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Front cover photos: (a) Red Fox with native prey. (b) Feral Cat inspecting lure. (c) 1080 baiting sign at entrance to Glenelg Ark. (d) Monitoring native species' response. (e) Laying baits for fox control. (f) Southern Brown Bandicoot. (g) Long-nosed Potoroo. (Photographer: DELWP).

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Summary

Context:

The Glenelg Ark project was established in 2005 to facilitate the recovery of selected native mammal species considered at risk from fox (Red Fox, *Vulpes vulpes*) predation. The project established continuous landscape-scale fox baiting across 90 000 ha of State Forest and National Park in south-western Victoria. Three native mammal species that were present in the project area at the time (in low numbers, with patchy distributions, and 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 adds new data to the outcomes of the fox control operation and the responses of targeted native species for 2018 to the previous information from 2005-2017. It incorporates fox population modelling as a measure of current fox densities across the Glenelg Ark operations area. This report also contains recommendations for future management options and suggests areas of further research.

Aims:

The aims of this Glenelg Ark project report are to update the long-term predator and native mammals' response dataset, to update the fox population model predictions of fox density, and to provide information to land managers and policy groups to inform decision-making regarding future directions.

Methods:

Differences between fox and feral Cat (*Felis catus*) activity (number of images per 3 hours pooled across 24 hours) at locations with and without fox control [i.e. at treatment monitoring locations (TMLs) and non-treatment monitoring locations (NTMLs)] were assessed using Bayesian regression models based on the number of independent images captured on camera traps from 2013 to 2018. Current fox density was predicted using an individually based, spatially explicit population model.

The response of the three native mammal species to the reduction in foxes was examined using multiseasonal occupancy models, with detection/non-detection data from 240 camera traps from three TMLs and three NTMLs.

Results:

Fox activity was 88% higher across the NTMLs [$\bar{x}$ = 3.6, confidence interval (CI) 3.2–4.1] than across TMLs ($\bar{x}$ = 0.4, CI 0.4–0.5). Fox density was predicted to be 72.6% lower across the Glenelg Ark operations area compared with pre-baiting densities, and this was achieved after 3 years of simulated baiting and was predicted to remain low under the current management strategy. Between 2013 and 2018, there was no difference in feral Cat activity between NTMLs [$\bar{x}$ = 0.25, standard deviation (SD) 0.74] and TMLs ($\bar{x}$ = 0.25, SD 0.70).

Long-nosed Potoroo occupied only a small number of the 120 surveyed sites across the three TMLs after 13 years of fox control. On average Long-Nosed Potoroos occupied 19 (CI 16–21) sites on TMLs compared to 12 (CI 10-15) on NTMLs. Since 2011 potoroos have declined on NTMLs from 13 sites (CI 10–17) in 2011 to 4 sites (CI 4–5). At the same time the number of sites occupied on TMLs has remained relatively constant ($\bar{x}$=17, CI 16–18).

Since baiting began in 2005, Southern Brown Bandicoot have on average occupied more sites ($\bar{x}$ = 22, CI 18-27) on TMLs compared with NTMLs ($\bar{x}$ = 13, CI 9-16). Southern Brown Bandicoot responded positively to fox control in the immediate post-baiting period. Between 2006 and 2009, Southern Brown Bandicoot occupied on average 30, CI 22–38) compared to 13 sites (CI 8–18). However, from 2010 to 2014 bandicoots declined to the point where there was no difference between TMLs and NTMLs. Then between 2015 and 2018,

Southern Brown Bandicoot occupied more sites on TMLS ($\bar{x}$ = 25, CI 23–28) than on NTMLS ($\bar{x}$ = 11, CI 10–13).

Overall Common Brushtail Possums occupied the most sites and showed a clear difference in occupancy Since 2005, Common Brushtail Possum have on average occupied 63 (CI 62–64) or 52% of sites across the TMLs compared with 51 (CI 50–54) or 42.5% of sites across NTMLs. Since 2010, Common Brushtail Possums have occupied on average 16 (CI 14–18) more sites on TMLs compared with NTMLs.

Conclusions and implications:

The data strongly indicate that the fox population within the Glenelg Ark operations area has been significantly reduced because of the ongoing and continuous baiting program. The fox model also suggests that any reduction in the baiting effort could easily result in a rapid loss of the gains made to date.

The lack of a significant response in Southern Brown Bandicoot and Long-nosed Potoroo is of concern. After 13 years of fox control, if the fox population has been reduced to levels low enough to allow for population growth in these species, it would be reasonable to expect a greater level of response. There are some possible reasons for why this has not occurred that are worthy of exploration.

1. The current sampling methodology is insensitive to the scale of change in occupancy, or occupancy is a poor metric for abundance; thus, the result is an underestimate of the scale of change that has occurred across the landscape.
2. Fox densities are still too high and limit the population growth of fragmented small populations of Southern Brown Bandicoot and Long-nosed Potoroo.
3. Feral Cats have replaced foxes as the main predator, with potentially similar dynamics to those of foxes.
4. Landscape disturbance, e.g. the long-term effects of frequent burning (both planned and natural), has resulted in a highly fragmented landscape. Populations are now restricted to isolated refugia, and species are unable to bridge the gaps between them.
5. Some combination of the above.

To address these possible causes and to more fully explore the issues and to fill knowledge gaps detailed recommendations are provided.

1 Introduction

The Glenelg Ark project was established in July 2005 to facilitate the recovery of selected native mammal populations considered at risk from fox (Red Fox, *Vulpes vulpes*) predation. The project established continuous landscape-scale fox baiting across 90 000 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 (1) the response of foxes to control activities, and (2) the response of native species that are at risk from fox predation to a reduced abundance of foxes. Without such monitoring and evaluation, it would be difficult to justify the reinvestment of scarce public conservation funds, to improve management actions based on scientific information, and to maintain community support. Thus, monitoring and evaluation forms an essential part of management and is not an imposition on it or an 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 also thought to be at risk from fox predation were selected for monitoring: the Southern Brown Bandicoot (*Isoodon obesulus*), the Long-nosed Potoroo (*Potorous tridactylus*) and the Common Brushtail Possum (*Trichosurus vulpecula*). The Southern Brown Bandicoot (weight ~1.0 kg) and the Long-nosed Potoroo (weight ~1.2 kg) are medium-sized ground-dwelling mammals that have high and moderate rates of fecundity, respectively (Menkhorst and Knight 2010). Both species are known to be preyed upon by foxes (Seebeck 1978) and have been reported to respond positively to a reduction in foxes (Kinnear et al. 2002; Arthur et al. 2012). The Common Brushtail Possum is a semi-arboreal species that weighs ~3.0 kg, has a low rate of fecundity (Kerle and How 2008), and is known to be eaten by foxes (Triggs et al. 1984) and to respond to fox control (Kinnear et al. 2002).

Foxes have played a role in the decline and extinction of Australian mammals (Short and Smith 1994; Salo et al. 2007), and there are examples of mammal recovery following sustained reduction in fox abundance (McLeod et al. 2008). Feral Cats (*Felis catus*) are also known to be present in the Glenelg Ark operations area. Feral Cats have been implicated in the extinction of 19 species and the catastrophic declines recorded across north-Australia (Woinarski et al. 2011; Fisher 2015; Ziembicki et al. 2014). In Victoria, there are 43 species listed under the *Flora and Fauna Guarantee Act 1988* (FFG Act) or the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) as threatened by feral cat predation. It is probable that, today, populations of the target species can only persist in refugia that may be quite atypical of the species' preferred niche. In terms of the Hutchinson (1978) concept of the niche (see also Kinnear et al. 1985), fox and feral Cat predation affects the dimensions of a species' realised niche by exaggerating the requirements for protective shelter and for food to be nearby. Niche theory predicts that a release from predation would relax the requirements for shelter and proximity of food, and thus permit expansion of the realised niche, i.e. there would be greater breadth of habitat when foxes and feral Cats were controlled.

In either case, there are strong theoretical reasons to believe that if fox predation is the key limiting factor, significantly reducing or removing foxes should result in a strongly positive response in habitat use or abundance. Based on 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 the limitation imposed by predation, and the number of sites occupied by the targeted prey species should increase.

We assessed changes in foxes and feral Cats by comparing their activity (number of independent images captured by a digital camera at a monitoring site) at locations having an ongoing history of continuous fox control (with fortnightly replacement baiting) with that at locations having no history of fox control. We include monitoring of feral Cats to establish baseline data and to provide information that could be used help managers when a broadscale control tool becomes available. We also used a spatially explicit, agent-based fox population model to compare pre and post-baiting fox densities. We assessed the responses of the 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 the native species to the reduction in fox abundance at sites in Glenelg Ark was assessed each spring, using detections resulting from species contact with hair-tubes from 2005 to 2012, and using detections by digital cameras from 2013 to 2018.

We examined the differences (if any) in the occupancy and detection estimates of Common Brushtail Possums, Long-nosed Potoroo and Southern Brown Bandicoot from 2005 to 2018 at the six monitoring locations within the Glenelg Ark project area.

This report updates the previous monitoring and evaluation report (Robley et al. 2018) by incorporating new data on the outcome of the fox control operation, and modelled estimates of fox density and of the response of the targeted native species from 2013 to 2018. This report also contains recommendations on future management options and suggests areas of further research. The outcome is that land managers, policy-makers and the community can now make informed, evidence-based assessments of the success of broad-scale mainland fox control operations and have the information necessary for decision-making about future directions.

2 Methods

2.1 Glenelg Ark operations area

The Glenelg Ark operations area is in far south-western 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 the average minimum and maximum temperatures are 8.1°C and 17.6°C, respectively.

2.2 Monitoring and evaluation design

Three 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 to native species. These areas were matched as best as possible for ecological vegetation class and fire history (Appendix 1). There had been little fox control in the TMLs and NTMLs prior to 2005. 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 a random allocation of treatment and non-treatment sites was not feasible.

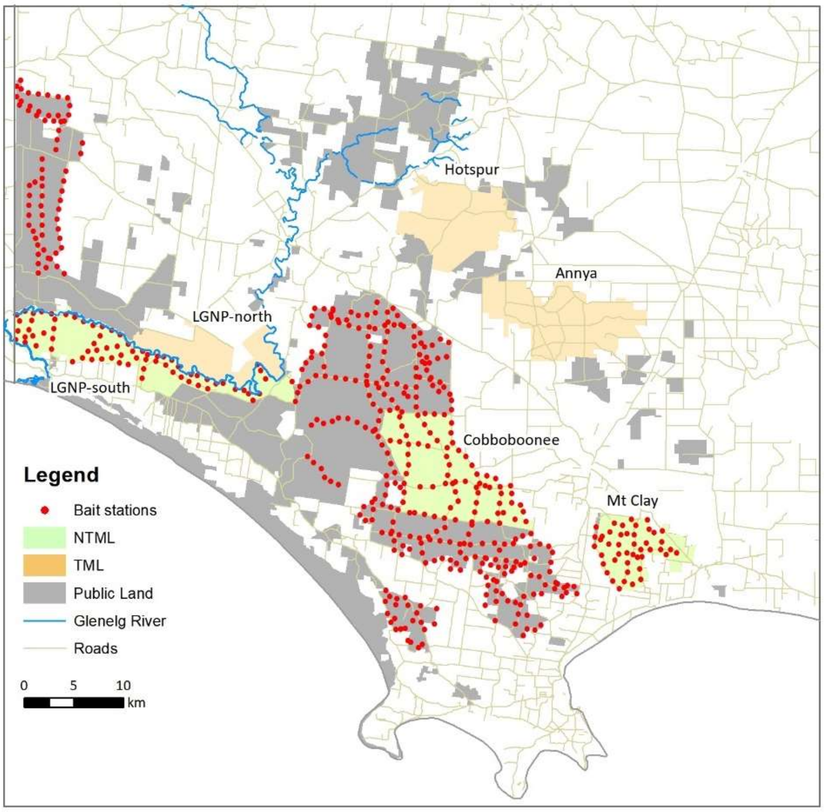

Figure 1. Glenelg Ark operations area. Red dots = poison bait stations. Tan areas = NTMLs. Pale green areas = TMLs. Fox baiting along the coast was discontinued in 2017. LGNP = Lower Glenelg National Park.

The six monitoring locations were:

1. Lower Glenelg National Park – south (LGNP-south; TML; 8954 ha)
2. Lower Glenelg National Park – north (LGNP-north; NTML; 4659 ha) (separated from LGNP-south by the Glenelg River)

3. Cobboboonee National Park (TML; 9750 ha)
4. Annya State Forest (NTML; 8520 ha)
5. Mt 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).

2.3 Measuring changes in fox and feral Cat activity

We examined the difference in fox and feral Cat activity between treatment and non-treatment locations from 2013 to 2018 using data generated from camera traps (see Section 2.5 for details of when and where camera traps were set). We used the number of independent images (separated by >3 hours) 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 implemented in BRMS package (Bürkner 2017) in R (R Development Core Team 2016) using RStudio (RStudio Team 2015, v 1.1.463), with treatment (NTML and TML) and location (each of the six individual locations) set as fixed effects and year set as a random effect; the presence of foxes was included in the feral Cat model as a fixed effect to test the influence foxes might have on feral Cat activity. The (log)number of cameras that operated on any given day was used as an offset in the models to allow for differing numbers of camera days per sampling period. Models were warmed up with 1000 interactions and sampled using 2000 interactions. Model evaluation was performed using approximate leave-one-out cross-validation based on the posterior likelihood in the 'loo' package (Vehtari et al. 2019).

2.4 Modelling change in fox density

Assessing changes in fox populations resulting from management interventions can be problematic, because foxes are shy and cryptic and occupy reasonably large areas. There are several methods commonly used to attempt to quantify changes in fox populations, such as activity indices (e.g. passive track counts, spotlight counts, the number of images per time step) or direct assessment of changes in abundance (e.g. mark–recapture using individuals identified using DNA extracted from in faecal samples). We used an index of activity calculated from images captured by cameras to assess differences in activity between TMLs and NTMLs as a measure of success. While this approach has been applied to assess pest animal control (Bengsen et al. 2014), the relationship between index values and actual abundance or density remains unknown. An alternative approach is to predict changes in fox density/abundance using computer modelling.

A spatially explicit, agent-based model (ABM; Hradsky et al. 2019) was built to predict fox populations and was run in the open-source software Netlogo (version 6.0.2; Wilensky 1999) and R (R Development Core Team 2016). The model can be used to explore the expected change in fox populations resulting from alternative management strategies (Robley et al. 2018; Hradsky et al. 2019), or to assess the effectiveness of current management actions, which is what was done in this study.

The model was run at 1-ha resolution over the 90,000-ha operations area of Glenelg Ark. This allowed for a buffer of ~30 km around the Glenelg Ark operations area and captured >95% of dispersing female and >90% of dispersing male foxes that might have reached this area of interest, assuming an average home range size of 4000-ha and dispersal distances that scale accordingly (Trewhella et al. 1988).

Fifteen interactions of the model were run for an initial period of 15 years to allow the fox population to stabilise. Baiting was commenced to coincide with post-dispersal and pre-breeding period in fox's life cycle, which corresponds with week 13 and was undertaken for 18 years to allow sufficient time for control and population responses to play out.

2.5 Measuring response in native mammal species

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 location of the monitoring sites was based on descriptions of the habitat preferred by the target native mammal species (Menkhorst 1995) and aligned with Ecological Vegetation Classes (EVCs); the number of sites was allocated according to the proportion of preferred habitat within each TML and NTML. The position of the monitoring sites within locations was randomly allocated but constrained to be within 50 m of tracks. A site was assumed to sample the area potentially occupied by the target species, with home ranges for Southern Brown Bandicoot and Long-nosed Potoroo ranging from 2 and 4 ha (Bennett 1987; Scott et al. 1999; Ricciardello 2006; MacGregor et al. 2013).

Monitoring prior to the commencement of poison baiting was conducted in winter 2005, then typically in spring (2005, 2008–2018). In 2006, sampling was undertaken in late winter, and the spring 2007 samplings at Mt Clay and Hotspur were delayed, with monitoring undertaken in the 2007–2008 summer.

From 2005 to 2012, at each monitoring site nine 'Handiglaze' hair-tubes (Murray 2005; Fig. 3) (baited with peanut butter, rolled oats and golden syrup) 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. Beginning in spring 2013, hair-tubing was discontinued, and a single digital camera (Reconyx RapidFire HC600, 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 attached to the nearest tree at 20–30 cm above the ground. A lure of truffle oil, peanut butter, rolled oats and golden syrup 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.

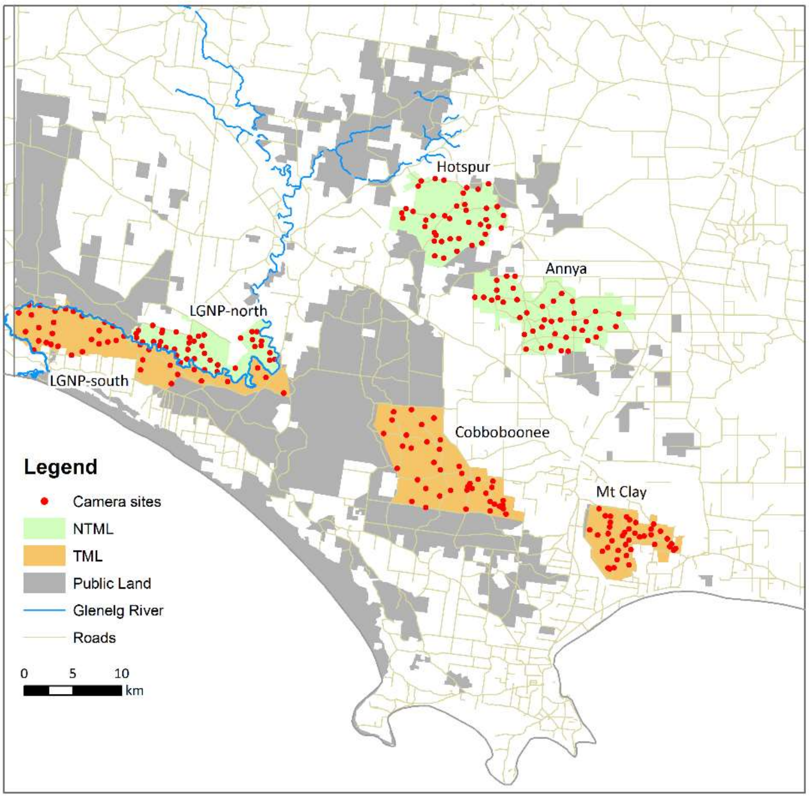

Figure 2. Monitoring sites in the TMLs (treatment monitoring locations; tan polygons) and NTMLs (non-treatment monitoring locations; green polygons) of Glenelg Ark are indicated by red dots. LGNP = Lower Glenelg National Park.

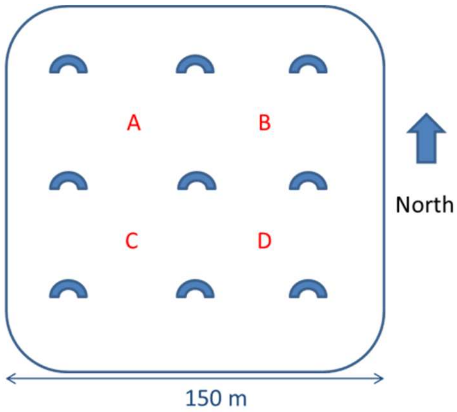

Figure 3. Layout of nine hair-tubes and possible location (A, B, C or D) of the single digital camera at a monitoring site

2.5.1 Data analysis

Long-term site occupancy changes in native mammals

To assess the long-term responses of the three selected mammals, we used multiseason occupancy models to estimate the occupancy (ψ), detection (p), local colonisation (γ) and local survivorship (ε) at monitoring locations between 2005 and 2018 (MacKenzie et al. 2003, 2006). The models were constructed in a Bayesian framework (Kéry 2010), using a space–state formulation (Royle and Kéry 2007). Models were implemented 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, based on all Gelman and Rubin convergence diagnostic potential scale reduction factors being less than 1.05 (Gelman et al. 2004). Initially, models were run with detection probability varying among locations through time, but these models did not converge, possibly due to the sparseness of the data, so a simpler model was run with detection being consistent among locations and years. This did not substantially change the results from the models run in previous years with the more complex set of predictors for detection. Flat naïve priors were used for the initial occupancy and detection rates (specifically a beta(1,1) distribution). However, flat priors caused boundary issues for localised colonisation and extinction rates; to avoid those issues, priors with less weight at the boundaries were used (specifically, a beta(2, 2) distribution).

A separate model was constructed for each of the native species of interest. The data for each species were 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). Hair-tube detection of Long-nosed Potoroo and Southern Brown Bandicoot differed depending on whether Common Brushtail Possums were detected at the site, and this was considered by the models. Hair analysis from the tubes indicated that the sticky-tapes were being swamped with Common Brushtail Possum hairs (B. Triggs, pers. comm.), and therefore Long-nosed Potoroo and Southern Brown Bandicoot could have been under-reported.

3 Results

3.1 Rainfall

Mean annual rainfall (recorded at the Portland Airport, ~20 km from the project area centre) varied substantially over the period 1983–2018 (BOM 2018) (Fig. 4). The years 1994–2001, prior to the commencement of this study, saw consistently below-average rainfall. For the period covering this study (2005–2018), 8 of the 13 years were below average; however, 2011, 2014 and 2017 were the three highest rainfall years over the 34-year period, with rainfall of 30.9%, 42.7% and 23.6%, respectively, above the 34-year average.

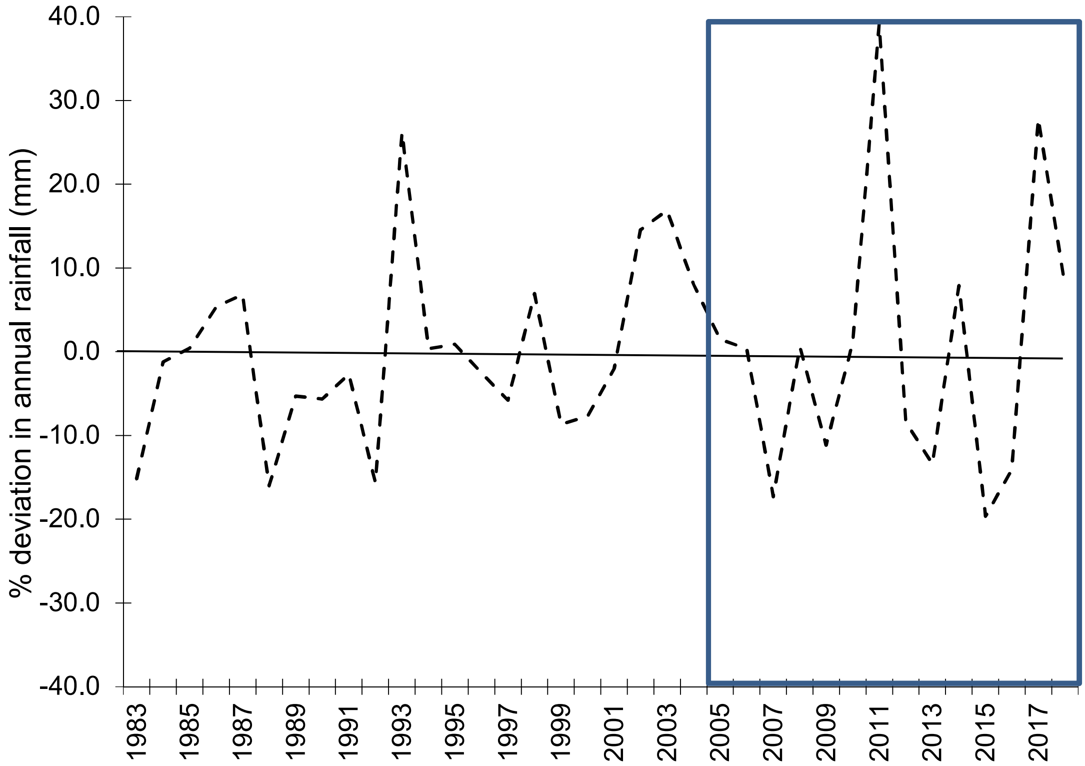

Figure 4. Deviation in mean annual spring rainfall from the 1983–2018 (35-year) average. The blue box highlights the period of monitoring at Glenelg Ark. Data is from the rainfall station at Portland Airport.

3.2 Fox and feral Cat activity

There was strong support for the regression model that foxes were more active on NTMLs than on TMLS. Fox activity between 2013 and 2018 was 88% higher across the NTMLs ($\bar{x}$ = 3.62, CI 3.22–4.08) than across the TMLs ($\bar{x}$ = 0.42, CI 0.36–0.49), based on the number of unique observations per 3 hours pooled across 24 hours on the camera traps.

Fox activity was significantly higher on all three individual NTMLs compared with the three TMLs (Table 1; Fig. 5), indicating that the impact of fox control was generalised across the TMLs. There was a slight decline in fox activity from 2013 to 2018 across all three NTMLs (Fig. 5), but this was not significant.

Table 1. Fox activity across all treatment monitoring locations. Estimate = mean number of images separated by >3 hours in a 24-hour period. SE = standard error, LCI and UCI = lower and upper confidence intervals, respectively**.**

Treatment	Location	Mean	SE	LCL	UCL
NTML	Annya	3.89	0.40	3.20	4.76
NTML	Hotspur	3.61	0.37	2.94	4.41
NTML	LGNP-N	3.32	0.34	2.70	4.05
TML	Cobboboonee	0.49	0.07	0.37	0.63
TML	LGNP-S	0.39	0.06	0.29	0.51
TML	Mt Clay	0.37	0.05	0.28	0.49

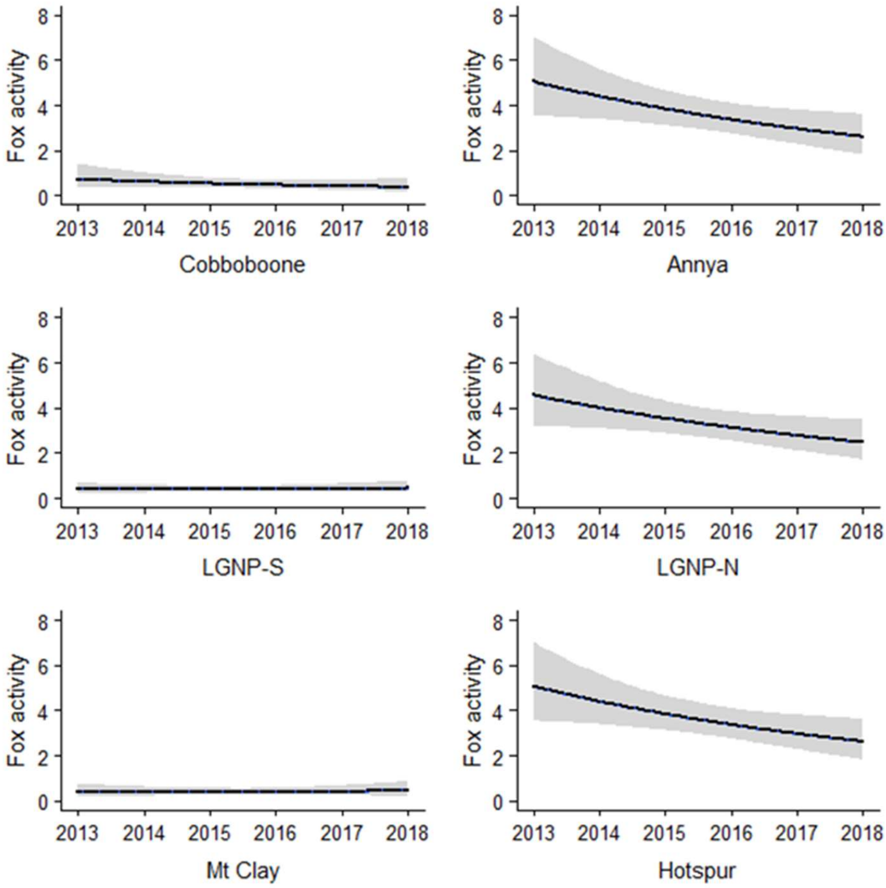

Figure 5. Fox activity (number of images separated by 3 hrs, per 24 hrs pooled across locations). TMLs = Cobboboonee, Lower Glenelg National Park-south (LGNP-S) and Mt Clay. NTMLs = Annya, Hotspur and Lower Glenelg National Park-north (LGNP-N). Grey shading indicates the 95% credible intervals.

There was no support in the regression models that feral Cat activity was higher at locations treated for fox control. Between 2013 and 2018, there was no difference in feral Cat activity between NTMLs ($\bar{x}$ = 0.25, SD 0.74) and TMLs ($\bar{x}$ = 0.25, SD 0.70).

There was considerable variation in feral Cat activity between locations: LGNP-north (TML) and LGNP-south (NTML) had the highest levels of feral Cat activity, and Annya (NTML) and Mt Clay (TML) had the lowest levels of feral Cat activity (Fig. 6). Feral Cats showed signs of decline from 2013 to 2018 at five of the six locations, with the strongest declines being observed at LGNP-south. Feral Cat activity trended upwards at Cobboboonee.

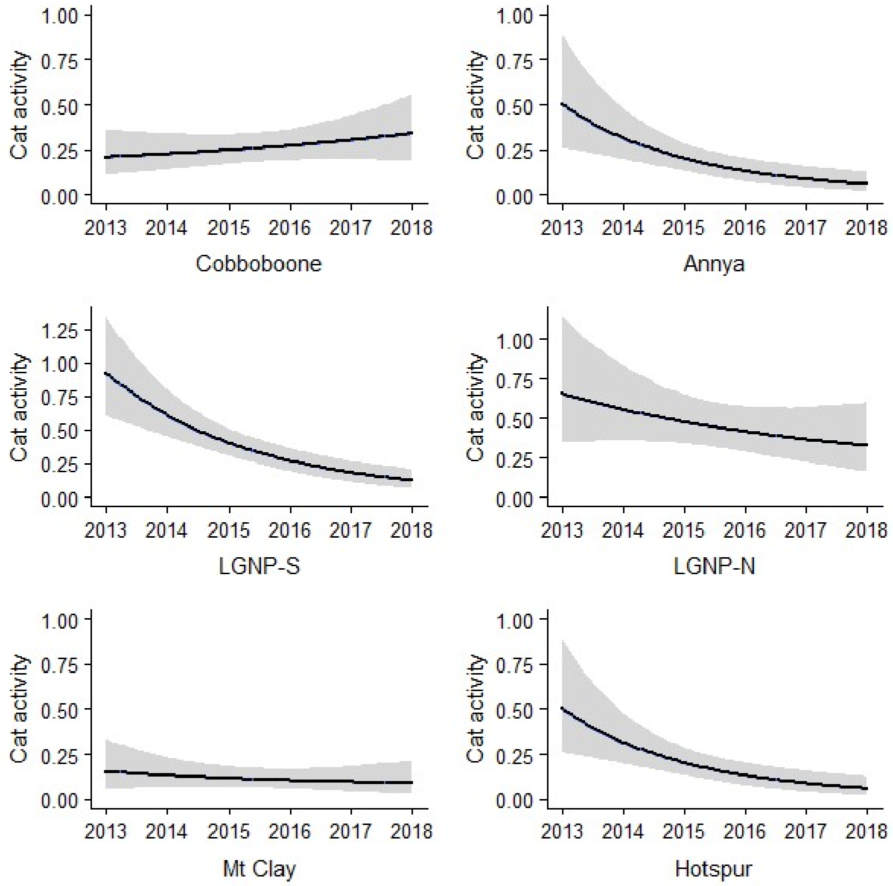

Figure 6. Feral Cat activity (number of images separated by 3 hrs, per 24 hrs pooled across locations). TMLs = Cobboboonee, Lower Glenelg National Park-south (LGNP-S) and Mt Clay. NTMLs = Annya, Hotspur and Lower Glenelg National Park-north (LGNP-N). Grey shading indicates the 95% credible intervals.

3.3 Fox density

Using the FoxNet model, fox density within the Glenelg Ark operations area was predicted to be 72.6% lower than pre-baiting densities after 18 years of baiting. The mean fox density just prior to the commencement of baiting was predicted to be 1.7 foxes/km^2 (minimum 1.1, maximum 2.5), whereas at 18 years post-baiting the mean density was predicted to be 0.45 foxes/km^2 (minimum 0.21, maximum 0.65). Figure 7 shows the density of foxes across the landscape in June, 18 years post-baiting. The month of June represents a time when dispersing animals have left the natal home range prior to the commencement of the annual breeding cycle. Fox density was predicted to have stabilised at the new lower density 3 years following the commencement of the baiting operation.

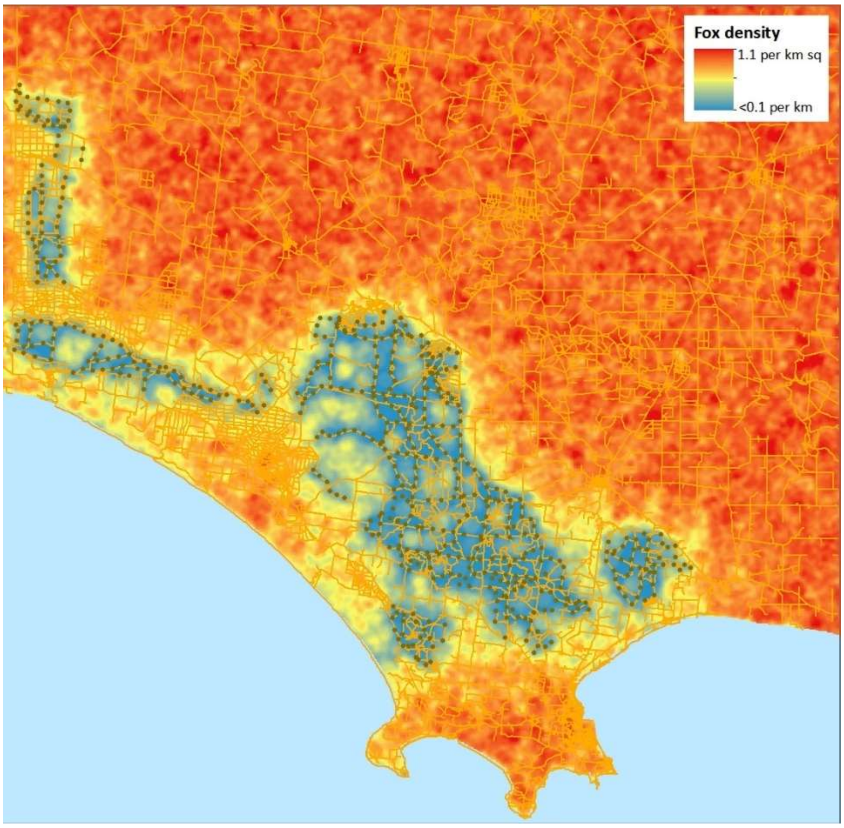

Figure 7. Predicted median fox densities 18 years after baiting commenced (determined using the agent-based model). Values represent the median fox density that year. Blue = <0.1 foxes/km^2, Red = 1.1 foxes/km^2, brown circles = bait station locations.

3.4 Response of selected native mammals 2005–2018

Overall, Common Brushtail Possums occupied the most sites, and their occurrence showed a clear difference between TMLs and NTMLs. Long-nosed Potoroo and Southern Brown Bandicoot occupied more sites across TMLs compared with NTMLs, but the difference was not significant. The differences in site occupancy were not uniform, there being considerable variation between locations.

3.4.1 Common Brushtail Possum

Since 2005, Common Brushtail Possum have on average occupied 63 (CI 62–64) or 52% of sites across the TMLs compared with 51 (CI 50–54) or 42.5% of sites across NTMLs. Since 2010, Common Brushtail Possums have occupied on average 16 (CI 14–18) more sites on TMLs compared with NTMLs (Fig. 8).

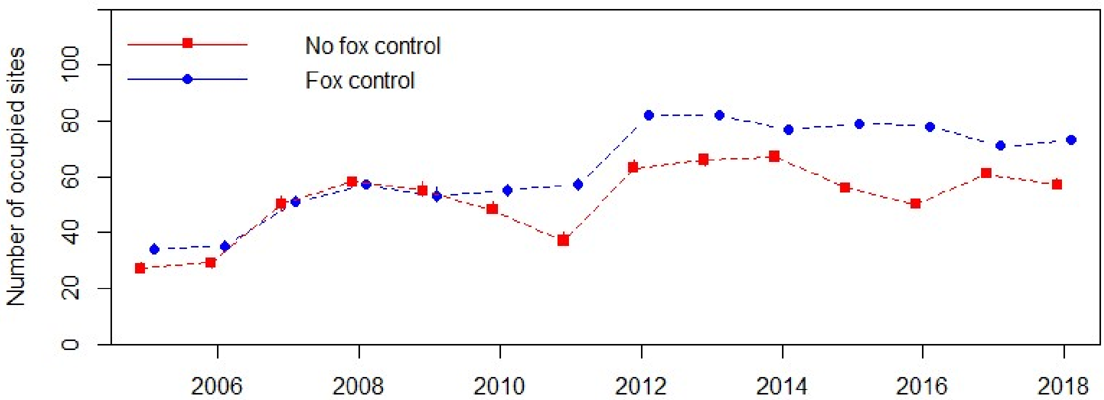

Figure 8. The overall occupancy of Common Brushtail Possums over time on TMLs (blue dotted line) and NTMLs (red dotted line). Vertical lines = 95% credible interval estimates.

At the individual location level, there was considerable variation in the number of sites occupied. Common Brushtail Possum were most common at LGNP-south (TML) [occupying almost nearly all the 40 monitored sites in all years since 2007 ($\bar{x}$ = 38)] and at LGNP-north (NTML) they occupied $\bar{x}$ = 34 sites in all years since 2013. Common Brushtail Possum were least common at Mt Clay (TML), occupying on average of 5.5 sites over the past 14 years. Occupancy at Hotspur (NTML) steadily increased from 2005 to 2009 and has fluctuated but remained within a similar range since that time. The number of occupied sites increased sharply at Cobboboonee (TML) in 2012 and, despite a slow decline in recent years, remains higher than in 2005; at Annya (NTML), however, occupancy peaked in 2007 and 2012, each peak being followed by periods of decline (Fig. 9).

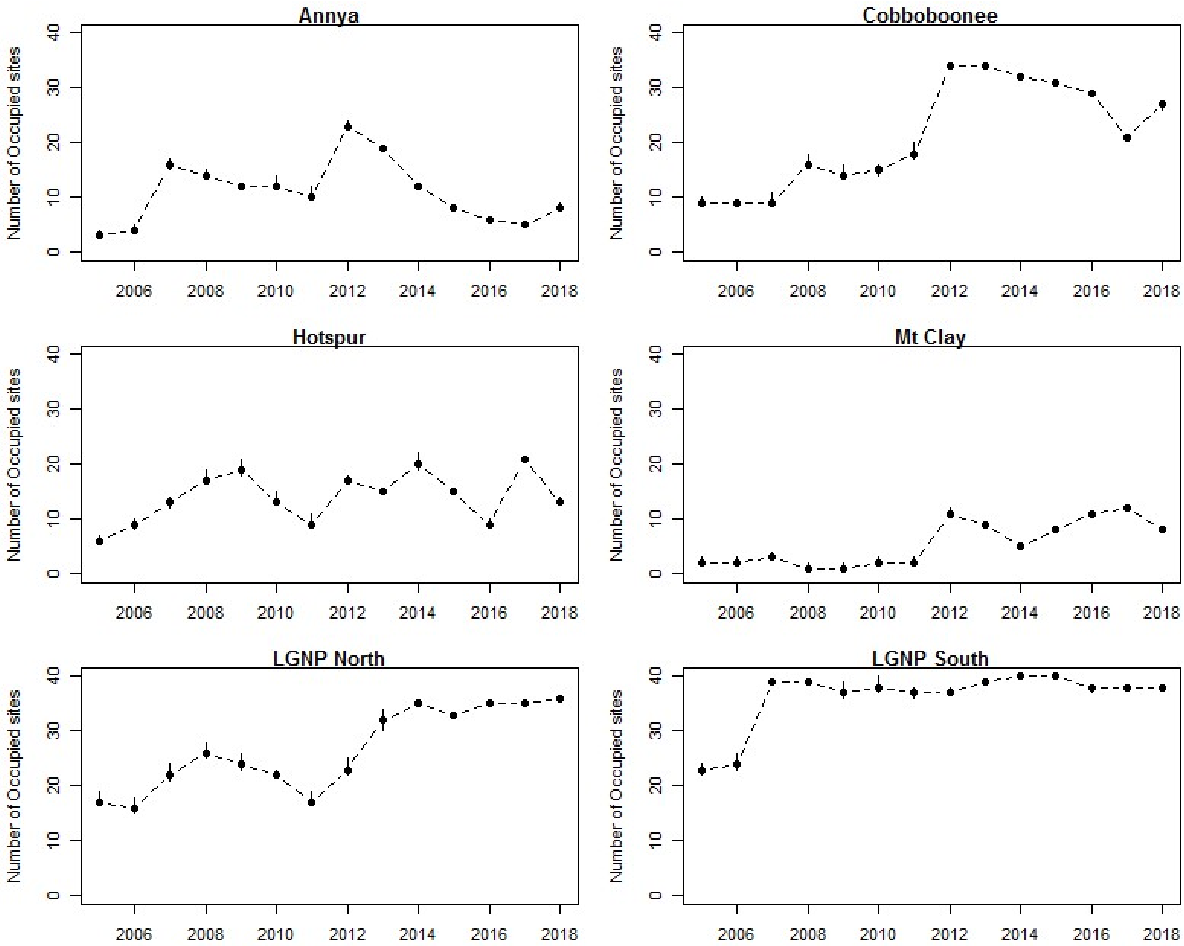

Figure 9. Estimated numbers of sites occupied by Common Brushtail Possums. NTMLs = Annya, Hotspur and LGNP-north; TMLs = Cobboboonee, Mt Clay and LGNP-south. Vertical lines = 95% credible interval estimates.

3.4.2 Long-nosed Potoroo

Long-nosed Potoroo occupied only a small number of the 120 surveyed sites across the three TMLs after 13 years of fox control. On average Long-Nosed Potoroos occupied 19 (CI 16–21) sites on TMLs compared to 12 (CI 10-15) on NTMLs. Since 2011 potoroos have declined on NTMLs from 13 sites (CI 10–17) in 2011 to 4 sites (CI 4–5). At the same time the number of sites occupied on TMLs has remained relatively constant ($\bar{x}$=17, CI 16–18) (Fig. 10).

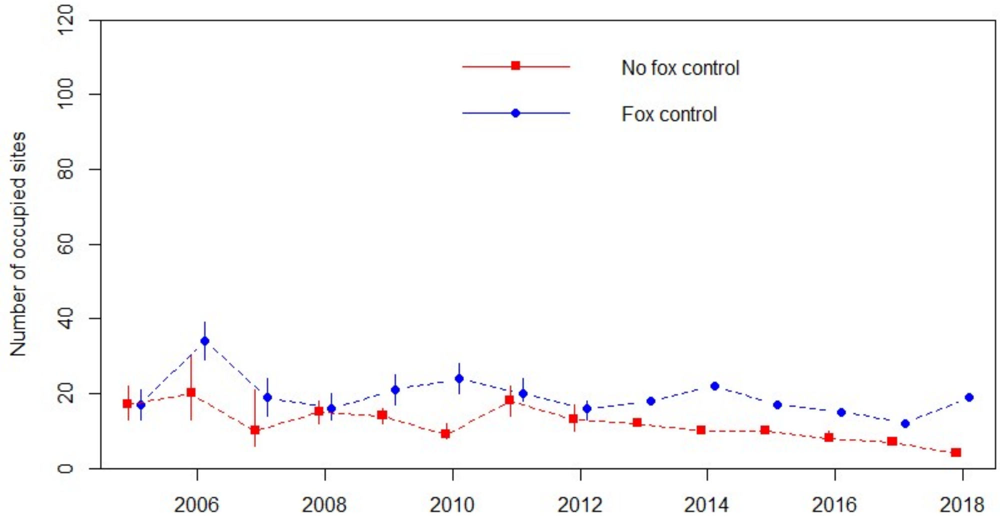

Figure 10. The estimated number of sites occupied by Long-nosed Potoroo over time on TMLs (blue dotted line) and NTMLs (red dotted line). Vertical lines = 95% credible interval estimates.

Model predictions are influenced by the data from previous years and indicate that, while there is some probability that Long-nosed Potoroo went undetected at some sites, the species is likely to be present in such low numbers that local extinction is probable at Annya (NTML) (detected at 1 site in 2018), Hotspur (NTML) (detected at 1 site in 2018) and LGNP-north (NTML) (detected at 2 sites in 2018).

While overall the number of occupied sites was higher on TMLs compared with NTMLs, the number of sites remains low after 13 years of fox control (2006-2018). There has been a slight increase in Long-nosed Potoroo since 2005 at Cobboboonee (TML), a slight overall decrease at LGNP-south (TML), and a decline in occupancy from 15 sites in 2010 to 3 sites in 2016 at Mt Clay (TML); at the latter location, they are showing signs of recovery, occupying 8 sites in 2018 (Fig. 11).

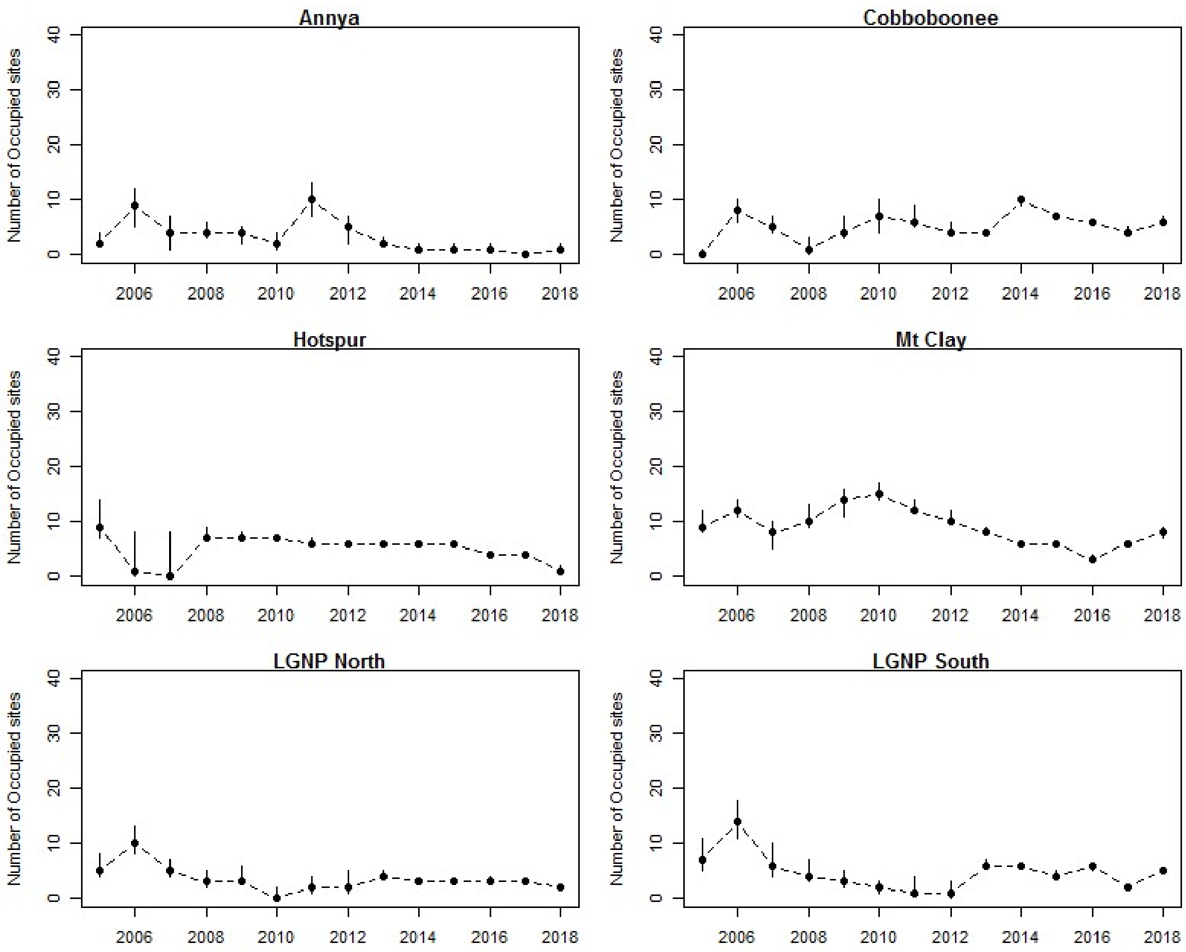

Figure 11. Estimated numbers of occupied sites for Long-nosed Potoroo in each location over time. NTMLs = Annya, Hotspur and LGNP-north; TMLs = Cobboboonee, Mt Clay and LGNP-south. The vertical lines represent the 95% credible interval estimates.

3.4.3 Southern Brown Bandicoot

Southern Brown Bandicoot occupy only a small number of the 120 possible sites across the three TMLs after 13 years (2005-2018) of fox control (Fig. 12). Since baiting began in 2005, Southern Brown Bandicoot have on average occupied more sites ($\bar{x}$ = 22, CI 18-27) on TMLs compared with NTMLs ($\bar{x}$ = 13, CI 9-16). Southern Brown Bandicoot responded positively to fox control in the immediate post-baiting period. Between 2006 and 2009, Southern Brown Bandicoot occupied on average 30, CI 22–38) compared to 13 sites (CI 8–18). However, from 2010 to 2014 bandicoots declined to the point where there was no difference between TMLs and NTMLs. Then between 2015 and 2018, Southern Brown Bandicoot occupied more sites on TMLS ($\bar{x}$ = 25, CI 23–28) than on NTMLS ($\bar{x}$ = 11, CI 10–13).

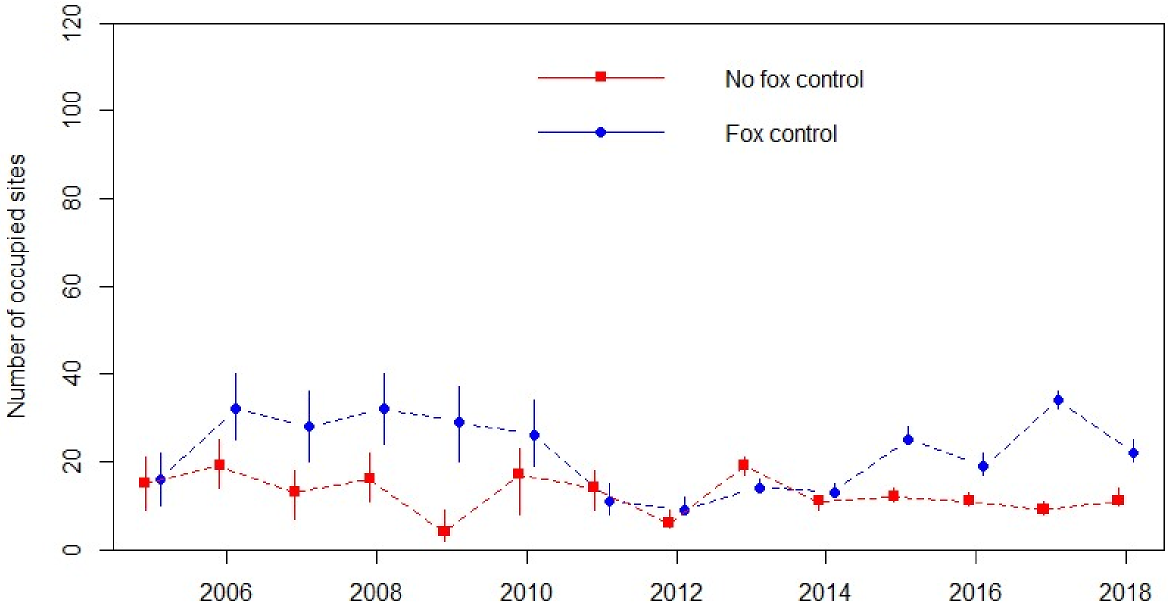

Figure 12. The overall occupancy of Southern Brown Bandicoot over time on TMLs (blue dotted line) and NTMLs (red dotted line). Vertical lines = 95% credible interval estimates.

Southern Brown Bandicoot are currently most common at Mt Clay (TML), having sharply increased in 2017 to occupy 25 of the 40 sites. Bandicoot occupancy peaked at Cobboboonee (TML) in 2009 and 2015, at $\bar{x}$ = 9 (CI 5–13) and $\bar{x}$ = 10 (CI 9–11), respectively (Fig. 13). At LGNP-south (TML), Southern Brown Bandicoot peaked in 2008, at $\bar{x}$ =14 (CI 9–20), but have subsequently declined steadily, now occupying $\bar{x}$ =3 (CI 3–4) sites.

Southern Brown Bandicoot are least common at LGNP-north (NTML), where they are likely to be in such low abundance that local extinction is probable (Fig. 13). The number of sites occupied by the Southern Brown Bandicoot at Annya (NTML) and Hotspur (NTML) have fluctuated over the years, but essentially have remained steady at $\bar{x}$ = 8 (CI 7–10) and $\bar{x}$ =10 (CI 9–12), respectively.

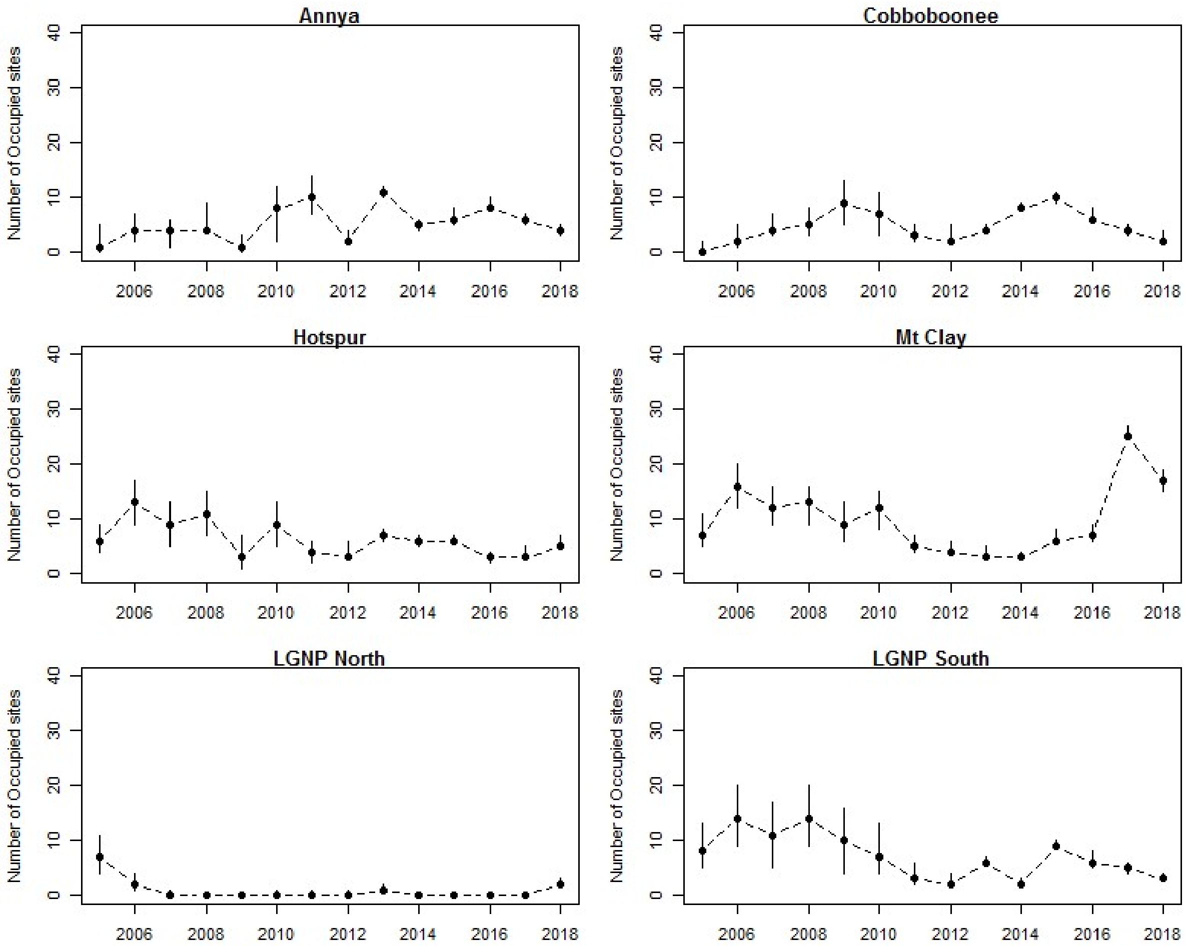

Figure 13. Estimated number of occupied sites for Southern Brown Bandicoot in each location over time. NTMLs = Annya, Hotspur and LGNP-north; TMLs = Cobboboonee, Mt Clay and LGNP-south. The vertical lines represent the 95% high-density intervals.

4 Discussion

Overall, site occupancy for Common Brushtail Possums, Long-nosed Potoroo and Southern Brown Bandicoot currently remains higher on TMLs than on NTMLs. Fox activity remains significantly lower on treatment sites compared with non-treatment sites, but feral Cat activity was not significantly associated with treatment effect. The modeled fox density across the Glenelg Ark operations area is significantly lower compared with pre-control density predictions and is consistent with the observed differences indicated by the camera trap monitoring.

After 13 years of fox control, which has clearly reduced the level of fox activity significantly, there is still only a limited response in the expansion of sites occupied by Long-nosed Potoroo and Southern Brown Bandicoot. The lack of a widespread response by these two species suggests they are limited to, or can only exist in, refugia (Kinnear et al. 2002; Long et al. 2005). This could potentially expose these populations to the effects of small population sizes (e.g. genetic bottlenecks, or Allee effects), compounding the risk of local extinction. Loss of genetic diversity is associated with decreased fitness (survival) and reduced evolutionary potential (less likelihood of adapting to future environmental changes) (Bouzat 2010). Allee effects can affect recovery of low-density populations: individuals in low population densities may have difficulty in locating a mate and may produce fewer offspring (Heiko et al. 2008). This limited response needs further investigation to define what actions, if any, can be taken to improve the distribution and long-term viability of these two species.

The high rates of occupancy of sites at LGNP-south and north by Common Brushtail Possums may be driven by other factors. Geary (2017) modelled the distribution of Common Brushtail Possums across the Glenelg Ark monitoring locations against a range of environmental variables and found that climate, proximity to farmland and topography were more influential on their distribution than predator control.

Attempting to determine the effect of fox control on species' responses by comparing the numbers of sites occupied by native species on TMLs with the numbers on NTMLs assumes that individual locations are ecologically similar. However, occupancy within individual location types (TML or NTML) varies, suggesting that conditions [e.g. resources (food and shelter), density of foxes and/or feral Cats, or levels of disturbance (habitat suitability)] are not uniform within a location type. What the underlying differences in conditions might be and just how these differences might act to affect native species abundance is not known and warrants further investigation.

Similar responses by native species considered to be at direct risk from fox predation have been reported elsewhere. When reviewing a 15-year fox control program at Booderee National Park, New South Wales, Lindenmayer et al. (2018) reported an increase in Common Brushtail Possums and macropods, and an initial increase in abundance of Long-nosed Bandicoot (*Perameles nasuta*) that was followed by a decline. Wayne et al. (2017) reported a decline in Woylie (*Bettongia penicillata*) at sites in south-west Western Australia after the implementation of intensive fox control. In that study, predation by feral Cats was implicated in the decline; however, Lindenmayer et al. (2018) reported a very low number of feral Cats at Booderee NP and were unable to explain the drivers of the observed declines.

There are five plausible explanations for the general lack of significant change in the occupancy of these species at the treated locations in the Glenelg Ark area.

1. The current sampling methodology is insensitive to the scale of change in occupancy, or occupancy is a poor metric for abundance; thus, the result is an underestimate of the scale of change that has occurred across the landscape.
2. Fox densities are still too high and limit the population growth of fragmented small populations of Southern Brown Bandicoot and Long-nosed Potoroo.
3. Feral Cats have replaced foxes as the main predator, with potentially similar dynamics to those of foxes.

4. Landscape disturbance, e.g. the long-term effects of frequent burning (both planned and natural), has resulted in a highly fragmented landscape. Populations are now restricted to isolated refugia, and species are unable to bridge the gaps between them.
5. Some combination of the above.

Several potential effects flow from these hypotheses.

Some redesigning of the sampling method is required to address the potential sampling issue. The combined cumulative probability of detection per site in 2018 for both Southern Brown Bandicoot and Long-nosed Potoroo was very low (P = 0.16 and 0.24, respectively), meaning that there was only a 16% chance of detecting a Southern Brown Bandicoot at a site in 2018, if in fact one was present. To increase the probability of detection there are two possible alternatives: (1) increase the number of sites surveyed, or (2) increase the duration of the surveys. MacKenzie and Royal (2005) provide an overview of designing occupancy studies and recommend that for rare species it is more efficient to sample more sites less intensively. This is because species need to be detected to calculate robust estimates of occupancy, and a species can only be detected at a site where it is present; hence, when occupancy is low, increasing the number of sites surveyed is likely to increase the number of sites where the species is detected.

Both the activity index and the spatial population model indicated that foxes are significantly lower in areas with fox control. The distribution of fox densities across the landscape in Figure 7 shows the likely extent of the area of influence the control operation has on foxes, and clearly demonstrates the need to maintain ongoing broadscale fox control. The mean density of foxes predicted by the model ranged between 0.2 and 0.7 foxes/km^2 after 9 years of baiting. This would roughly equate to 180–450 foxes remaining resident within the operations area of Glenelg Ark.

If foxes have not been reduced below the threshold at which Southern Brown Bandicoot and Long-nosed Potoroo can escape limitation, then further reducing the remaining fox population should result in increased survival in these populations, and thus an increased number of sites occupied by them. Model predictions (Robley et al. 2017) suggest that a decrease in bait spacing to 500 m, with continued replacement at fortnightly intervals, could further reduce fox density. These models could be extended to include fox control on private land, under various plausible scenarios, e.g., to create buffers around public land blocks to examine the potential efficacy of this strategy in further reducing fox densities.

Regardless of the strategy implemented, measuring the outcome (both in terms of any further reduction in foxes and response in native species) will require additional effort.

Within the Glenelg Ark operations area, the sustained reductions in fox populations may have resulted in increased activity (and possibly abundance) of feral Cats. Although the activity index was not significantly different between TMLs and NTMLs, the point estimates suggest a higher level of activity on TMLs. Robley et al. (2010) showed that the number of sites occupied by feral Cats was higher at LGNP-south (ψ = 0.69 ± 0.10 SE) compared with the non-treatment location of LGNP-north (ψ = 0.050 ± 0.13 SE), and several studies elsewhere have described increases in feral Cat abundance following reductions in fox numbers (Algar and Smith 1998; Catling and Reid 2003). This effect has also been described following local declines in Dingo (*Canis lupus dingo*) abundance in Queensland (Pettigrew 1993). Catling and Burt (1995) also reported that the abundance of feral Cats was negatively correlated with both foxes and Dingoes at a site in New South Wales. Read and Bowen (2001) did not manipulate predator populations 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. Based on the Glenelg Ark ongoing management initiative, a PhD candidate of the University of Melbourne is investigating the effects of fox control on feral Cat density and on the native mammal community in Glenelg Ark and the Otway Ranges.

Feral Cats have recently been declared a pest species in Victoria under the *Catchment and Land Protection Act 1994*, obligating public land managers to control this pest. Under the new arrangements, management strategies in Victoria are limited to cage-trapping and shooting. The deployment of toxic baits from the air or on the ground, and the capture and destruction of feral Cats in leg-hold traps are not currently permitted. These restrictions limit the capacity of public land managers to develop and implement effective management practices in Victoria.

However, the recent declaration of feral Cats as a pest species and the near-future availability of a feral Cat bait provides the opportunity to explore integration of fox and feral Cat control at some locations. Table 2 summarises one possible study design that could explore the effect of landscape-scale feral Cat control at Glenelg Ark. The design has fox and feral Cat control at two existing treatment locations, and only fox control at one, with no changes to the non-treatment monitoring locations.

Table 2. Possible study design to investigate the effect of landscape-scale feral Cat control on native mammals at Glenelg Ark

Location	2018/2019	2019/2020	2020/2021	2021/2022	2022/2023
Cobboboonee	Fox	Fox/Cat (B)	Fox/Cat (B)	Fox/Cat (B)	Fox/Cat (B)
Mt Clay	Fox	Fox/Cat (B)	Fox/Cat (B)	Fox/Cat (B)	Fox/Cat (B)
LGNP-south	Fox	Fox	Fox	Fox	Fox
Annya	No fox or cat	No fox or cat	No fox or cat	No fox or cat	No fox or cat
LGNP-north	No fox or cat	No fox or cat	No fox or cat	No fox or cat	No fox or cat
Hotspur	No fox or cat	No fox or cat	No fox or cat	No fox or cat	No fox or cat

(B) = poison baiting.

Fire is a complex phenomenon that can have positive and negative effects on the functioning of ecosystems around the world (Bond and Keeley 2005), but it can also present a serious threat to human life and property when uncontrolled. Managing the ambiguity of fire has been a long-term and significant challenge for land managers (Bond and Keeley. 2005). Increasingly, land managers are recognising that fire can be used to manage biodiversity assets and promote ecological processes (Bradstock et al. 2012), and that interactions between the disturbance created by fire and other processes (e.g. predation, variations in climate, and habitat fragmentation) can act in concert to shape ecosystem responses (Dale et al. 2001; Beschta and Ripple 2009).

The deliberate and repeated application of fire to the landscape (i.e. planned burning) can have significant ecological impacts through altering the composition and structure of the vegetation (Burgess et al. 2015). Planned burning can reduce vegetation cover, nesting sites and the protection of individuals from predation (Radford 2012; Lentic et al. 2013). This is because the presence of physically complex habitat (e.g. dense vegetation or woody debris) can impede the ability of predators to detect and chase prey, thereby increasing the probability that the prey will evade the predator (e.g. Mandelik et al. 2003).

In addition, planned burning can have either positive or negative impacts on food availability. The simplified habitat structures that can be created by planned burning can be important for foraging, e.g. by Long-nosed Potoroo exploiting hypogeal fungi following fire (Bennett 1993; Vernes et al. 2004; Norton et al. 2015), in some cases, despite the increased risk of fox predation (Norton et al. 2015). Planned burning can also decrease the availability of food resources for some species, through reducing the composition and structure of the vegetation (Woinarski et al. 2004). When foraging for food, small- to medium-sized ground-dwelling mammals, such as antechinus, potoroos and bandicoots, require a complex understorey with a diverse shrub layer and a ground cover of logs, leaf litter and woody debris (Paull and Date 1999).

Two projects are commencing in 2019 that will address some of the knowledge gaps concerning the interaction between fire, predation and native species. One will investigate whether fox control affects native mammal resilience to planned burning events. This project will compare recovery rates of native mammals following planned burning operations in locations with and without fox control. This project builds on previous investigations at Glenelg Ark, which found significant declines in floristic composition and diversity, habitat structure, and mammal occurrence, and shifts in the diet of foxes following planned burn operations (Robley et al. 2016). The second project will look at developing a species distribution model for Southern Brown Bandicoot in the Glenelg region, and use spatially explicit metapopulation modelling to explore the potential effects of fire and predator management on this species.

An outstanding issue in assessing the effectiveness of the fox control is the relationship between relatively cost-effective and simple measures, such as activity (the number of camera images/day), and more expensive and difficult measures of abundance, e.g. mark–recapture estimates of abundance from individual identifications using DNA sampling from scats. The relationship between activity and abundance has been assumed to be linear (i.e. it has been assumed that a unit decline in 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 relationship this has to fox abundance. This lack of understanding clouds the interpretation of the native species monitoring results, and of the effectiveness of the fox control strategies.

The Glenelg Ark monitoring program has focused on changes in three medium-sized mammal species in response to a reduction in fox abundance across the landscape. Other species that are present in the Glenelg Ark area, 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. The population responses of the Heath Mouse are currently not being monitored within Glenelg Ark, in part because there has been no standard survey protocol. A protocol is now available (R. Hill, DELWP, pers. comm.) and should be implemented to assess the relative status of this species across TMLs and NTMLs. Other native species that in theory should respond positively to a reduction in foxes are Common Ringtail Possums (*Pseudocheirus peregrinus*), and owls such as Australian Masked Owl (*Tyto novaehollandiae*) and Powerful Owl (*Ninox strenua*) due to increases in native prey such as possums, and which have all been reported in the Glenelg region. Contrary to expectations, however, Lindenmayer et al. (2018) reported a decline in large forest owls at Booderee National Park after fox baiting.

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 long-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. In addition, the project provides a framework and infrastructure through which other management-focused research questions can be addressed (e.g. the response of other small mammals, and the impact of possible unintended consequences, such as over-browsing and changes in the feral Cat population).

4.1 Recommendations

To address the above issues and improve management outcomes, we suggest the following actions. These actions can be undertaken, as stand-alone activities or in various combinations, to more fully explore the issues and to fill knowledge gaps, thus enabling improved management.

Item	Recommendation	Detail
Native species response	Increase the number of camera sites to improve detection rates for Southern Brown Bandicoot and Long-nosed Potoroo. Timing: Before 20/21 sampling year. Responsibility: Project Officer and ARI.	Low detection rates of both species may reflect an actual low abundance of these species, or the low occupancy rates may be an artefact of the sampling effort. Deploying more cameras within a location may resolve this dichotomy, by either increasing detection rates and increasing accuracy of occupancy estimates or decreasing the level of uncertainty by determining that occupancy is indeed very low and other management actions need to be considered.
	Model the patterns in the changes in occupancy from 2005 to 2018 to investigate whether species are dispersing into new areas or whether they are limited to certain habitats; investigate factors that may influence the spread of recovery. Timing: Before 20/21 sampling year. Responsibility: ARI, UoM and Project Officer	Quantifying and understanding the factors that influence the rate of recovery and spatial spread of threatened species in relation to management intervention is a key issue in conservation biology. Recovery at a landscape scale may depend on characteristics such as the preferred direction of spread and the distance between 'suitable' locations. Studying these characteristics is essential for making appropriate management decisions. We propose using a hierarchical model that takes spatial structure, distance between sites, and the possibility of directional spread into account. This information will improve our understanding of the drivers and the limitations of species recovery following fox control.
	Develop Southern Brown Bandicoot and Long-nosed Potoroo habitat suitability surfaces for the Glenelg Ark project area (using presence/absence data) to aid in setting species response targets and to identify potential new control and/or monitoring sites. Timing: Before 20/21 sampling year. Responsibility: ARI, UoM and Project Officer.	Use in conjunction with the first recommendation. The limited responses of the Southern Brown Bandicoot and Long-nosed Potoroo may be due to a lack of suitable habitat for these species. We propose that the site occupancy information be used to explore this possibility. Freely available remotely sensed habitat data (e.g. vegetation type, topography, fire history, distance to drainage lines, distance to forest edge, and landscape productivity data) can be combined with the 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 will also identify potential new locations for monitoring and/or fox control actions.
	Using expert elicitation, describe the benefits of fox control for the Heath Mouse (*Pseudomys shortridgei*); select sites for targeted monitoring of TMLs and NTMLs. Timing: Before 20/21 sampling year. Responsibility: ARI, and Project Officer.	Current monitoring sites were placed in locations based on best understanding of 'suitable' habitat for the three main target species at the time. Heath Mouse species distribution models have been developed and 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 in abundance between TMLs and NTMLs.
Improved fox control	Increase the density of baits on TMLs, and robustly assess the outcome in terms of fox, feral Cat and native species response. Timing: 2019/20. Responsibility: Project Officer.	Modelled predictions of changes in fox density in response to an increase in bait density undertaken in 2017/2018 show a further decline in foxes is likely. To make informed management and investment decisions, any outcomes from changes to management need to be robustly assessed.

Integrated predator control	Implement targeted feral Cat control at selected locations with known populations of Southern Brown Bandicoot and Long-nosed Potoroo. Timing: Before 20/21 sampling year. Responsibility: Project Officer/ARI.	Feral Cats may limit the response of Southern Brown Bandicoot and Long-nosed Potoroo to fox control. With the recent declaration of feral Cats as a pest species, it is now possible to implement targeted control at specific locations with the currently available tools (cage trapping, shooting). Based on the results of the above recommended actions to model species habitat and distribution, select areas for targeted control action.
	Undertake feasibility analysis, including cost for broadscale baiting of feral Cats within the Glenelg Ark operational area. Timing: Before 20/21 sampling year. Responsibility: Project Officer/ARI.	Current control tools are limited to cage trapping and shooting. While these tools can play a role in feral Cat control, broadscale control will only be achieved using baiting. Curiosity bait is being registered by the Federal Government and is likely to be operationally available in 12-18 months. Planning when, where and how to use this tool should commence soon to ensure timely implementation can occur once regulatory and policy approvals are in place.
Alternative survey methods for foxes and feral cats	Assess the feasibility and cost of genotyping DNA from fox scats collected using scat-detector dogs. Timing: Before 20/21 sampling year. Responsibility: Project Officer.	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.
	Assess the feasibility and cost of genotyping DNA from hair samples collected using hair snare traps for feral Cats. Timing: Before 20/21 sampling year. Responsibility: Project Officer.	Genotyping DNA from hair samples have been used successfully to enumerate feral cat populations. A similar approach could be used in Glenelg Ark to assess differences between baited and comparable unbaited areas. However, attracting cats to hair snare traps requires an effective lure. Undertake trials to assess lure types and their relative effectiveness at different times of the year.
	Undertake trials to assess lure types and their relative effectiveness at different times of the year. Timing: Before 20/21 sampling year. Responsibility: Project Officer/ARI.	If cost and feasibility analysis indicate that implementing hair snare monitoring is possible a set of effective lures will be required to reliably attract feral Cats to hair snares.
Scientific support	Develop service agreement for the continued scientific support and advice concerning the ongoing implementation and development of Glenelg Ark. Timing: Before December 2019. Responsibility: Project Officer.	Evaluation and interpretation of the monitoring data, development of new projects addressing emerging issues, and general guidance to the project from the scientific community has been instrumental in its success.
Monitoring and reporting	Continue annual monitoring, evaluation and reporting. Timing: Before June 2020. Responsibility: Service provider/Project Officer.	Continue annual monitoring, evaluation and reporting to closely track changes in predators and prey, thus allowing more responsive management of emerging issues, e.g. a decline in Southern Brown Bandicoot, or a change in feral Cat abundance.

Filling specific knowledge gaps	Develop and support a set of potential student projects to fill identified knowledge gaps. Timing: Before December 2019. Responsibility: Project Officer/ARI.	The current monitoring program does not assess changes in small native mammals [e.g. Heath Mouse and White-footed Dunnarts (*Sminthopsis leucopus*)], or unintended consequences (e.g. the interactions between small native mammals with fox control, feral Cat control and fire). A series of student projects could fill these knowledge gaps, taking advantage of the infrastructure that Glenelg Ark provides. Where these might already be occurring, continue to provide logistical and in-kind support. Look for opportunities to provide financial support.

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Appendix

Appendix 1. Ecological vegetation classes within each treatment and non-treatment location, and the number of cameras allocated to each location

Monitoring area	Ecological Vegetation Class	Area (ha)	%	No. of cameras
Mt Clay State Forest (treatment)	Lowland Forest	1950	44	18
	Heathy Woodland/Damp Heathy Woodland/Damp Heathland Mosaic	1597	35	14
	Herb-rich Foothill Forest	847	20	8
Hotspur State Forest (non-treatment)	Lowland Forest	3097	51	20
	Heathy Woodland	2235	37	15
	Wet Heathland	493	11	4
Cobboboonee National Park (treatment)	Lowland Forest	7557	84	34
	Wet Heathland/Heathy Woodland Mosaic	1035	15	6
Annya State Forest (non-treatment)	Lowland Forest	5704	70	32
	Damp Sands Herb-rich Woodland	1106	18	7
LGNP-south (treatment)	Damp Sands Herb-rich Woodland/Heathy Woodland Mosaic	2855	34	14
	Heathy Woodland/Limestone Woodland Mosaic	2855	34	14
	Damp Sands Herb-rich Woodland	1319	17	7
	Damp Sands Herb-rich Woodland/Heathy Woodland/Sand Heathland Mosaic	972	14	6
LGNP-north (non-treatment)	Wet Heathland/Heathy Woodland Mosaic	2041	45	18
	Damp Sands Herb-rich Woodland	2021	44	18
	Damp Sands Herb-rich Woodland/Heathy Woodland Mosaic	417	10	4

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