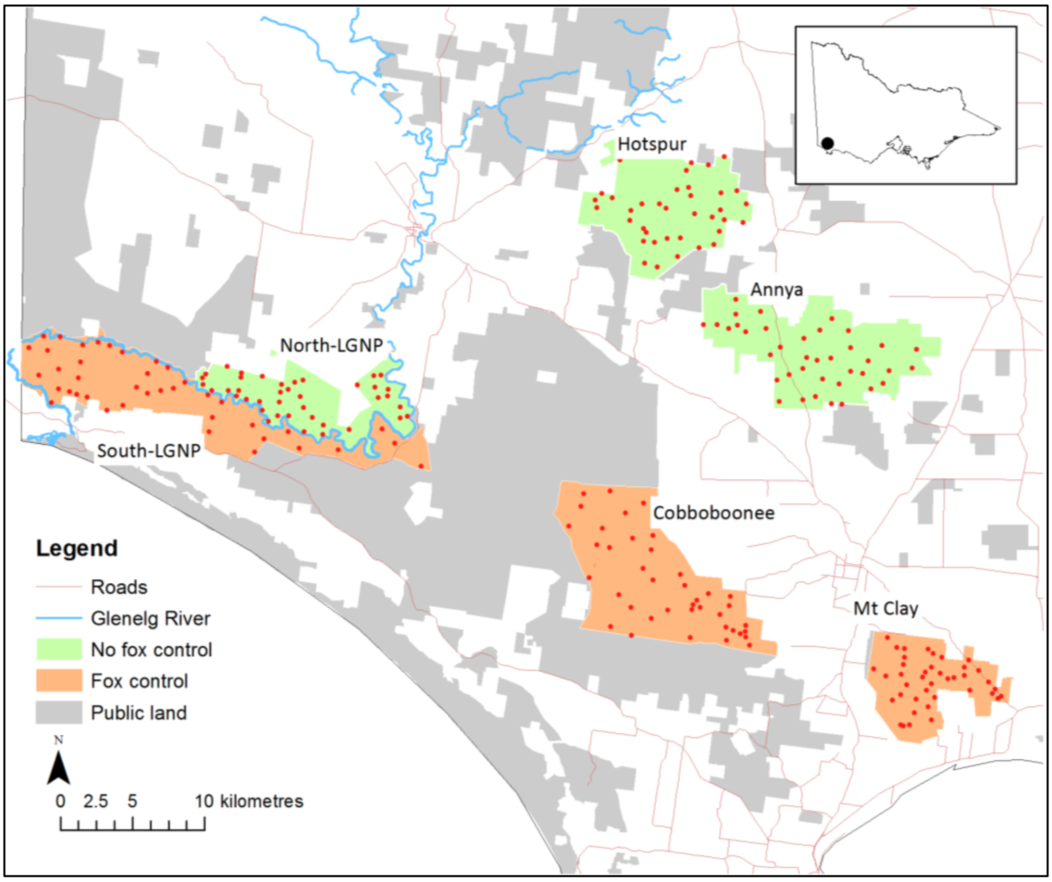


Figure 2. Monitoring sites in the treated (tan polygons) and non-treated (green polygons) monitoring locations of Glenelg Ark are indicated by red dots.

Monitoring was typically undertaken in spring (2005, 2008–2015). Initial sampling, prior to the commencement of poison baiting, was conducted in winter 2005. In 2006, sampling was undertaken in late winter due to staff resource issues, and the spring 2007 samplings at Mt Clay and Hotspur were delayed due to staff being allocated to planned burning duties; as a result, monitoring was undertaken in summer 2007/2008.

From 2005 to 2012 at each monitoring site, nine 'Handiglaze' hair-tubes (Murray 2005) (baited with peanut butter, rolled oats and honey) were set and checked daily for four consecutive days, with tapes being replaced each day. These daily surveys represented four repeat surveys of the monitoring site per sampling period (Fig. 3). In spring 2013 and 2014, a single digital camera (Reconyx RapidFire ProPC90, Reconyx, LLP Wisconsin, USA) was set at one of four possible locations within a hair-tube grid at each monitoring site (Fig. 3). The location of the camera within a monitoring site was determined by a series of coin tosses. Cameras were placed at an equal distance from the nearest hair-tube to reduce the influence of the presence of the lure in the hair-tube. Cameras were attached to the nearest tree at 20–30 cm above the ground. A lure of truffle oil, peanut butter, rolled oats and honey was secured to the ground in a small, ventilated container 2 m in front of the camera. Cameras were operated for a minimum of 30 days, with each day representing a repeat survey of the monitoring site per sampling period.

In 2015, hair-tubes were discontinued, and only cameras were employed as the survey tool.

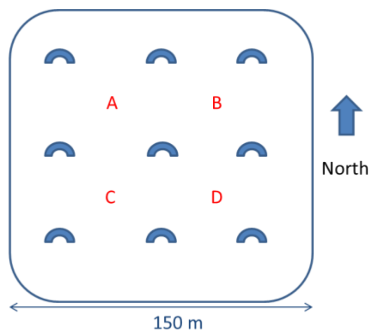


Figure 3. Layout of nine hair-tubes and possible location (A, B, C or D) of the single digital camera at a monitoring site.

2.4.1 Costs of hair-tube and camera-trap methods

The costs of each method were determined based on the number of person-days required to deploy and retrieve either hair-tubes or cameras, including the cost of processing hair-tube tapes or camera images, respectively, prior to analysis, and the cost of analysis of the hairs. Data analysis for determining occupancy estimates was not included because it would have been the same regardless of the method of detection used. Costs were based on 2013–2014 staff costs for DELWP regional services field staff.

2.4.2 Data analysis

Long-term site occupancy changes in native mammals

To assess the long-term responses of the selected native mammals, we used a multiseason occupancy model to estimate the occupancy (ψ), detection (p), local colonisation (γ) and local survivorship (ε) for monitoring sites within a location from 2005 to 2015 (MacKenzie et al. 2003, 2006). Models were constructed in a Bayesian framework (Kéry 2010), using a space–state formulation (Royle and Kéry 2007).

Separate models were constructed for each of the three native species of interest. The data for each species was summarised for each monitoring site. Each model allowed for differences in parameters at each of the six locations: Annya, Hotspur and LGNP-north (NTMLs); and Cobboboonee, Mt Clay and LGNP-south (TMLs). The models also allowed for differences in daily detection rates due to whether a hair-tube or camera was being used for detections in 2013 and 2014. Additionally, hair-tube detection of Long-nosed Potoroos and Southern Brown Bandicoots was allowed to differ depending on whether Common Brushtail Possums were detected at the site. [Hair analysis from the tubes indicated that the tapes were being swamped with possum hairs (B. Triggs pers. comm.), and therefore potoroos and bandicoots could have been under-reported.]

Hair-tubes and digital cameras

To determine which method (hair-tube or digital camera) best estimated occupancy (ψ) rates, we compared the relative detection rates obtained for each of the three species using each method. We also used data from each method and from the combination of methods to assess the difference in the detection rates and in the number of sites occupied between treated and non-treated areas in 2013 and 2014. To do this we used single-season occupancy models (Mackenzie et al. 2003, 2006) in a state–space formulation in a Bayesian framework. Three separate models were constructed in order to analyse the data: camera trap data, hair-tube data, and combined camera trap and hair-tube data, with fox treatment (i.e. fox control/no fox control) as a factor. Using a combination of detection methods often improves the probability of detecting the species of interest. We compared this 'best' method with the separate hair-tube and camera approaches to gain an understanding of the relative merit of each individual approach.

Given that a hair-tube had detected the species of interest at some of the sites, this information was used in the determination of the camera detection rate. For example, hair-tubes may have detected sp. A at 10 of the 40 sites; we would then know that if a camera failed to detect sp. A at one of these sites, it was a lack of detection and not a true absence. This information was used to obtain more precise detection estimates for the camera trap technique. When compared with the combined approach, it allowed us to assess whether camera traps were 'better' at detecting sp. A. A similar approach was applied to the data from the combined method compared with the hair-tube-only data. Of particular interest was any difference in detection between hair-tubes or camera traps relative to the combined method, especially with regard to the other two species in the presence of Common Brushtail Possums, because this species tended to dominate hair-tubes.

Each time a model was run it produced a mean estimate of occupancy. We ran models for 10,000 iterations and compared the individual mean estimates with the overall mean estimate (or posterior distribution). The proportion of times the average occupancy was higher for fox control sites than for non–fox control sites was used to determine whether fox control impacted occupancy rates for a given species. If this proportion was >0.95 or <0.05, it was assumed to be strong evidence that fox control did or did not influence occupancy, respectively.

The models were constructed in JAGS (Plummer 2003) via R (R Development Core Team 2016), using the package R2jags (Su and Yajima 2012). Model chains were run until the chains converged. Convergence was defined as when all Gelman and Rubin's convergence diagnostic potential scale reduction factors were <1.05 (Gelman et al. 2004).

Depending on the distribution of the species, some parameters (e.g. occupancy, colonisation or extinction) may have been poorly estimated. For example, if very few sites within an area were occupied, then the estimate of occupancy for the following year would be uncertain because the probability of the true number of sites occupied could be low or very high.

Inferences derived from these models were based on changes in occupancy at a site level (i.e. at the sites where hair-tubes or cameras were located), rather than at the broader landscape level (e.g. differences between Cobboboonee National Park and Hotspur State Forest).

3 Results

3.1 Rainfall

Mean annual rainfall (recorded at the Portland Airport, ~20 km from the project area centre) differed substantially from the long-term average in a number of years over the period 1990–2015 (Fig. 4). The years 1993–2000 saw consistently below-average rainfall. The year 2006 saw the largest departure from the long-term annual mean, with a 37% reduction. In the 10 years since the project began in 2005, there have been 7 years with below-average rainfall.

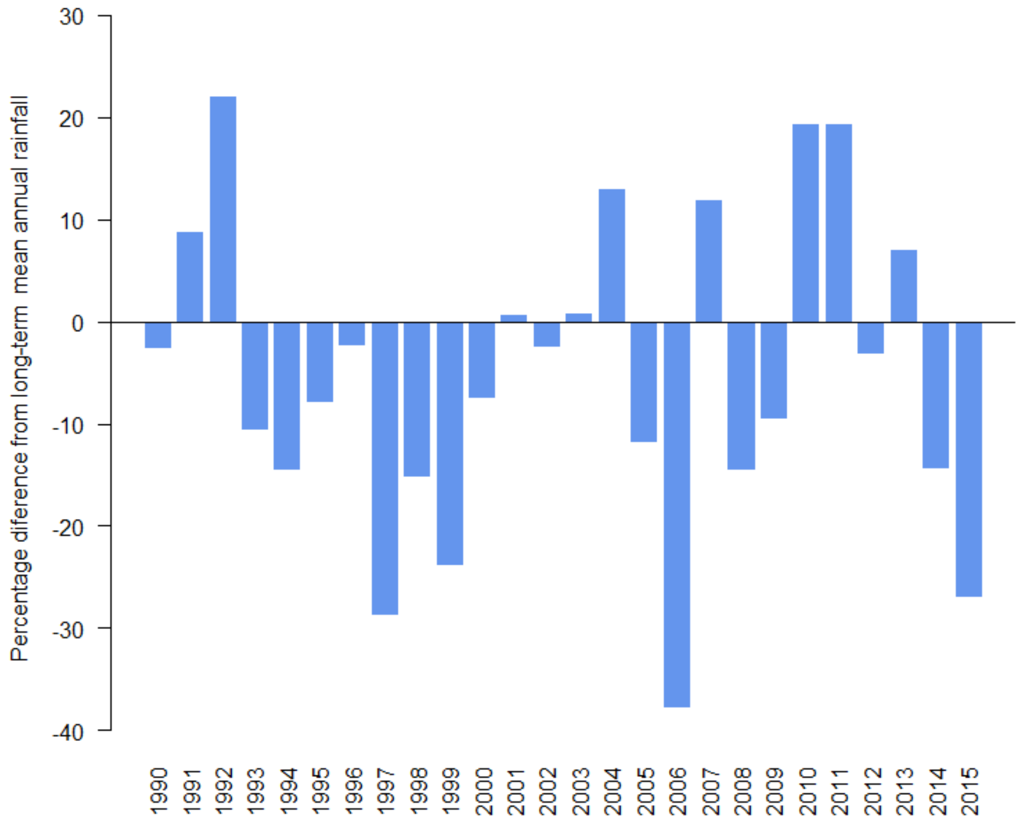


Figure 4. Difference in mean annual rainfall (%) from the long-term (1908–2015) average for 1980 to 2015. Data from the rainfall station at Portland Airport.

3.2 Fox and feral Cat activity

3.2.1 Fox activity

Fox activity was significantly lower at locations with fox control compared with locations with no fox control (Fig. 5). There was no difference in fox activity between years for 'fox control' locations; similarly, there was no difference in fox activity between years for 'no fox control' locations (Appendix 1).

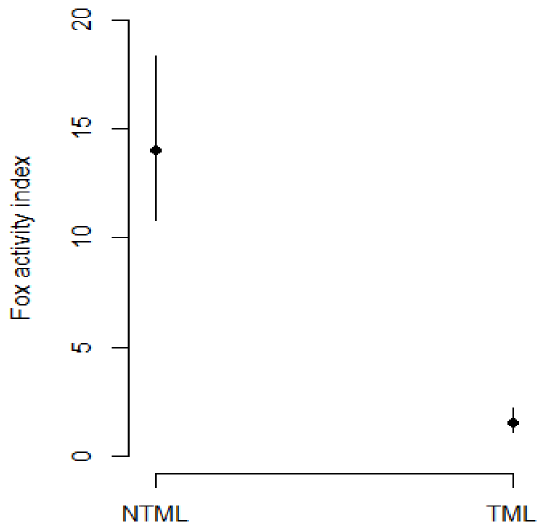


Figure 5. Fox activity (number of images per day at each camera site) at treatment monitoring locations (TMLs) and non-treatment monitoring locations (NTMLs). Bars are 95% credible intervals.

3.2.2 Feral Cat activity

There was no significant difference in feral Cat activity between treatment and non-treatment monitoring locations (Fig. 6), or between years (Appendix 1). Feral Cat activity was very low across the TMLs and NTMLs.

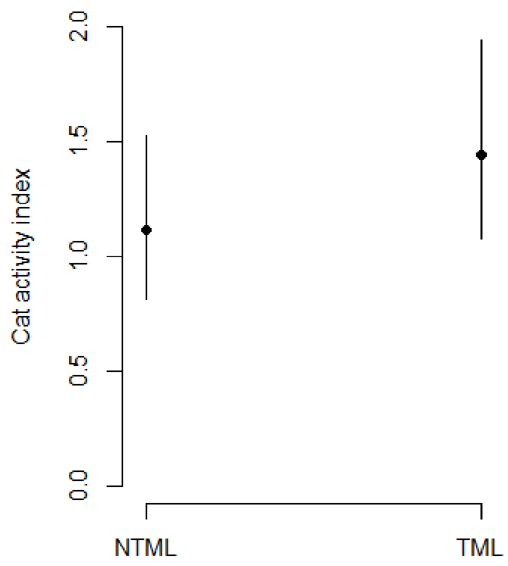


Figure 6. Feral Cat activity (number of images per day at each camera location) across treatment type (TML = fox control, NTML = no fox control) as measured by digital cameras. Bars are 95% credible intervals.

3.3 Transitioning from hair-tubes to camera traps

3.3.1 Detection rates

Common Brushtail Possums

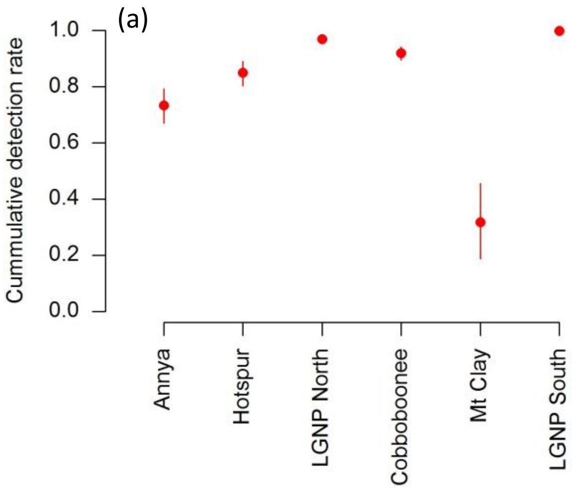
The combination of hair-tubes and cameras was generally better at detecting possums, with 178 site detections at 41.8% of sites over the 2 years (Table 1). Common Brushtail Possums were detected at all locations in both 2013 and 2014. At 83 sites (19.4%), only one method detected possums (61 camera-only detections and 22 hair-tube-only detections).

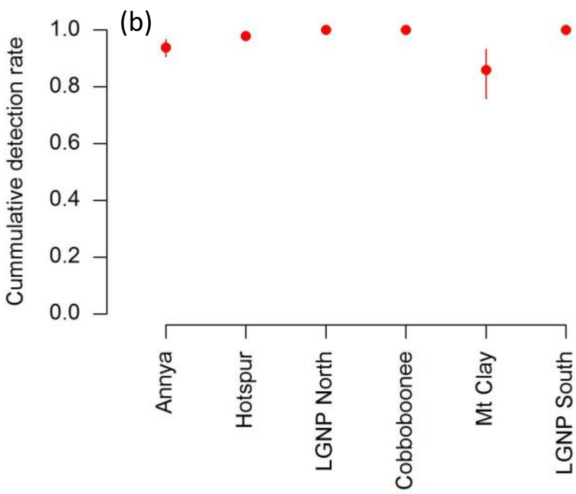
Table 1. The number of monitoring sites at which Common Brushtail Possums were detected at each location, the method(s) used to detect them and the naïve occupancy rate. Both = combination of hair-tube and camera methods.

Location	Year	Both methods	Camera	Hair-tube only	Neither	Total	Naïve occupancy rates
Annya	2013	4	5	4	27	40	0.325
Cobboboonee	2013	17	9	3	6	35	0.828
Hotspur	2013	7	4	4	21	36	0.417
Mt Clay	2013	1	6	0	25	32	0.219
LGNP-north	2013	20	6	1	4	31	0.871
LGNP-south	2013	34	4	0	1	39	0.974
Annya	2014	4	4	3	29	40	0.275
Cobboboonee	2014	23	4	2	6	35	0.829
Hotspur	2014	6	9	4	17	36	0.528
Mt Clay	2014	1	4	0	27	32	0.157
LGNP-north	2014	23	5	1	2	31	0.935
LGNP-south	2014	38	1	0	0	39	1.000
Total		178	61	22	165	426	

Cumulative detection rates (for 9 hair-tubes over 4 days and 1 camera over 30 days) for Common Brushtail Possums varied between locations for both types of device (Fig. 7a and 7b), but were consistently higher when using cameras.

Figure 7. Cumulative detection rates [using hair-tubes (a), or cameras (b)] for Common Brushtail Possums at the six monitoring locations in Glenelg Ark.





Long-nosed Potoroos

Cameras were the best method for detecting Long-nosed Potoroos, with detections at 29 sites (6.8% of sites over the 2 years). Long-nosed Potoroos were detected at all locations, but at limited sites within each location (Table 2). At 36 (84.4%) sites, only one method detected Long-nosed Potoroos (29 camera-only detections and 7 hair-tube-only detections). Interestingly, 3 of the 6 locations in 2013 (Hotspur and both LGNP locations) and 5 locations in 2014 had no sites at which both methods detected Long-nosed Potoroos. Of those locations (Hotspur in 2013 and 2014, and Mt Clay and LGNP-north in 2014) had no hair-tube detections, even though Long-nosed Potoroos were known to be present (via the camera data at one site).

Table 2. The number of sites at which Long-nosed Potoroos were detected at each location, the method(s) used to detect them and the naïve occupancy rate. Both = combination of hair-tube and camera methods.

Location	Year	Both	Camera only	Hair-tube only	Neither	Total	Naïve occupancy rates
Annya	2013	1	0	1	38	40	0.050
Cobboboonee	2013	3	1	0	31	35	0.114
Hotspur	2013	0	1	0	35	36	0.028
Mt Clay	2013	3	4	1	24	32	0.250
LGNP-north	2013	0	3	1	27	31	0.129
LGNP-south	2013	0	3	3	33	39	0.154
Annya	2014	1	0	0	39	40	0.025
Cobboboonee	2014	3	4	1	27	35	0.229
Hotspur	2014	0	1	0	35	36	0.028
Mt Clay	2014	0	4	0	28	32	0.125
LGNP-north	2014	0	3	0	28	31	0.097
LGNP-south	2014	1	5	0	33	39	0.154
Total		12	29	7	378	426	

Cumulative hair-tube detection rates for Long-nosed Potoroos varied between locations and according to the presence/absence of Common Brushtail Possums (Fig. 8a; Table 3). There was strong evidence that at LGNP-north and LGNP-south, detection rates for Long-nosed Potoroos were reduced when Common Brushtail Possums were present. However, there was strong evidence that at Mt Clay Long-nosed Potoroo detection rates increased when Common Brushtail Possums were detected. Cumulative camera-trap detection rates for Long-nosed Potoroos varied between locations but were all uniformly high (above 0.9). Cobboboonee and Mt Clay had higher detection rates than the other locations (Fig. 8b).

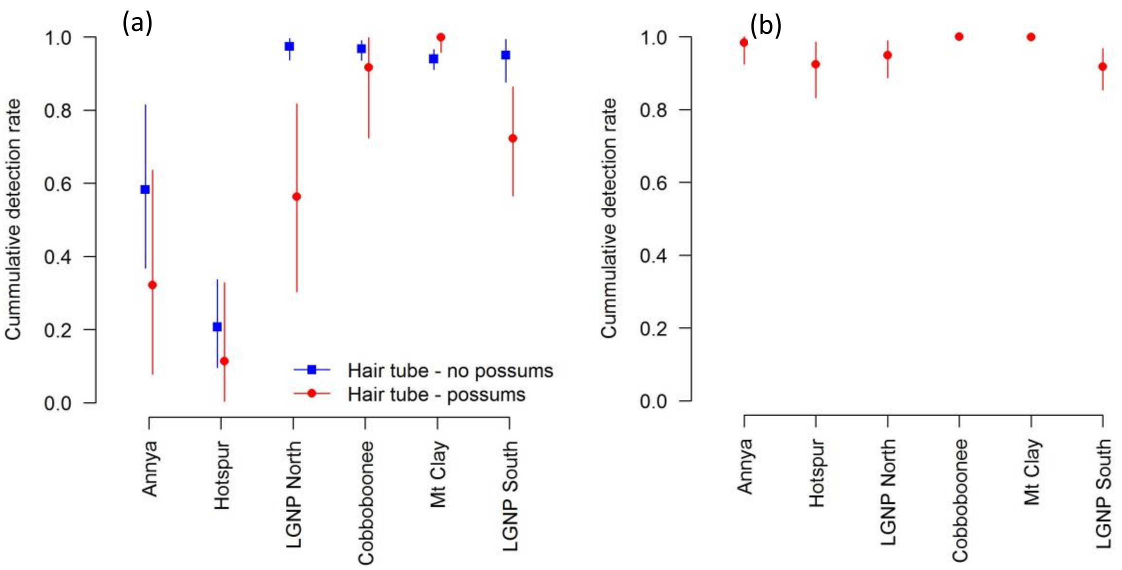


Figure 8. Cumulative detection rates [using hair-tubes (a), or cameras (b)] for Long-nosed Potoroos at the six monitoring locations in the Glenelg Ark area. Blue = no possums present; red = possums present.

Table 3. Differences in hair-tube detection rates for Long-nosed Potoroos at sites with possums detected and sites without possums detected. Green shading highlights where there was substantial evidence that the detection probabilities were larger for Long-nosed Potoroos when Common Brushtail Possums were detected. Blue shading highlights where there was substantial evidence that the detection probabilities were smaller for Long-nosed Potoroos when Common Brushtail Possums were detected. HDI=Highest Density Interval (similar to the credible interval in frequentists statistics).

Location	Median	Lower 95% HDI	Upper 95% HDI
Annya	–0.100	–0.249	0.045
Hotspur	–0.025	–0.086	0.044
LGNP North	–0.408	–0.571	–0.233
Cobboboonee	–0.111	–0.353	0.125
Mt Clay	0.364	0.022	0.531
LGNP South	–0.252	–0.418	–0.080

Southern Brown Bandicoots

Cameras were the best method for detecting Southern Brown Bandicoots, with detections at 39 camera only sites (9.15%) and eight combined sites. Southern Brown Bandicoots were detected at all locations, but at limited sites within each location (Table 4). At 44 sites (10.4%), only one method detected this species (39 camera-only detections and 5 hair-tube-only detections). Interestingly, four of the six locations in 2013 (Cobboboonee, Mt Clay and both LGNP areas) had no site at which both methods detected Southern Brown Bandicoots. Two of those areas (Cobboboonee and Mt Clay) had no hair-tube detections, even though Southern Brown Bandicoots were known to be present (via the camera data) at seven sites across the two areas combined. Conversely, one of those areas (LGNP-north) had no camera detections, even though Southern Brown Bandicoots were known to be present (via the hair-tubes at one site). In 2014, three locations (Hotspur, Mt Clay and LGNP-south) had no site at which both methods detected Southern Brown Bandicoots, even though they were known to be present via camera traps. Bandicoots were not detected by any method or combination of methods at LGNP-north in 2014.

Table 4. The number of sites at which Southern Brown Bandicoots were detected at each location, the method(s) used to detect them and the naïve occupancy rate. Both = combination of hair-tube and camera methods.

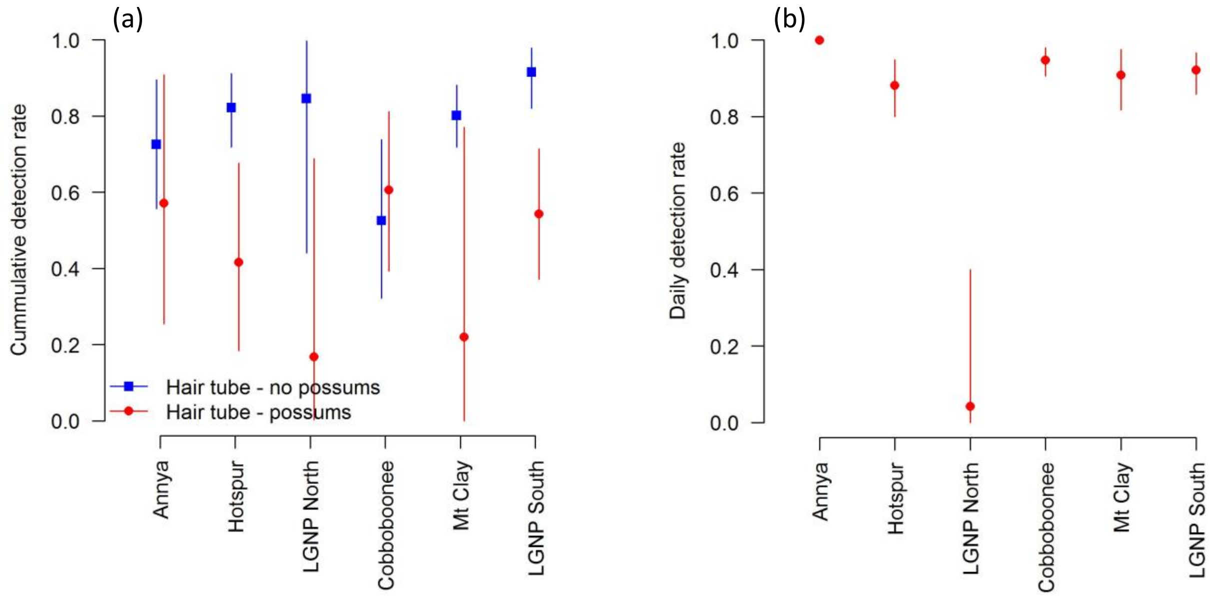
Location	Year	Both	Camera only	Hair-tube only	Neither	Total	Naïve occupancy rates
Annya	2013	3	6	1	30	40	0.250
Cobboboonee	2013	0	4	0	31	35	0.114
Hotspur	2013	1	4	1	30	36	0.167
Mt Clay	2013	0	3	0	29	32	0.094
LGNP-north	2013	0	0	1	30	31	0.032
LGNP-south	2013	0	5	1	33	39	0.154
Annya	2014	3	1	0	36	40	0.100
Cobboboonee	2014	0	7	1	27	35	0.229
Hotspur	2014	1	4	0	31	36	0.139
Mt Clay	2014	0	3	0	29	32	0.094
LGNP-north	2014	0	0	0	31	31	0.000
LGNP-south	2014	0	2	0	37	39	0.051
Total		8	39	5	374	426	

Cumulative hair-tube detection rates for Southern Brown Bandicoots varied between some locations and according to the presence/absence of Common Brushtail Possums (Table 5 and Fig. 9a). There was strong evidence that at Hotspur, Mt Clay and LGNP-south, detection rates were reduced when Common Brushtail Possums were present. LGNP-north had a similar result, but without enough evidence to be convincing. Cumulative camera-trap detection rates for Southern Brown Bandicoots varied between locations. Annya had higher detection rates than the other locations, whereas LGNP-north was lower than the other locations (Fig. 9b).

Table 5. Differences in hair-tube detection rates for Southern Brown Bandicoots at sites with possums detected and sites without possums detected. Blue shading highlights where there was substantial evidence that the detection probabilities were smaller when Common Brushtail Possums were detected. HDI=Highest Density Interval (similar to the credible interval in frequentists statistics).

Location	Median	Lower 95% HDI	Upper 95% HDI
Annya	–0.083	–0.274	0.123
Hotspur	–0.220	–0.352	–0.087
LGNP North	–0.266	–0.391	0.002
Cobboboonee	0.037	–0.104	0.167
Mt Clay	–0.281	–0.434	–0.132
LGNP South	–0.298	–0.583	–0.023

Figure 9. Cumulative detection rates [using hair-tubes (a), or cameras (b)] for Southern Brown Bandicoots at the six monitoring locations in the Glenelg Ark area. Blue = no possums present; red = possums present.



3.3.2 Sites occupied in 2013–2014

Common Brushtail Possums

Overall, the combined camera and hair-tube model and the camera only model tended to provide higher and less variable estimates of occupancy. In 2013, the number of sites occupied by Common Brushtail Possums as estimated from the camera-only model were generally similar to the estimates obtained from the combined camera and hair-tube data, while results from the hair-tube model were generally lower. The exceptions to this were at Hotspur where the combined model estimate was higher than the camera only model (Fig. 10a). Estimates of site occupancy were higher based on all three model outputs at Cobobboonee and LGNP-south compared to Annya and Hotspur. The model estimates for locations combined indicated that at no fox control locations the combined estimate was best, with camera only estimates higher than hair-tube only estimates. At fox control sites, there was no difference between camera only and the camera and hair-tube combined model estimates, while hair-tube only estimates were significantly lower.

In 2014, across individual locations there was no significant difference between camera and hair-tube, and combined camera and hair-tube model estimates, with the exception of LGNP-north where camera model estimates were higher and Hotspur where the combined estimate was higher. When locations are combined, there was no difference between camera and hair-tube model estimates at no fox control sites, while the combined model approach provided significantly higher estimates of occupancy. At fox control locations, the same pattern was evident but the differences were not significant (Fig. 10b).

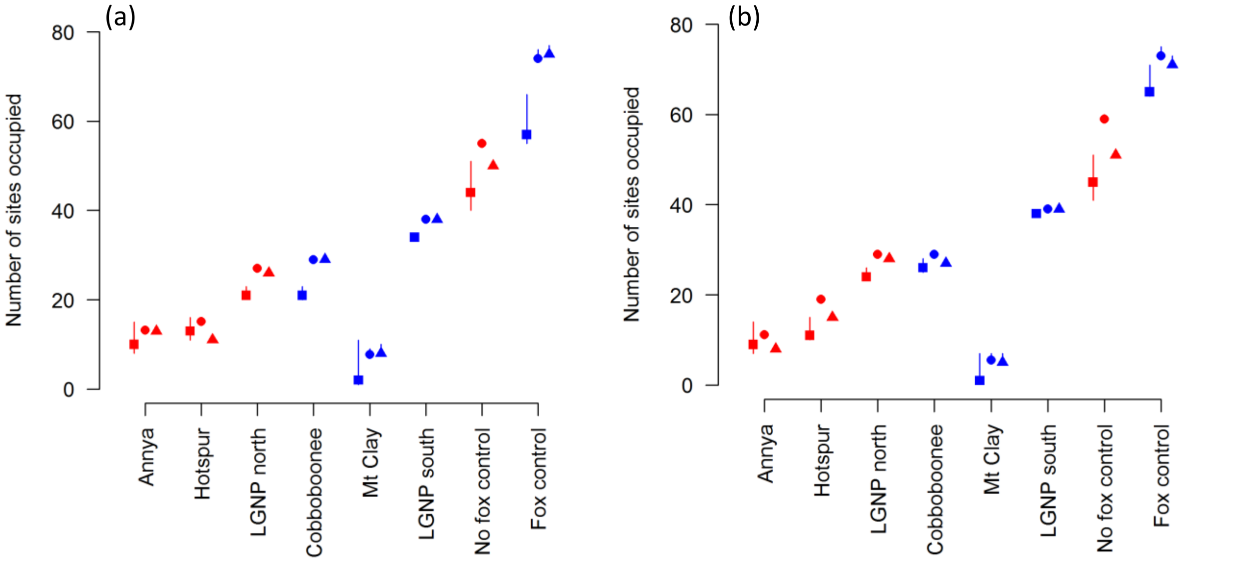


Figure 10. Estimated number of sites occupied by Common Brushtail Possums in 2013 (a) and 2014 (b): hair-tube data only (squares), camera-trap data only (triangles) or both combined (circles). Results sorted by location; no fox control = red; fox control = blue. Symbols represent the mean value, and bars represent the 95% density interval.

Long-nosed Potoroos

Overall there was little difference in occupancy estimates across all models in both years. In 2013, low levels of detection at Annya resulted in very large density estimates. At Mt Clay the camera only and the combined camera and hair-tube estimates were higher than for the hair-tube only model. There was some evidence that the Long-nosed Potoroo occupancy rate was higher overall at TMLs [camera only data - 0.151, 95% density interval (CI) 0.09, 0.22] compared with at NTMLs (camera only data - 0.071, 95% CI 0.27, 0.117) (Fig. 11a).

In 2014, camera only and combined camera and hair-tube model estimates were higher at LGNP-north, Mt Clay and LGNP-south than hair-tube only estimates. There was a significant effect from method on the estimated number of overall sites occupied in 2014 when comparing no fox control locations and fox control locations based on the camera only and combined camera and hair-tube only data.

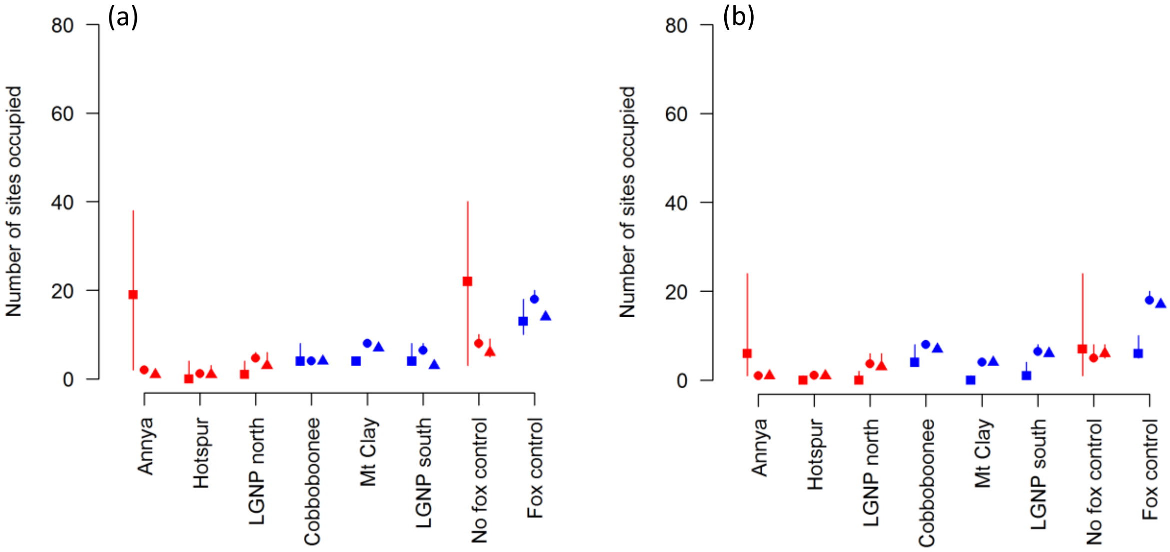


Figure 11. Number of sites occupied by Long-nosed Potoroos in 2013 (a) and 2014 (b) using either hair-tube data only (squares), camera-trap data only (triangles) or both combined (circles). Results sorted by location; no fox control = red; fox control = blue. Symbols represent mean values, while bars represent the 95% density intervals.

Southern Brown Bandicoots

Overall there was little difference in occupancy estimates across all models in both years. In 2013, low levels of detection at LGNP-north, on both hair-tubes and cameras, at Cobboboonee, Mt Clay and LGNP-south on hair-tubes resulted in very large density estimates (Fig 11a). At Annya and Hotspur the camera only and the combined camera and hair-tube estimates were higher than for the hair-tube only model. In 2013 there was some evidence from the combined model that the Southern Brown Bandicoot occupancy rate was lower overall at TMLs compared with at NTMLs but that this had reversed in 2014 (Fig. 11b). In 2014 there was little difference in occupancy across all models (Fig 11b).

Overall there was no detectable difference between the three methods in either 2013 or 2014, in part due to large density intervals for the hair-tube only data and at some locations camera only data which resulted from low levels of detection. In 2013, camera only and combined camera and hair-tube model estimates were higher only for Annya.

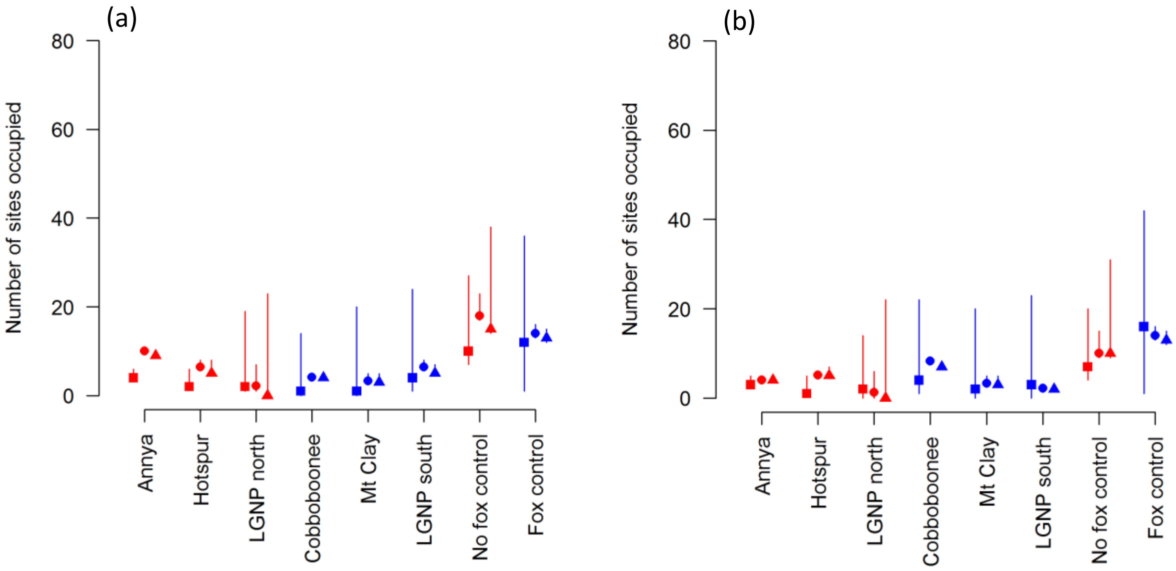


Figure 12. Number of sites occupied by Southern Brown Bandicoots in 2013 (a) and 2014 (b) using either hair-tube data only (squares), camera-trap data only (triangles) or both combined (circles). Results sorted by location; no fox control = red; fox control = blue. Symbols represent mean values, while bars represent the 95% density intervals.

3.3.3 Differences in occupancy at fox treatment and non-treatment locations

We investigates the probability that fox control influenced occupancy at the oval location level, i.e., occupancy at LGNP-south, Cobobbonee and Mt Clay) combined compared to the non-fox control locations combined.

In 2013, all three methods showed strong evidence for a positive fox control effect for Common Brushtail Possums; cameras only and the combined approach both provided strong evidence of a positive effect for Long-nosed Potoroos and inconclusive evidence for a positive effect for Southern Brown Bandicoots. In 2014, cameras only, and cameras and hair-tubes combined showed strong evidence of a positive effect on Common Brushtail Possums and Long-nosed Potoroos and inconclusive evidence of a positive effect on Southern Brown Bandicoots (Table 6). Hair-tubes alone failed to detect any difference in the level of average occupancy for any of the three species in either year. Cameras-only showed no evidence of an effect for Southern Brown Bandicoots in any year, or of an effect for possums or potoroos in 2014.

Table 6. Probability that the average occupancy estimates across all fox control locations combined were greater compared to combined non–fox control location. The green shading indicates when the proportion of average modelled occupancy estimates were higher for combined fox control locations than for the combined non-fox control locations. Where the proportion (from the 10 000 model iterations) was >0.95 it indicates strong evidence that fox control influenced occupancy.

Species	Year	Hair-tube only	Camera only	Combined
Common Brushtail Possum	2013	0.96	1.00	1.00
Long-nosed Potoroo	2013	0.28	0.95	0.98
Southern Brown Bandicoot	2013	0.51	0.26	0.23
Common Brushtail Possum	2014	0.80	0.73	1.00
Long-nosed Potoroo	2014	0.41	0.31	0.99
Southern Brown Bandicoot	2014	0.74	0.53	0.76

These results are generally in line with the site location specific outcomes, with the exception of the camera only results for Long-nosed Potoroos in 2014. The difference arises from the large turn-over rate in site specific occupancy. While the number of sites occupied is similar, the actual sites change-over time, increasing the uncertainty in occupancy rates.

3.3.4 Costs of implementing methods in the field

The cost of implementing the hair-tube-only method would be 25% greater than that of implementing the camera-only method (Table 7). There would be no significant cost savings in implementing both methods simultaneously because hair-tubes need to be checked and tapes changed every 4 days, whereas cameras are set out for 30 days and tasks cannot be combined to gain efficiency.

Table 7. Relative costs per year of implementing hair-tube and camera-trap surveys in 2013/2014.

Task	Person-days	Cost (2014 dollars)
Hair-tubes		
Set up and pick up	24	$43,200
Assemble and process tapes	12	$3,600
Identification of hairs		$7,650
Total		**$54,450**
Cameras		
Site preparation	6	$10,800
Set up cameras	6	$10,800
Pick up cameras	6	$10,800
Organise equipment	3	$900
Sort images	6	$7,200
Total		**$40,500**
Difference		**$13,950**

3.4 Changes in the number of sites occupied 2006–2015

3.4.1 Common Brushtail Possums

The overall number of sites occupied by Common Brushtail Possums has doubled since 2005 and was higher at locations with fox control (Fig. 13). The number of sites occupied increased at the TMLs and NTMLs in 2007; however, possums declined in 2011 and recovered to 2007 levels in 2012, before declining at NTMLs in 2015. At the TMLs, site occupancy has increased from 2011 and has remained high.

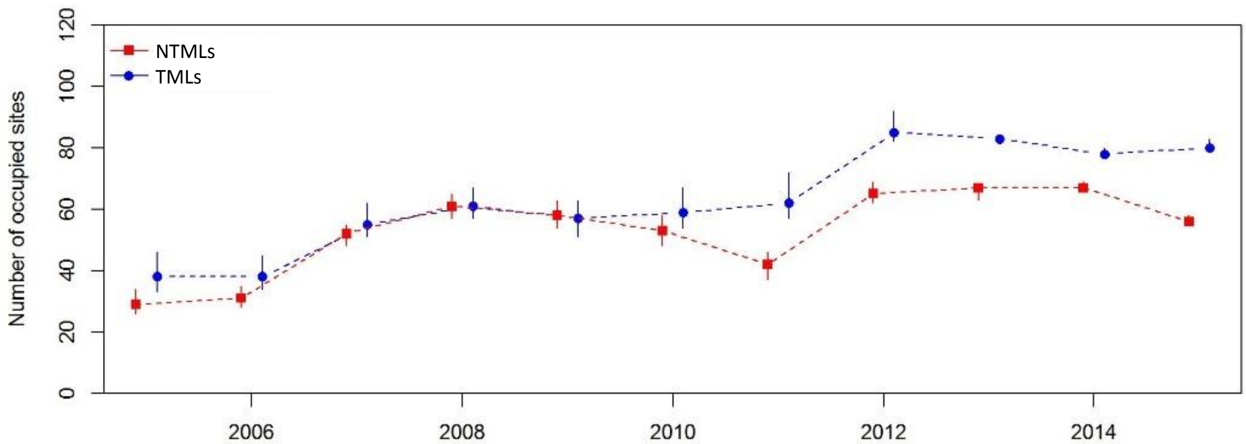


Figure 13. Estimated number of sites occupied by Common Brushtail Possums over time at TMLs and NTMLs.

The number of sites occupied by Common Brushtail Possums varied with location, and while possums generally increased at both TMLs and NTMLs, the increases were greater at Cobboboonee and LGNP-south (Fig. 14). The number of sites occupied at Mt Clay has remained constant and low, indicating that perhaps the Mt Clay sites are poor-quality habitat for Common Brushtail Possums. The number of sites occupied has declined at Annya since 2012, which would contribute to the observed difference in site occupancy.

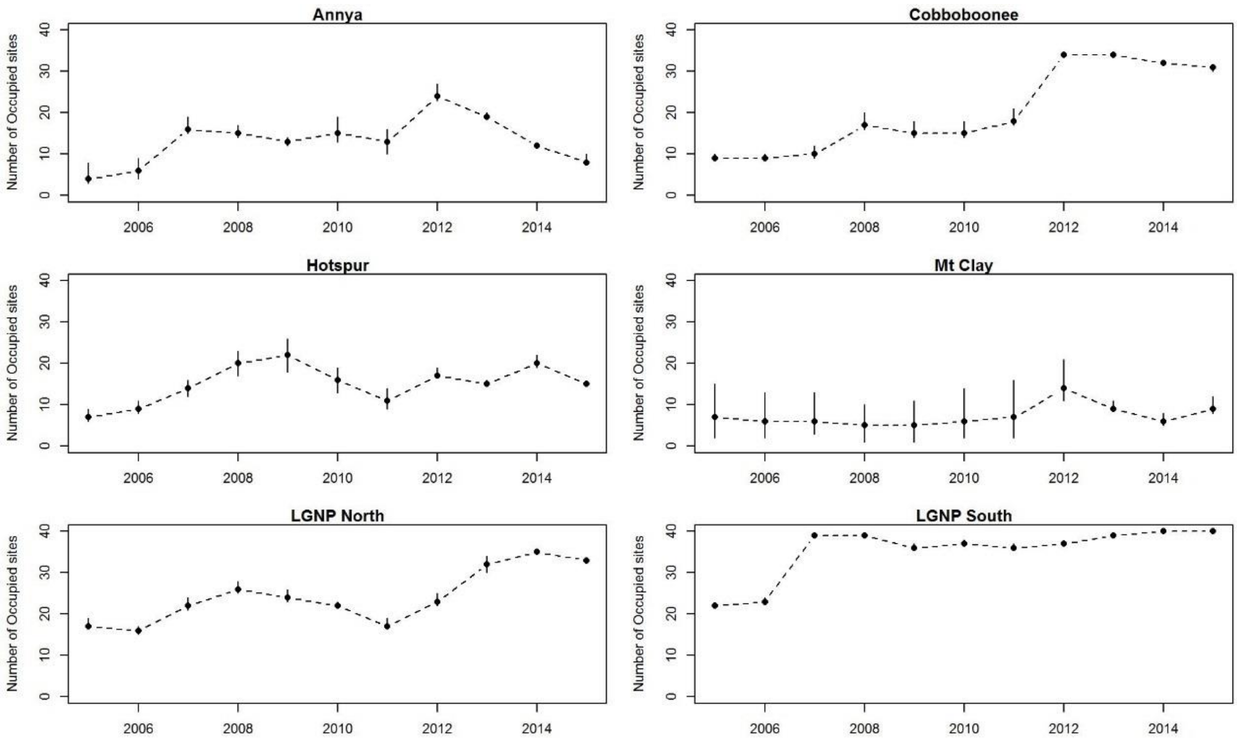


Figure 14. Estimated number of sites occupied by Common Brushtail Possums in each region over time. Dots indicate the medians and the bars represent the 95% high-density intervals. Left panels – NTMLs, right panels- TMLs.

3.4.2 Long-nosed Potoroos

The number of sites occupied by Long-nosed Potoroo across locations with fox control doubled in 2006, immediately after fox control commenced before declining to a low in 2008; it has remained relatively higher than in 2005 compared with locations with no fox control. Across locations with no fox control, the number of sites occupied has steadily declined (Fig. 15).

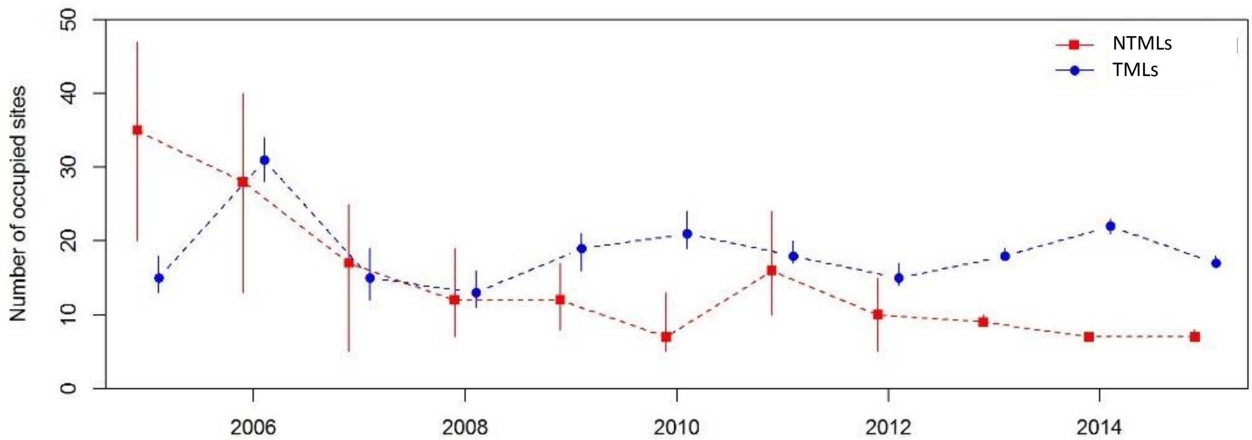


Figure 15. Estimated number of sites occupied by Long-nosed Potoroos over time at TMLs and NTMLs.

Individual locations show variation in the number of sites occupied by Long-nosed Potoroos over time (Fig. 16). Initial increase in the number of sites occupied occurred at all three locations with fox control, with the biggest increase at LGNP-south; however, Long-nosed Potoroos then declined to below initial levels at this site and remained low. The number of sites occupied at Cobboboonee has increased, and at Mt Clay they are at the same level in 2015 as they were in 2005. At both Annya and LGNP-north, the number of sites occupied is the same in 2015 as in 2005, while at Hotspur the number of sites occupied has steadily declined.

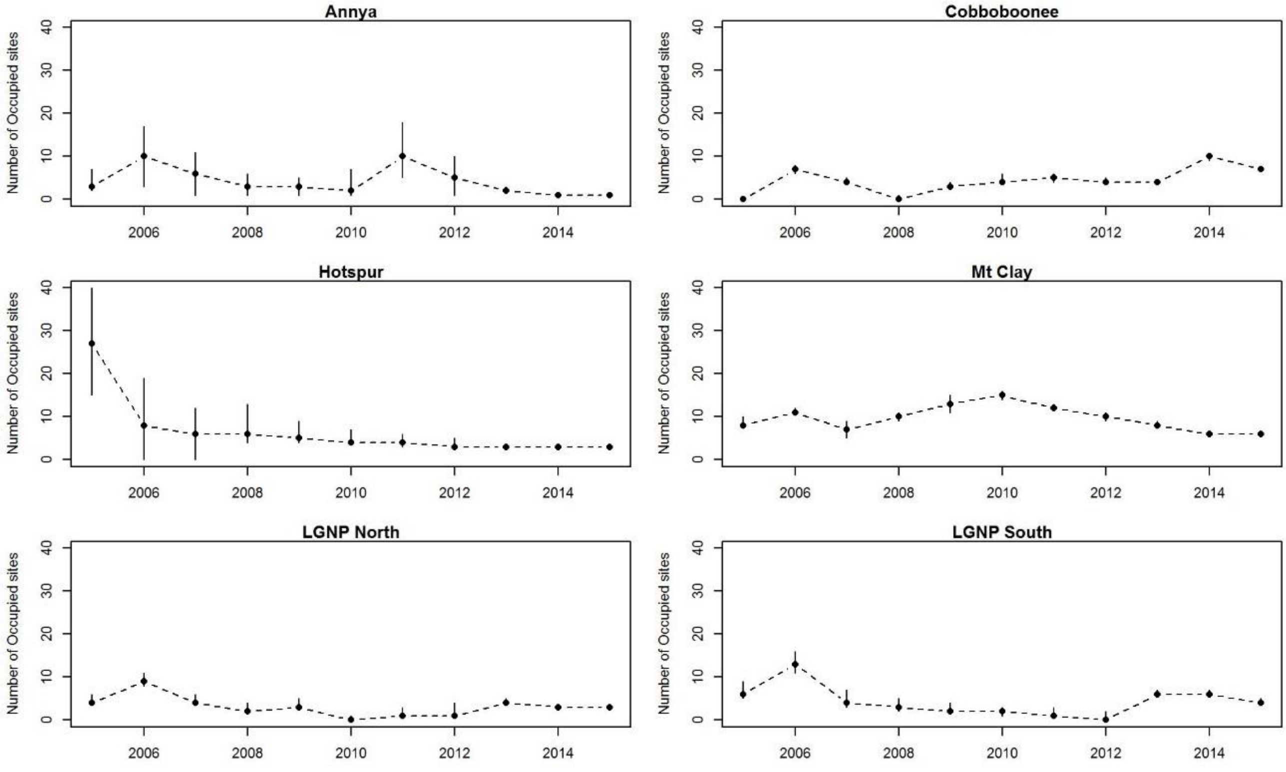


Figure 16. Estimated numbers of sites occupied by Long-nosed Potoroos in each region over time. Dots indicate the medians and the bars represent the 95% high-density intervals. Left panels – NTMLs, right panels- TMLs.

3.4.3 Southern Brown Bandicoots

The number of sites occupied by Southern Brown Bandicoots across locations with fox control doubled in 2006 and remained higher than the 2005 levels, until a sharp decline in 2011. The number of sites occupied across locations with fox control was higher in 2015 compared with in 2005, whereas across locations with no fox control, the number of occupied sites has remained relatively constant (Fig. 17).

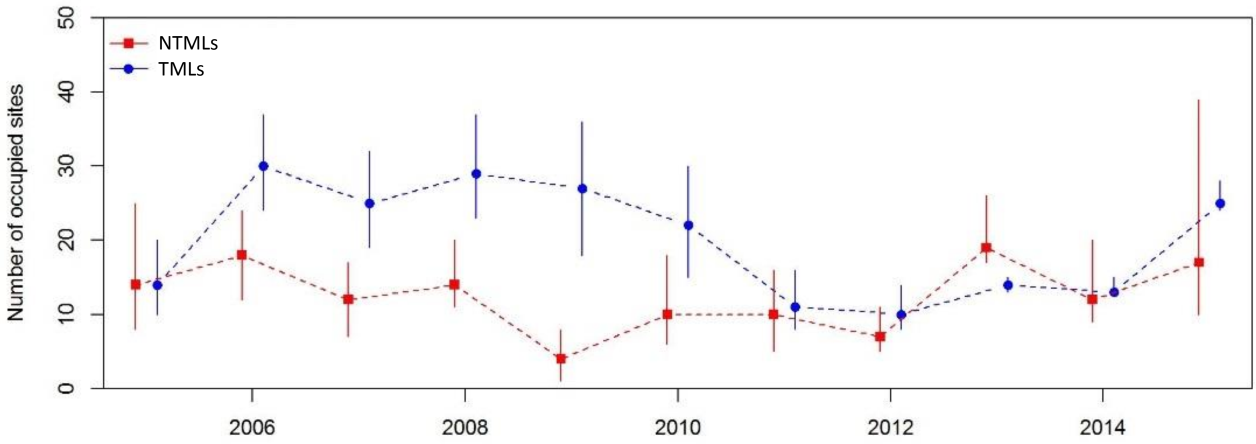


Figure 17. Estimated number of sites occupied by Southern Brown Bandicoots over time at TMLs and NTMLs.

The number of sites occupied by Southern Brown Bandicoots at individual locations showed moderate levels of variation through time (Fig. 18). At NTMLs, site occupancy remained constant from 2005 to 2015. Southern Brown Bandicoots at LGNP-north nearly became locally extinct between 2007 and 2012, before a slight increase till 2015. At TMLs, Cobboboonee is the only site that shows some indication of a positive increase in site occupancy. At LGNP-south and Mt Clay, the number of sites occupied was moderate (between 10 and 15 sites, respectively), before a steady decline at Mt Clay from 2010. At LGNP-south, the site occupancy declined from 2008, but it recovered in 2015 to 2005 levels.

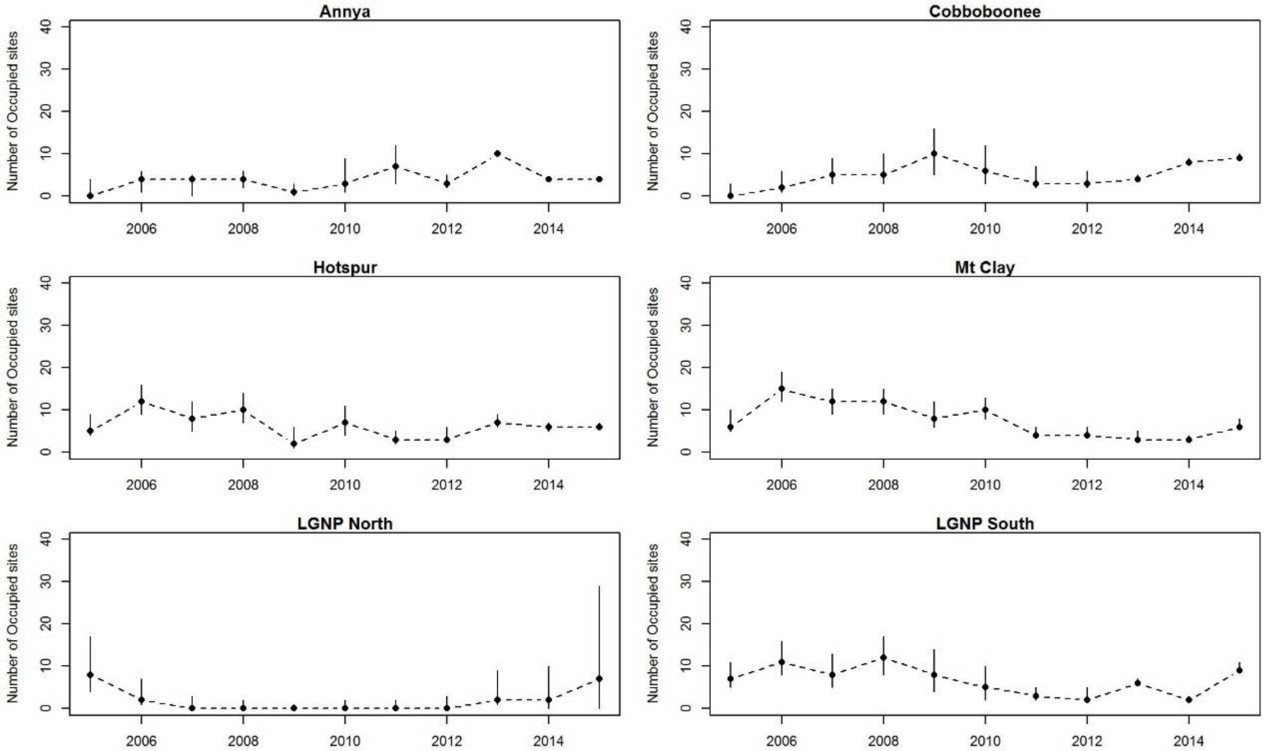


Figure 18. Estimated number of sites occupied by Southern Brown Bandicoots in each region over time. Dots indicate the medians and the bars represent the 95% high-density intervals. Left panels – NTMLs, right panels- TMLs.

4 Discussion

At 3 years since the last assessment of the Glenelg Ark project, native species site occupancy remains higher at locations with fox control than it was at locations with no fox control in 2005. The project has also successfully transitioned from using hair-tubes to using digital cameras as the main monitoring tool.

The comparison of numbers of sites occupied for locations with and without fox control assumed that individual locations were ecologically similar. However, individual locations showed a degree of variation, suggesting that conditions (habitat, underlying predator density, etc.) were not uniform. This was supported by the detection rates for each site in 2013 and 2014. Differences in detection rate have suggested that there were location differences affecting detection rates, or that hair-tubes and/or cameras detect species differently at different locations. What the underlying condition differences might be and just how they might act to affect native species abundance is currently not known. The number of sites occupied by Common Brushtail Possums at LGNP-south doubled in 2007, possibly as a result of declines in underlying fox abundance, while at Cobboboonee the same level of change did not occur until 2012and appeared to be due to the previous 2 years (2010-11) of above-average rainfall (BOM 2016). Long-nosed Potoroos at all three TMLs showed signs of a positive response in 2006, with increase and decrease through time at Cobboboonee and Mt Clay, and a general decline at LGNP-south.

The long-term dataset shows that, compared with 2005, the number of sites occupied by all three native species pooled across TMLs is higher than at sites pooled across NTMLs. This trend was also supported by the 2013–2014 camera data, and the combined camera and hair-tube assessment, which both showed strong evidence of an increase in Common Brushtail Possums and Long-nosed Potoroos from 2013 to 2014. Bandicoot response in 2013 and 2014 was inconclusive, and it has only been in 2015 that an increase in site occupancy has occurred, notably at LGNP-south.

The comparison of hair-tubes and cameras revealed that, at some sites, either hair-tubes or cameras failed to detect the species of interest, even though it was known to be present; thus, a combination of methods may prove to be the 'best' approach. However, the resources required to undertake this approach would be unsustainable. Cameras are the more efficient and cost-effective option of the two methods (being cheaper to operate and providing a broader range of species detections, including detection of foxes and feral Cats), whereas hair-tubes (as were used in this study) are limited to detecting medium-sized (mostly) ground foraging species. To our knowledge no other study has investigated the differences in detection rates and occupancy estimates between camera traps and hair-tubes.

The monitoring program has focused on the changes in three medium-sized mammal species in response to a reduction in foxes across the landscape. Other species that are present in the Glenelg Ark area, in particular smaller mammals such as the Heath Mouse (*Pseudomys shortridgei*), may also respond to fox control. The Heath Mouse is a small endemic rodent restricted to heaths and heathy woodlands in southern Australia (Menkhorst 1995). A substantial part of the Heath Mouse distribution in Victoria occurs within the Glenelg Ark operations area. Population responses of Heath Mouse are currently not monitored within Glenelg Ark, in part because there has been no standard survey protocol. Development of a monitoring protocol for the Heath Mouse is underway (R. Hill pers. comm.); when finalized it should be implemented to assess the relative status of this species across TMLs and NTMLs.

The camera surveys clearly indicate a difference in fox activity between the treated and non-treated areas. However, there are three issues that should be investigated:

(i) Can foxes be reduced further with improvements to the control operation?

(ii) The relationship between activity (the number of camera images/day) and abundance is assumed to be linear, i.e., a unit decline ion activity is linearly related to a unit decrease in actual abundance; however, this is almost certainly not the case. Thus, while fox activity may have decreased, it remains unclear what effect this has had on the abundance of foxes. This lack of understanding clouds the interpretation of the native species monitoring results and the interpretation of the effectiveness of the fox control strategy employed at Glenelg Ark.

(iii) A third issue is the possible response of feral Cats to the reduction in foxes. The mesopredator release hypothesis (sensu Soulé et al. 1988) predicts that when a dominant predator is reduced in abundance, this allows a population increase in lower-order predators that results in an increase in predation on shared prey species. It is possible that by reducing foxes, an increase in feral Cats could replace or increase predation pressure on potoroos, bandicoots and other shared prey items in the Glenelg Ark area. This possibility requires further study.

Bait density is a critical issue in fox control operations, and specific knowledge on what density is required to significantly reduce fox populations is still lacking, despite decades of investment in fox control across Australia. Saunders and McLeod (2007) suggest that bait density should exceed fox density, and that under most Australian conditions a bait density of 5–10 baits per square kilometre is sufficient. Bait density during the current operation was much less than this (mean 0.85 baits/km^2, S.D. = 0.12).

Increasing the frequency of bait replacement can also increase the baiting program efficacy. Thompson and Fleming (1994) and Fleming (1996) achieved fox population reductions of 70% and 91%, respectively, over 9–14 days using daily bait replacement. While this level of replacement would be cost-prohibitive in long-term operations such as Glenelg Ark, increasing replacement rates to twice per week, or at least once per week may be achievable compared to the once a fortnight as currently done; in combination with increased bait density, this could increase the level of fox population reduction.

Individual-based, spatially explicit, stochastic population models can be used to investigate the impact of a variety of management strategies for fox control [such as with a range of bait densities, frequency of bait replacement and timing of baiting (continuous, seasonal or periodic)] and their effects on native species recovery (e.g. Bonesi et al. 2007). The success of fox control programs is likely to be influenced by the life-history traits and the space-use patterns of foxes and their prey. Rushton et al. (2006) used this approach to assess the impact of culling of a fox population in Britain and found that managing immigration was a key factor in the success or failure of fox control. Conner et al. (2008) evaluated coyote management strategies, also using this approach, and found that spatially intensive removal was more efficient and long-lasting than random removals. The proposed model approach can explore various management scenarios in a spatial context by varying life history parameters and management strategy and the outputs assessed in order to determine the most cost-effective strategies.

Glenelg Ark has been in operation for 15 years, and the initial response of the targeted native mammal species indicates that they responded positively to the reduction of foxes. It could be that these species have reached a new equilibrium with a lowered level of background fox predation and that a different factor (e.g. food, habitat, predation by feral Cats) may now be the key limiting mechanism. A number of potential scenarios flow from this assumption.

First, to test this assumption if the remaining fox populations could be further reduced, then further increases in growth rates in the mammal population might occur, in turn increasing the number of sites occupied. Reviewing the current fox control strategy and exploring options aimed at further reducing the fox population are warranted.

Second, removing fox predation as the primary limiting factor may have resulted in an overabundance of native herbivores. For example, Common Brushtail Possums occupy >80% of sites, and wallaby species are anecdotally reported to have increased substantially in the last 10 years. This may lead to over browsing, thus changing the composition and structure of the habitat. Dexter et al. (2013) found that over browsing by an overabundance of wallabies (which resulted from fox control operations) caused a shift in the vegetation community structure at Booderee National Park in New South Wales. This is an issue requiring further investigation.

Third, within the Glenelg Ark operations area, the sustained reductions in fox populations may have resulted in increased activity (and possibility abundance) of feral Cats. While camera surveys indicated that there was no significant difference in feral Cat activity between treated and non-treated locations, the point estimates indicated a higher level of activity at TMLs. Several studies have described increases in feral Cat abundance following reductions in fox numbers resulting from fox control operations (Algar and Smith 1998; Catling and Reid 2003) and following local declines in Dingo (*Canis lupus dingo*) abundance in Queensland (Pettigrew 1993). Catling and Burt (1995) have also reported that the abundance of feral Cats is negatively correlated with both foxes and Dingoes at a site in New South Wales. Read and Bowen (2001) did not manipulate predators, but reported that feral Cat abundance peaked when fox numbers were low and when rabbit numbers were relatively high.

There is a critical need for a better understanding of how feral Cats respond to fox control operations in mesic habitats in south-eastern Australia. Using the Glenelg Ark ongoing management initiative, it would be possible to undertake targeted camera surveys to assess differences in feral Cat activity (and possibly abundance) between locations with and without fox control.

Feral Cat management strategies in Victoria are limited by the current Victorian legislative framework. For example, there are limitations on being able to destroy feral Cats on public land without presenting them first to a local council, on being able to deploy toxic baits from the air or to surface-lay baits, on being able to capture and destroy feral Cats in leg-hold traps, and on being able to capture and release feral Cats (for research); these limits restrict the capacity of public land managers to develop and implement effective management practices in Victoria.

The Glenelg Ark monitoring program has continued to operate effectively, providing information to land managers and to DELWP and Parks Victoria policy groups on the response of the targeted native mammal species. It has adopted new approaches to monitoring, and is providing insights into other factors that may contribute to the longer-term sustainability of the target species and of other components of the ecosystem. Glenelg Ark is in a strong position to adapt its focus in the light of these insights. The project also provides the framework and infrastructure in which other management-focused research questions can be addressed (e.g. small mammal response, the impact of possible unintended consequences such as over-browsing, and changes in the feral Cat population).

Recommendations

The following recommendations are made for improvement of the outcomes of Glenelg Ark.

Item	Recommendation	Detail
Native species' response	Move to using digital cameras as the main monitoring tool for native species.	There was no overall significant difference in detection rates between cameras and hair-tubes; however, cameras are less costly to operate and are able to 'capture' a wider range of species in all weather conditions.
	Develop bandicoot and potoroo habitat suitability surfaces for the Glenelg Ark project area using presence/absence data to aid in setting species response targets and potential new control and / or monitoring sites.	The limited response of bandicoots and potoroos may be due to a lack of suitable habitat for these species. We propose that the site occupancy information be used to explore the possible limitation of suitable habitat. This data combined with freely available remotely sensed habitat data (e.g., vegetation type, topography, fire history, distance to drainage lines, forest edge) can be combined with information on detection and non-detection of species at sites to develop a species habitat suitability surface across the project area. This information will be useful in understanding the expected increase in species occurrence and also identify potential new locations for monitoring and or fox control actions.
	Using species distribution models of the benefits of fox control for the Heath Mouse, select sites for targeted monitoring on treatment and non-treatment locations.	Current monitoring sites were placed in locations based on Ecological Vegetation Divisions mapping and the best understanding of 'suitable' habitat at that time. Predictive species distribution models that incorporate the likely benefit of fox control have been developed in recent years. These could be used to select sites more likely to have the Heath Mouse present. If fox control has delivered a positive benefit, there should be a detectable difference between treated sites and non-treated sites.
Fox control	Review the predator control program and investigate options for improving where needed.	Bait density and the frequency of bait replacement, as well as bait type and placement can affect the outcome of fox control. A general review of the program with consideration of the items above is warranted.
	Use spatially explicit individual-based population models of the reduction in foxes from control operations to develop strategies for increased reduction in fox populations.	Use existing empirical data and expert elicitation to develop models testing a range of baiting scenarios in order to assess their impact on fox abundance. Despite decades of fox control, we have little understanding of what the best strategy is for reducing and maintaining lower fox abundances.
Differences in fox and feral Cat abundances across treatment/non-treatment areas	Undertake camera monitoring specifically to assess the effectiveness of the control operation, and use the information to assist in the development of an integrated feral Cat and fox control strategy.	Determine the number of camera sites required through a power analysis to assess differences (if they exist) in fox and feral Cat activity on treated and non-treated sites.
Alternative survey methods for foxes and feral Cats	Assess and cost the feasibility of genotyping DNA from fox scats collected using scat detector dogs.	Scat detector dogs and genotyping DNA from scats have both been used successfully to enumerate fox populations before and after fox control. A similar approach could be used in Glenelg Ark to assess differences between baited and comparable unbaited areas.
Scientific support	Continue to source scientific support and advice concerning the ongoing implementation and development of Glenelg Ark.	Evaluation and interpretation of monitoring data, development of new projects addressing emerging issues, and general guidance to the project has been essential to its success.

Item	Recommendation	Detail
Monitoring and reporting	Continue annual monitoring, evaluation and reporting.	Continue annual monitoring and reporting in order to closely track changes in predators and prey, thus allowing more responsive management of emerging issues, e.g. a decline in Southern Brown Bandicoots; a change in feral Cat abundance.
Filling specific knowledge gaps	Develop a set of potential student projects to fill identified knowledge gaps.	The current monitoring program does not assess changes in small native mammals (e.g. Heath Mouse and White-footed Dunnarts), or unintended consequences (e.g. the possible negative impacts on biodiversity of overabundant medium- and small-sized herbivores, e.g. wallabies and Common Brushtail Possums). A series of student projects aimed at filling these knowledge gaps and taking advantage of the infrastructure that Glenelg Ark provides would be possible.

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Appendices

Appendix 1. Fox and feral Cat activity model output

Table A1.1. Bayesian non-linear mixed model parameter estimates for fox activity. Family: negative binomial (log). Formula: success ~ (treat – 1) * year + offset(log(trials)). Number of observations: 672; samples: 4 chains, each with iteration = 2000; warm-up = 1000; thin = 1; total post–warm-up samples = 4000. ll-95% CI = Lower 95% confidence interval; u-95% CI = upper 95% confidence interval.

Fixed effects	Estimate	Estimate error	l-95% CI	u-95% CI
Fox control	–4.5	0.16	–4.82	–4.2
No fox control	–2.15	0.06	–2.26	–2.04
2014	0.08	0.22	–0.33	0.52
2015	0.08	0.21	–0.34	0.5
No fox control: 2014	–0.14	0.23	–0.6	0.28
No fox control: 2015	–0.33	0.22	–0.79	0.12
Family-specific parameters:				
Shape	18.82	10.13	9.09	42.89

Table A1.2. Bayesian non-linear mixed model parameter estimates for Cat activity. Family: negative binomial (log). Formula: success ~ (treat – 1) * year + offset(log(trials)). Number of observations: 672; samples: 4 chains, each with iteration = 2000; warm-up = 1000; thin = 1; total post–warm-up samples = 4000. ll-95% CI = Lower 95% confidence interval; u-95% CI = upper 95% confidence interval.

Fixed effects	Estimate	Estimate error	l-95% CI	u-95% CI
Fox control	–4.34	0.15	–4.66	–4.05
No fox control	–4.75	0.19	–5.13	–4.39
2014	–0.59	0.24	–1.07	–0.11
2015	–0.68	0.24	–1.16	–0.21
No fox control: 2014	0.38	0.36	–0.32	1.1
No fox control: 2015	0.77	0.34	0.09	1.42
Family-specific parameters:				
Shape	4.01	3.25	1.25	12.23

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