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Glenelg Ark 2005–2011: Evidence of Sustained Control of Foxes and Benefits for Native Mammals

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Summary

The Glenelg Ark project was established in 2005 under the Department of Sustainability and Environments Weeds and Pests on Public Land Initiative. The project aimed to facilitate the recovery of native mammal populations in far south-west Victoria by undertaking broadscale, continuous Red Fox (*Vulpes vulpes*) baiting using buried Foxoff® baits across 100 000 ha of state forest and national park.

The monitoring and evaluation program measured the effectiveness of the fox control operation as well as the response of a range of native species at risk from Red Fox predation. The monitoring program used a conditional case-control study design to identify whether changes in site occupancy by native species (the ‘case’) are conditional on the presence of foxes, by comparing sites that have foxes present but are otherwise similar (the controls) with those that have a substantially reduced index of fox abundance.

Three Treatment Monitoring Areas (i.e. areas that received fox control) and three Non-Treatment Monitoring Areas (i.e. areas that did not receive fox control) were paired in the landscape and replicated three times. We used differences in the proportion of baits taken and activity measured on sand plots as indices of fox abundance, and the number of sites occupied by three target native mammal species (including persistence and colonisation rates) as indices of abundance. To our knowledge this is the first time that site occupancy modelling has been applied to assessing the response of native species to a reduction in foxes.

The Glenelg Ark monitoring program has achieved its aims of quantifying the response of foxes and targeted native species to broadscale continuous fox control. As more data is collected, the uncertainty around the nature of these responses is expected to decline.

There was a significant and sustained decline in the index of fox abundance derived from bait take on the three treatment monitoring areas compared to the three non-treatment monitoring areas (76%–81% initial reduction). Assessment of comparative occupancy rates between a pair of treatment and non-treatment sites based on data from camera traps indicated a significantly lower level on baited sites. However, sand plot activity did not conclusively indicate differences between treatment and non-treatment sites.

Occupancy modelling showed that there was a significant positive effect of treatment on persistence and colonisation for Southern Brown Bandicoots (*Isoodon obesulus*) and Long-nosed Potoroos (*Potorous tridactylus*) each at a single but different site. There was a positive effect of treatment on persistence for Southern Brown Bandicoots at a second site and in colonisation at a third. There was a trend in a positive effect for Long-nosed Potoroos at a second site and no indication of a treatment effect on colonisation or persistence at the third site. Logistic modelling on the Southern Brown Bandicoot data for 2010 showed that while ecological vegetation class (EVC) and dominant tree species were important predictor variables, sites with fox control were five times more likely to have bandicoots present.

The results of the monitoring and evaluation program clearly demonstrate that foxes have been reduced and remain at relatively lower levels, and that there have been increases in the two native mammal species.

Results indicate a variable response by Southern Brown Bandicoots and Long-nosed Potoroos to fox control, however; it is critical that the fox control program continue and monitoring data be collected to reduce the uncertainty around the nature of this response. Determining the relative effectiveness of alternative methods for assessing differences in fox abundance, and examining the influence of rainfall, fire history and vegetation community on rates of persistence and colonisation will assist in clarifying the nature of the mammal fauna response.

1 Introduction

1.1 Background

Predation by the Red Fox (*Vulpes vulpes*) is a key threatening process for smaller native mammals and ground-nesting birds, and may be partly responsible for several extinctions (e.g. Lunney 2001). Predation by foxes is listed as a key threatening process under the *Victorian Flora and Fauna Guarantee Act 1988* and the Federal *Environmental Protection and Biodiversity Conservation Act 1999*. Fox control operations are conducted for threatened species and livestock protection on an estimated 10.5 million ha of land in Australia per year (Reddiex et al. 2004).

Poisoned baits are widely used to reduce populations of foxes throughout Australia. It is the main tool used in projects aimed at protecting or enhancing populations of native wildlife (Saunders and McLeod 2007).

Measuring the response of native wildlife to fox control is problematic. McLeod et al. (2008) review 14 studies that measured the response of native species' population size following fox control and found the results were inconsistent and sometimes conflicting. Possible explanations include: different species being effected in different ways; complex interactions between factors such as habitat and other predators; the baiting strategy being ineffective; the time frame of the control not being sufficient; or fox predation not having a measurable impact. Most studies only monitored the prey species of interest; however, Risbey et al. (2000) highlighted the complexity of ecosystems with a positive response to fox control in populations of feral cats, which then impacted on the preferred prey species.

An alternative to measuring changes in population size is to measure changes in the proportion of sites in an area that are occupied by the species in question (termed 'occupancy'). The simplest approach to estimating occupancy is to derive a naive estimate, i.e. the number of sites with at least one detection divided by the total number of sites in the study area. This assumes that if a species is present on a site then it will be detected, i.e. the probability of detection is 1. This method will likely result in a negatively biased estimate of site occupancy, as it is possible that sites that are considered unoccupied could in fact have species present that were undetected. MacKenzie and Kendall (2002) proposed that an unbiased estimation of occupancy can be achieved by repeated surveying of sites.

1.2 Glenelg Ark

The Glenelg Ark project was established in July 2005 to facilitate the recovery of native animal populations considered at risk from fox predation by undertaking large-scale, continuous fox baiting across 100 000 ha of state forest and national park in south-western Victoria (Figure 1).

To ensure 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 is measuring: a) the response of foxes to control activities, and b) the response of a range of native species that are currently at risk from fox predation. Without such a program, management will have no capacity to: (a) justify reinvestment of scarce public conservation funds, (b) improve management actions based on scientific information about the effectiveness of previous management actions, and (c) maintain community support. Thus, the monitoring and evaluation form part of management and are not an imposition or adjunct to it.

It is acknowledged that a coherent and robust response in native mammals may not be measurable for several years. The outcomes of the monitoring and evaluation program will enable policy makers, and the community, to assess the success and future direction of Glenelg Ark.

2 Methods

2.1 Monitoring and evaluation design

The aim of the Glenelg Ark monitoring and evaluation program is to provide quantitative evidence of the effect of the fox baiting program and the associated changes in native mammal species. The *a priori* assumption is that fox predation is the key threatening process and that species at risk from fox predation would escape population regulation and increase in abundance once foxes had been reduced. Implicit in this assumption is the notion that although there are many variables which impact on a population's growth (e.g. food, predation, disease, climate, social and genetic), the major controlling variable (in this case fox predation) will show strong signals through the noise of contingent events.

We use a conditional case-control study design to identify whether changes in site occupancy by native species (the 'case') is conditional on the presence of foxes, by comparing sites that have foxes present but are otherwise similar (the controls) to those that have a substantially reduced number of foxes.

Three paired monitoring areas known as Treatment Monitoring Areas (TMAs, i.e. areas that received fox control) and Non-Treatment Monitoring Areas (NTMA, i.e. areas that did not receive fox control) (Figure 1) are used to assess the effectiveness of the program. In order to achieve a broadscale reduction in foxes across the public land estate areas, treatment for fox control was consolidated in the southern half of the overall project area (Figure 1). This meant that random allocation of treatment and non-treatment sites was not feasible. The three paired areas are:

1. Lower Glenelg National Park south (TMA; 8954 ha) and Lower Glenelg National Park north (NTMA; 4659 ha); separated by the Glenelg River
2. Cobobboonee National Park (TMA; 9750 ha) and Annya State Forest (NTMA; 8520 ha)
3. Mount Clay State Forest (TMA; 4703 ha) and Hotspur State Forest (NTMA; 6940 ha).

This design will identify any patterns of association between a reduction in foxes and an increase in site occupancy of targeted native species but does not allow any statistical interpretation of causality or effect (Lande et al. 1994).

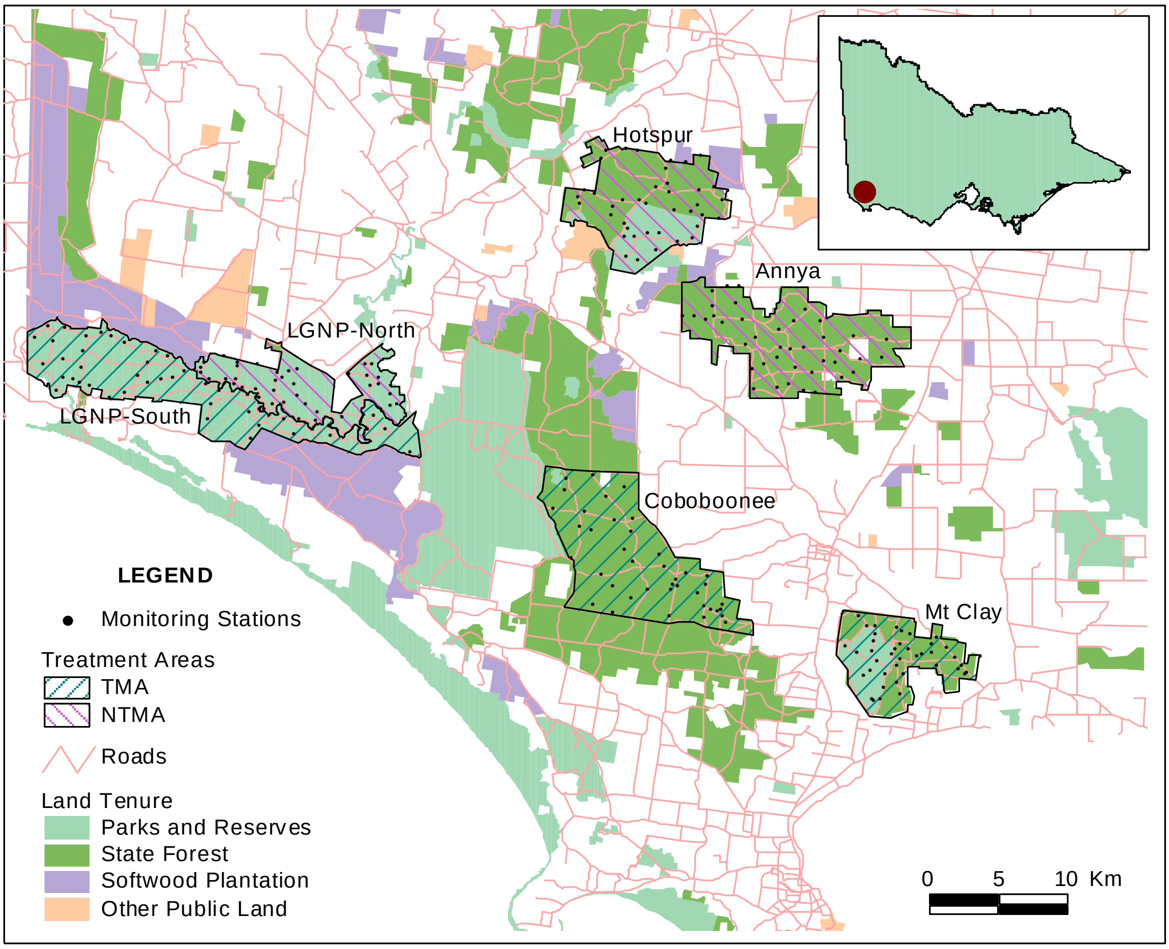


Figure 1. Glenelg Ark monitoring and evaluation areas.

2.2 Measuring changes in foxes

In the TMAs and NTMAs there had been little previous fox control prior to 2005. Each TMA bait station was constructed at a 1 km interval, and a single 1080 Foxoff bait was buried to a depth of 10 cm. Bait stations were checked and all baits replaced on a fortnightly basis throughout the year. An assessment of which species was likely to have taken bait was recorded at the time of bait checking. NTMA bait stations were constructed in the same manner and non-toxic Foxoff baits used. The number of bait stations ranged from 46 to 78 per area. Monitoring the results of the baiting program has three components:

1. demonstrating the impact of the initial knockdown of the fox population
2. monitoring the outcome of the sustained fox control measures
3. measuring changes in fox activity.

2.2.1 Demonstrating the impact of the initial knockdown of the fox population

To establish an index of pre-baiting fox activity, free-feed (i.e. non-toxic) baits were used for several weeks prior to the commencement of poisoning. Progressively higher frequencies of bait take occur as time passes during the free-feed phase, as foxes become familiar with the location of bait stations and the presence of the baits. The assessment period for pre-toxic bait take was commenced once variation in daily bait take had stabilised (<10–15% variation). Contagion causes the daily frequencies of bait take to form a curve that flattens out at high values. An index of fox abundance can be achieved by checking bait stations daily and recording visitations until the curve has reached a plateau. The mean of three or more days after the plateau is reached is used as the index. This may take many days to achieve, with studies by Thompson and Fleming (1994) showing that 10 days were needed, Fleming (1997) showing 16 days, and Allen et al. (1996) showing 21 days were needed for dingoes to achieve the required results.

In association with the baiting program, changes in fox activity before and after poison baiting were measured using the presence of fox signs on sand plots. Sand plots are areas of smoothed sand covering the width of a track. They are checked daily for three consecutive days for sign of foxes, with plots smoothed over after checking (Allen et al. 1996). Plots were distributed at a density of one plot per kilometre of road/track with a minimum of 20 plots per monitoring area. The proportion of sand plots recording tracks per night between TMA and NTMA provide estimates of the efficacy of the poisoning program. Once free-feed baiting had stabilised, two sessions of sand plot monitoring were undertaken. This was repeated several days after the initial poison baiting on the TMAs.

The difference in the percentage of free-feed bait taken and the percentage of poison bait taken was determined to quantify the immediate effect of 1080 poisoning on the fox population. Figure 2 is a diagrammatic representation of the expected outcome from the bait take and sand plot monitoring program.

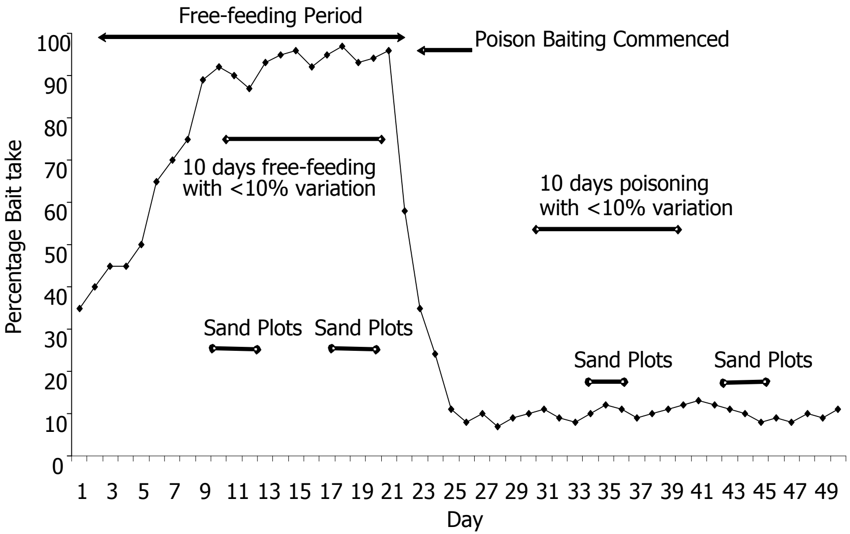


Figure 2. Steps in monitoring the effectiveness of the initial fox reduction baiting, showing an example of data that might be obtained.

2.2.1.1 Index of abundance

At each site, visitations by foxes to bait stations (or sand plots) were recorded. Investigative visitation and actual removal of baits were recorded separately, but all visitations contribute to the index. Because baits become unavailable to other foxes once bait has been removed the relationship between fox density and bait take is not linear. In addition, more than one animal may visit a station, but this will be recorded as one visit. This can be accounted for with the use of a frequency–density transformation (Caughley1997):

$$\nu = -\log_e (1 - f)$$

where f is the frequency of visitation to bait stations (or sand plots) by foxes, and v is the mean density of the occurrence of fox sign per bait station (Fleming 1997).

2.2.1.2 Percentage change

The percentage change in the pre- and post-control index of abundance was derived from the baiting data. The sand plot activity index on the treatment sites was calculated as follows:

$$\% \text{ Change} = (\text{pre-baiting index} - \text{post-baiting index}) / \text{pre-baiting index} \times 100$$

2.2.2 Sustained reduction in the fox population

Once the initial knockdown of foxes had been established, continued low levels of bait take and sand plot activity, measured at two weekly intervals on TMAs and NTMAs, provided evidence of a sustained reduction in foxes.

NTMAs also had free-feed bait laid initially to establish the presence of foxes. Free-feeding and sand plot activity monitoring was undertaken once a year between 2005 and 2010 on each NTMA. By comparing free-feed and poison bait take and sand plot activity between the yearly estimates, evidence can be provided that foxes remain unchanged on the NTMAs.

2.2.2.1 Supplementary assessment of sustained reduction in foxes using digital cameras

We assessed the occupancy rate of foxes at Lower Glenelg National Park north and south in October 2011 using data generated from 39 Reconyx PC900 digital cameras (Reconyx Inc., Wisconsin, USA) that were deployed for 21 days. Cameras were randomly deployed within randomly selected 2 km^2 blocks. Camera set up was the same as in previous years (Robley et al. 2010).

2.3 Measuring populations of key native species

Assessing changes in elusive native species abundance, presence or activity in response to fox control is problematic (McLeod et al. 2008). McLeod et al. (2008), in reviewing the effectiveness of fox control programs in Australia, noted that these difficulties have produced results that have been varied, sometimes conflicting, and hard to interpret.

While information on population-level parameters, such as age-specific fecundity and survivorship, would be ideal, these are difficult to obtain for species with a low abundance. Indices such as the minimum number of animals known to be alive (MNA) have been a popular measure of population size for mark–recapture studies of small mammals and have been used to assess responses to fox control (see e.g. Dexter and Murray 2008). The use of MNA has, however, been strongly questioned by Jolly and Dickson (1983), Nichols and Pollock (1983), Seber (1986), Montgomery (1987), Pollock et al. (1990), and Efford (1992) mainly on the basis that it is always negatively biased. In fact, Efford (1992) reports that in some cases the index could be biased by as much as 24–45%.

The use of occupancy modelling specifically accounts for spatial and temporal variability in the detection of species. It can incorporate site covariates and include the use of different detection methods, and requires relatively less effort and cost compared to more traditional methods of estimating abundance (Tyre et al. 2001; MacKenzie and Kendall 2002). To our knowledge this is the first time that site occupancy modelling has been applied to assessing the response of native species to a reduction in foxes.

Changes in site occupancy of the three target species (Long-nosed Potoroo *Potorous tridactylus*, Southern Brown Bandicoot *Isoodon obesulus*, and Common Brushtail Possum *Trichosurus vulpecula*) were monitored annually at 40 monitoring stations established within each TMA and NTMA. The selection of locations of monitoring sites was based on descriptions of the habitat of the target species (Menkhorst 1995). Site location was then stratified according to the proportion of habitat, based on EVC, within each TMA and NTMA (Appendix 1). 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 replaced each day. These daily surveys represented four repeat surveys of the monitoring site per sampling period.

Monitoring was typically carried out in spring (2005, 2007–2010). Initial sampling was conducted in winter 2005, pre-poison baiting. In 2006 sampling was undertaken in late winter due to staff resource issues, and the spring 2007 sampling was delayed due to staff being allocated to fire fighting delaying sampling until summer 2007. Severe fires affected large areas of Victoria in summer 2005–2006, which interrupted the delivery of this program. As a result, hair-tube monitoring was not completed at Mt Clay and Hotspur in spring 2006. Follow-up monitoring was undertaken in February 2007 at Mt Clay and Hotspur.

The repeat surveys allowed us to construct a detection history for each site, and thus a detection probability *p*. The above approach has been extended by MacKenzie et al. (2003) to enable the estimation of colonisation and local persistence (or extinction) probabilities of sites over multiple sampling periods, i.e. years or seasons. In the multi-period model, colonisation (γ_t) is analogous to reproduction rate, and is the probability that a site that was not occupied at time t_1 becomes occupied at t_2. Persistence ($ø_t$) is analogous to survival and is the probability that sites that were occupied at t_1 remain occupied at t_2. Extinction ε_t is simply 1 – persistence, and is the probability that a previously occupied site becomes unoccupied from one time period to the next.

Colonisation and persistence are important as they are the processes that drive occupancy. Site occupancy (ψ) in subsequent years results from the number of currently occupied sites multiplied by the probability of persistence ($\psi_t\, ø_t$), and the number of currently unoccupied sites ($1 - \psi_t$) multiplied by the probability of colonisation (γ_t). Specifically, the site occupancy at time $t + 1$ is given in terms of these colonisation and persistence parameters as:

$$\psi_{t+1} = \psi_t \phi_t + \left(1 - \psi_t\right)\gamma_t$$

i.e. occupied sites will remain occupied with probability ø, and unoccupied sites will be colonised with probability γ.

It was considered likely that the probability of detection using hair tubing (as opposed to the probability of occupancy) might have differed between years because of seasonal effects on foraging behaviour.

We modelled the occupancy rate in the six areas (three TMAs and three NTMAs) from winter 2005 to spring 2010. Initial probabilities of site occupancy at the start of the study were likely to have differed between the six study sites because of differences in habitat quality and other factors between the sites. Therefore, all models allowed for different initial rates of occupancy among the six areas.

It was considered possible over the medium term (5 to 7 years) that local rates of persistence and rates of colonisation would be higher at monitoring areas where predator control was conducted than at sites where no predator control was carried out. We estimated the effect of baiting on colonisation and persistence for each species between the treatment and non-treatment sites, with treatment effects estimated separately for each pair. Derived estimates of the overall treatment effect on persistence and colonisation for each pair was determined by taking the difference in those parameters between the TMA and NTMA. Values greater than zero indicate a positive effect of treatment. All parameters were estimated using WinBUGS 1.4 (Lunn et al. 2000). Appendix 2 provides details of the model construction.

Depending on the species, some parameters may be poorly estimated. For example, if all sites within an area were occupied, then an estimate of colonisation (γ_t) for that year will not be possible as there are no unoccupied sites to colonise. Similarly, if all sites are unoccupied, then estimates of persistence ($ø_t$) will not be possible.

For each process and area-pair, the resulting probability provides a measure of the evidence that the treatment effect is real i.e., a probability of greater than 0.99 provides strong evidence of a positive treatment effect, a probability between 0.95–0.99 provides moderate evidence of positive treatment effect, and a probability between 0.9–0.95 provides weak evidence of a positive treatment effect. Correspondingly, strong, moderate or weak evidence of a negative treatment effect are indicated by small probabilities (e.g. 0.01, 0.05 and 0.1 respectively).

The probabilities correspond to the 95% credible intervals (CI). In general, strong evidence of a positive treatment effect is indicated by the entire 95% CI being above zero. Similarly, strong evidence of a negative treatment effect is indicated when the 95% CI is below zero. Where the 95% CI includes 0, there will be either weak or no evidence for a treatment effect.

2.3.1 Logistic Regression of Southern Brown Bandicoot data

Additional analysis of the Southern Brown Bandicoot data was undertaken using logistic regression to examine the influence of a range of explanatory variables (Table 1) in predicting the presence of this species.

Logistic regression is a common tool used to estimated species distribution (Hosmer and Lemeshow 2000; Quinn and Keough 2002; Zuur et al. 2007). Several variables were considered in the full model and a backwards/forwards model selection was carried out using the corrected AIC_c due to the small sample size (Burnham and Anderson 2010). Once the best model was selected, cross-tabulation univariate models and χ^2 test for independence of each categorical predictor versus Southern Brown Bandicoot were used to check model validity and predictive capability. The significance levels were set at 5% and 25% for χ^2 tests and univariate models respectively (Hosmer and Lemeshow 2000). The levels within EVC, fox control, dominant tree, were deemed dependent with respect to Southern Brown Bandicoot presence. A Hosmer–Lemeshow test was used to evaluate any lack of fit for the model (Hosmer and Lemeshow 2000). The area under the receiver operator curve (ROC) was also calculated for the model to indicate its level of discrimination (Agresti 2002).

Site-specific explanatory variable were collected from all 240 monitoring stations across the three paired treatment/non-treatment sites in spring 2011.

Table 1. Site-specific variables collected at monitoring locations in Glenelg Ark

Explanatory Variable		**Description**
Fox Treatment		Presence or absence of fox baiting
Ecological Vegetation Class (EVC)		In accordance with DSE classifications
Fire Regime Data from DSE Corporate Data Library		Time since fire Season of fire Number of fires
Number of trees		Number of tree species at a site (within 5 m radius)
Structure	Projected Foliage	Cover Board Measurement – photographs taken of board (40 cm x 30 cm) from 30 cm above the ground at a distance of 1 m. Percentage of vegetation covering the board.
	Lateral Cover	Number of vegetation touches in five height categories: 0–20, 21–40, 41–60, 61–80, and 81–100 cm
	Leaf Litter	Presence/Absence of leaf litter touching pole from 10 sample points within a 5 m radius Litter depth measurements at each pole point (mm) from 10 sample points within a 5 m radius
	Soil Compaction	Penetrometer – measurement taken at 10 touch pole locations (Newtons/m^2)
	Coarse Woody Debris	Visual estimate of percentage of ground covered by logs and other wood debris within a 5 m radius

2.4 Rainfall

Rainfall records were collected from Dartmoor (Station no. 69055; latitude –37.26°, longitude 150.05°; approximately 20 km north of the centre of the study site). We used yearly rainfall deviations from the long-term mean based on winter–winter rain (1 July to 30 June). Since 1907, rainfall has averaged 70 mm year-1 (SD 46 mm) with a minimum of 7 mm and a maximum of 302 mm. From the beginning of the study period in 2005 until 2009 rainfall was generally below the long-term average, with a significant rainfall even in 2011 (Figure 3).

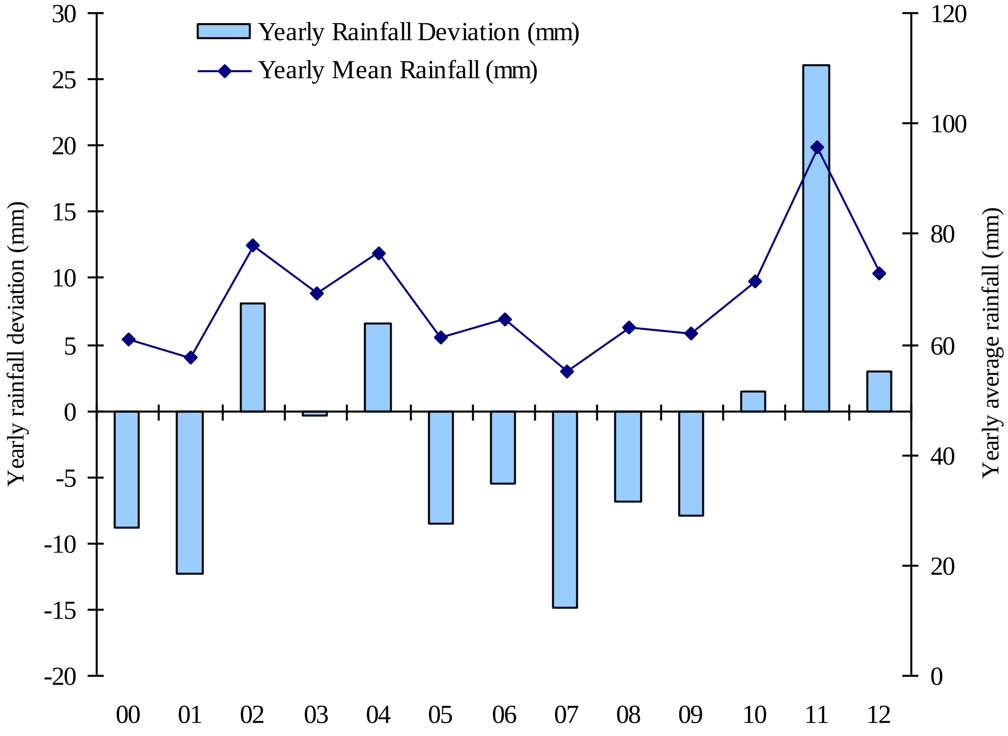


Figure 3. Yearly rainfall deviation (1 July to 30 June) for long-term average (bars) and average yearly rainfall (line).

3 Results

3.1 Response of foxes to control

3.1.1 Lower Glenelg National Park

Free-feeding in the TMA and the NTMA commenced in July 2005 and continued until late September 2005. Seventy-two poison baits were first laid in October 2005 in the TMA and were checked fortnightly, and taken baits were replaced. Free-feeding at 46 bait stations continued in the NTMA until the end of November 2005, baits being checked and replaced fortnightly.

There was a 79% reduction in the index of abundance following the implementation of the baiting program (Figure 4).

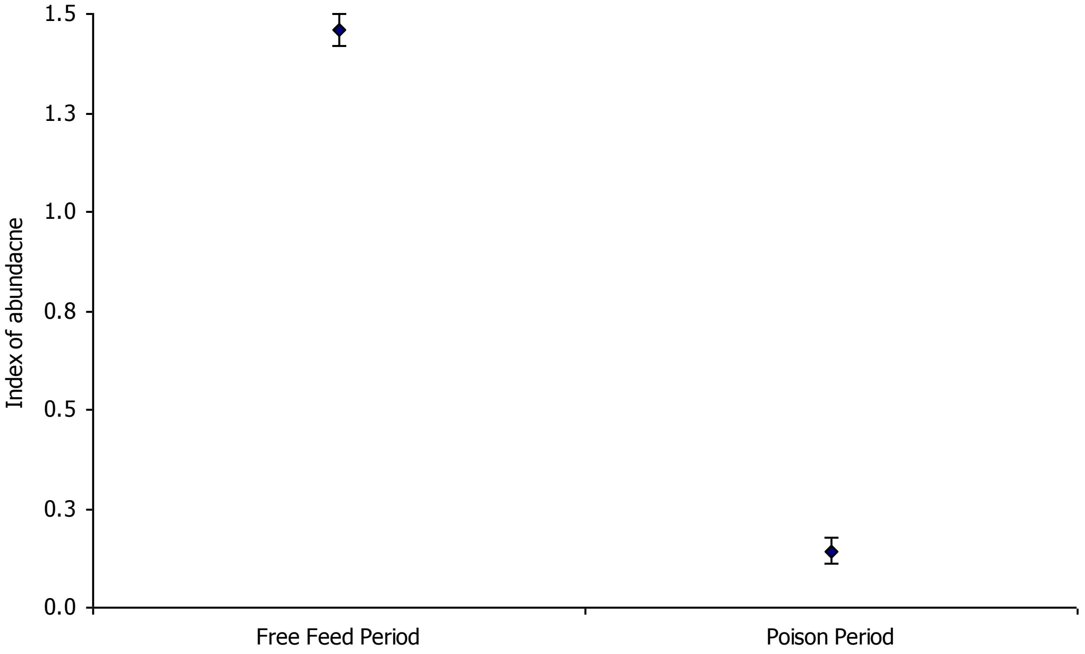


Figure 4. Index of abundance for the 3 month free-feed period (July–September 2005) and the three month poison period (October–December 2005) at Lower Glenelg National Park south. Bars are 95% binomial confidence intervals.

To assess whether a reduction in foxes was maintained in the longer term, we compared the index of abundance during the free-feed bait period in autumn 2006 to 2012 in the NTMA to the same period of poison bait take in the TMA. There was significantly less bait taken in the TMA compared to the NTMA as shown by the non-overlapping 95% confidence limits in Figure 5. Bait take in the treatment area was also less volatile compared to the non-treatment site, which displays large fluctuations in free-feed bait take. Despite these fluctuations there is a longer-term downward trend to free-feed bait take on the non-treatment site.

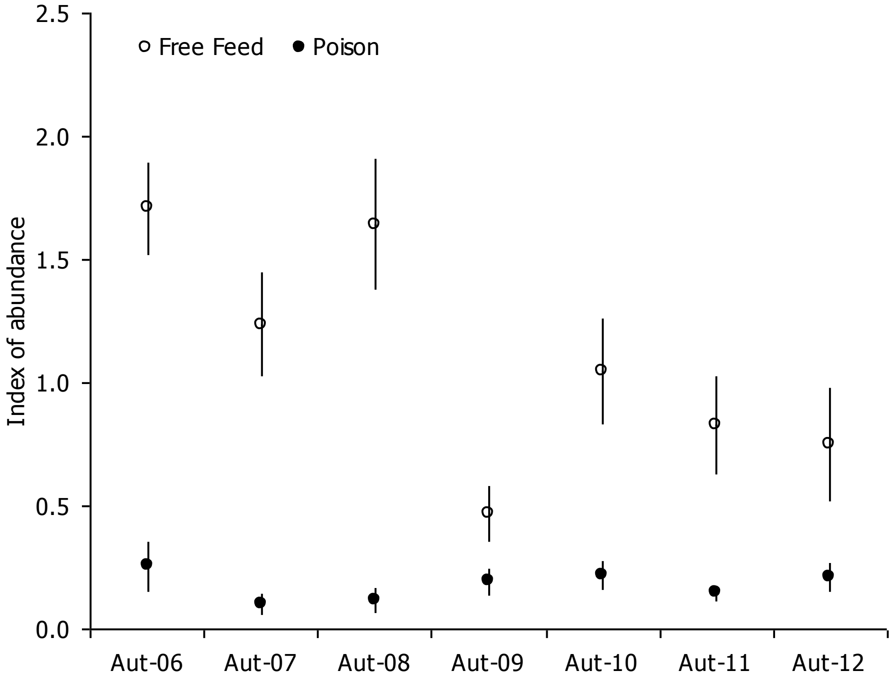


Figure 5. Index of abundance during the poisoned and free-feed period on Lower Glenelg National Park. Free-feeding occurred in autumn each year at LGNP north to compare with the baited site in the south. Bars are 95% confidence intervals.

Sand plot activity monitoring did not detect a decrease in fox activity until May 2008, 29 months after the implementation of poison baiting. The activity index has remained lower on LGNP south compared to LGNP north since May 2008 (Figure 6).

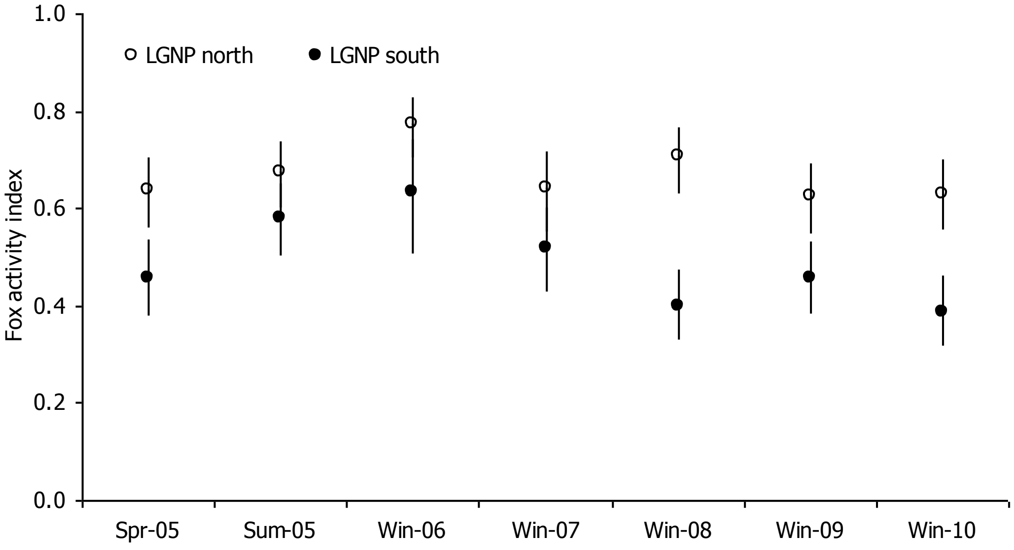


Figure 6. Fox activity index for Lower Glenelg National Park south and north. Fox baiting commenced in December 2005. Bars are 95% confidence intervals.

3.1.1.1 Camera survey for foxes

Results of the supplementary surveys to determine the difference in fox occupancy between LGNP south and north indicated a substantial difference in occupancy for foxes (Table 2; Figure 7).

Table 2. Model output for fox occupancy at Lower Glenelg National Park

Model	AIC	Delta AIC	AIC weight	Model Likelihood	*K**	-2*Log Likelihood
psi(treatment),p(.)	276.84	0	0.65	1	3	270.84
psi(.),p(.)	278.12	1.28	0.34	0.53	2	274.12
psi(treatment),p(treatment)	431.48	154.64	0	0	3	425.48
psi(.),p(treatment)	432.74	155.9	0	0	2	428.74

* number of parameters

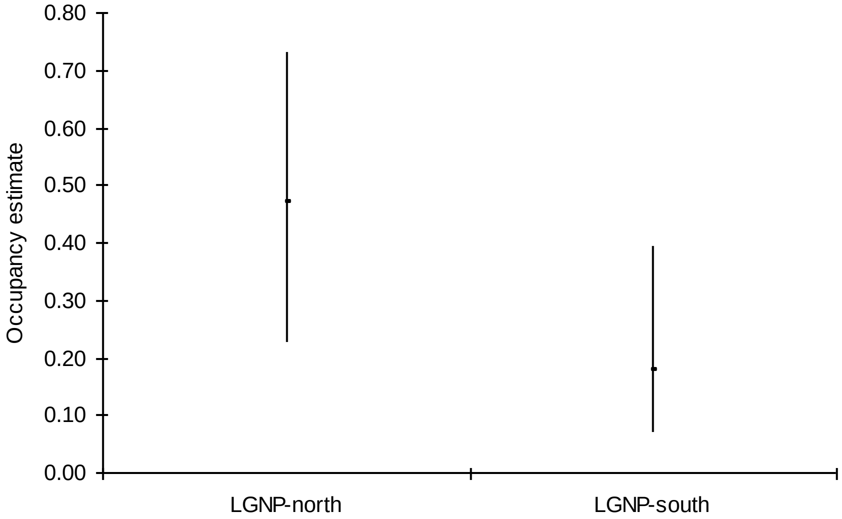


Figure 7. Derived estimates of occupancy for foxes at Lower Glenelg National Park south (baited) and north (non-baited). Mean and 95% credible limits.

The best model from the a priori set of candidate models for foxes was ψ(treatment),p(.), with estimated occupancy nearly three times higher on the non-baited north site compared to the baited south site. Model assessment indicated that the global model provided a reasonable fit to the data (*p* = 0.99, c-hat = 0.2819).

3.1.2 Cobboboonee National Park and Annya State Forest

Free-feeding commenced in July 2005 in both the TMA and NTMA. Seventy-one poison baits were first laid in October 2005 in Cobboboonee National Park. These were checked fortnightly, and taken baits were replaced. Free-feeding was continued in the 78 bait stations in Annya State Forest until November 2005. There was an 81% reduction in the index of abundance following the implementation of the baiting program (Figure 8).

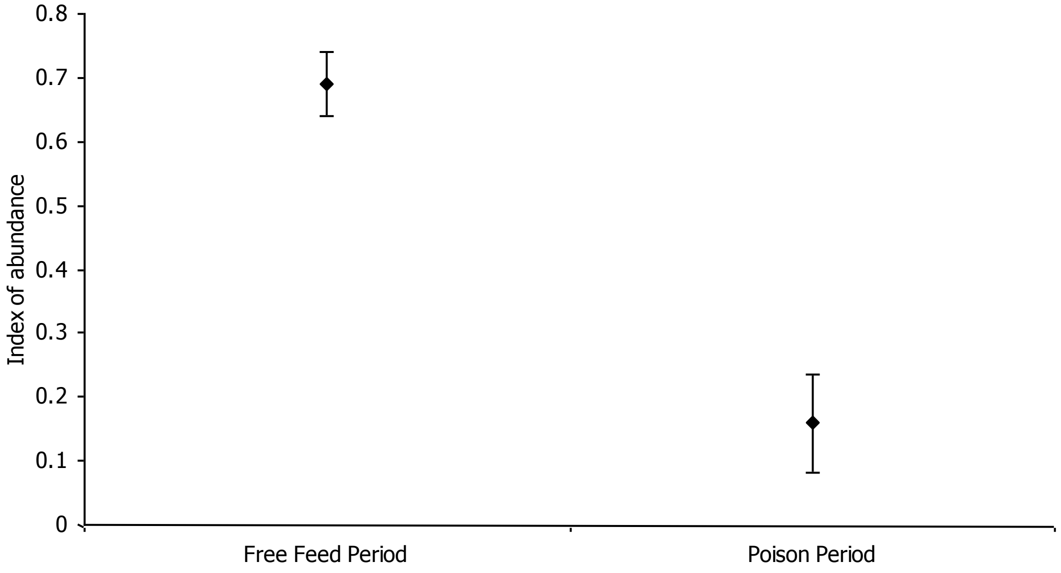


Figure 8. Index of abundance during the free-feed period (July 2005–September 2005) and poison period (October 2005–December 2008) at Cobboboonee National Park. Bars are 95% binomial confidence intervals.

The index of abundance was lower in Cobboboonee National Park in all years (2006–2012) compared to the same period in Annya State Forest (Figure 9). The rate of poison bait take between years remained constant while there was a marked increase in bait take on the non-treatment sites in 2011 and 2012.

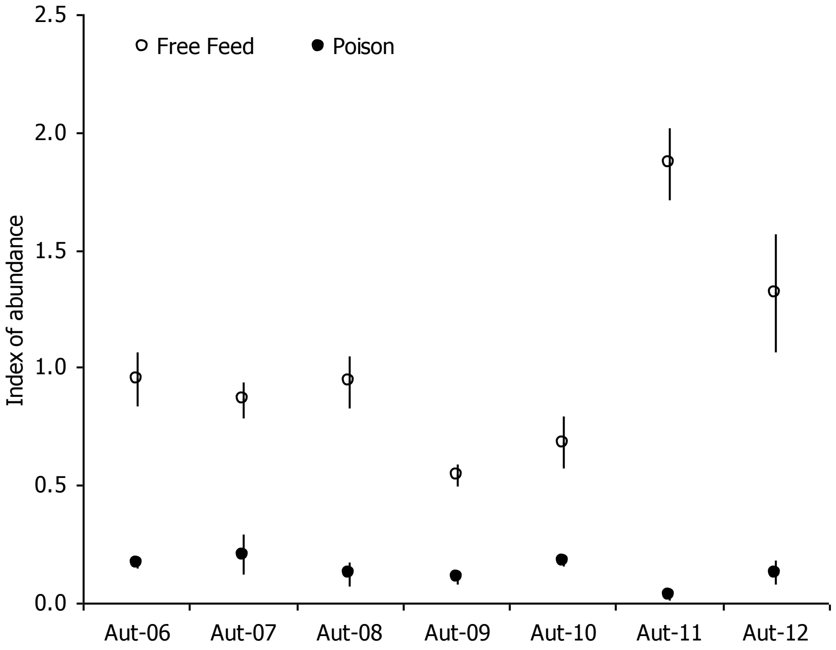


Figure 9. Index of abundance during poisoned and free-feed baiting at Cobboboonee National Park and Annya State Forest. Free-feeding occurred in autumn each year at Annya State Forest. Bars are 95% binomial confidence intervals.

Sand plot activity monitoring of foxes detected a difference in fox activity between the two sites prior to the baiting program being implemented in December 2005. The activity index has remained lower at Cobboboonee compared to Annya since the commencement of the baiting program (Figure 10).

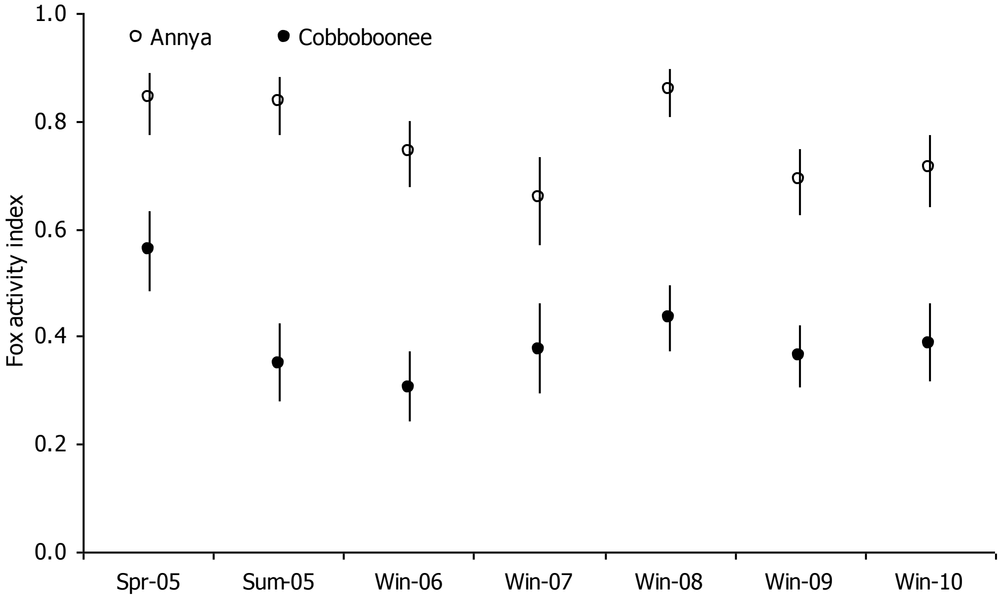


Figure 10. Fox activity index for Annya State Forest and Cobbboboonee National Park. Fox baiting commenced in December 2005. Bars are 95% confidence limits.

3.1.3 Mount Clay and Hotspur State Forest

Free-feeding commenced in July 2005 in both the TMA and the NTMA. Forty-five poison baits were first laid in October 2005 at Mount Clay. Free-feeding continued in the 48 bait stations at Hotspur State Forest until the end of November 2005.

There was a reduction in the index of abundance of 76% following the implementation of the baiting program (Figure 11).

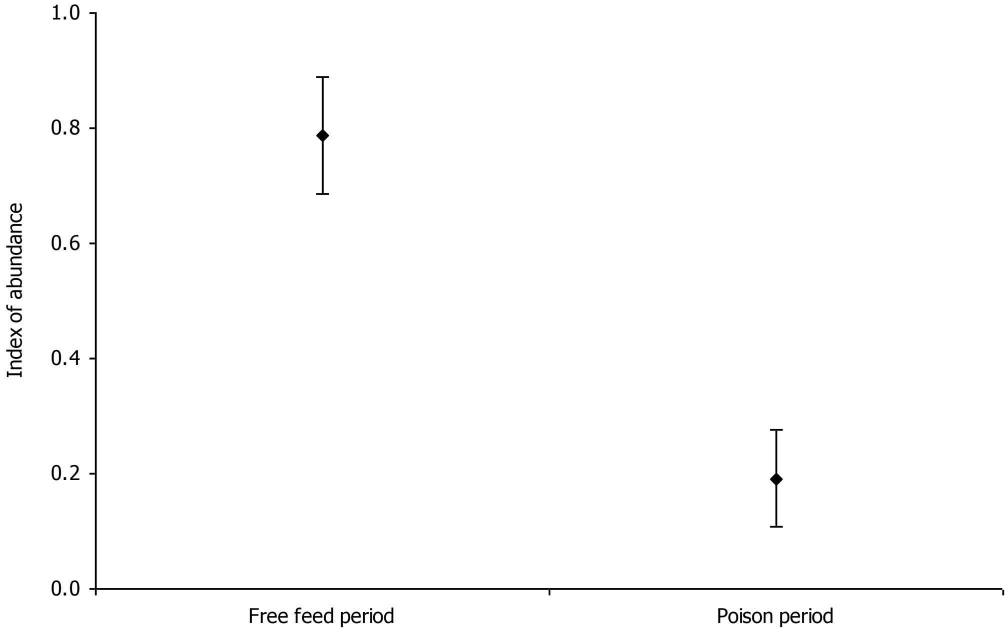


Figure 11. Difference in the proportion bait taken for the free-feed period and poison period at Mount Clay State Forest. Bars are 95% confidence intervals.

The index of abundance was less on the TMA compared to the NTMA (Figure 12). A decision was made not to continue to collect free-feed data from this monitoring pair in 2007 and 2008 because of limited resources.

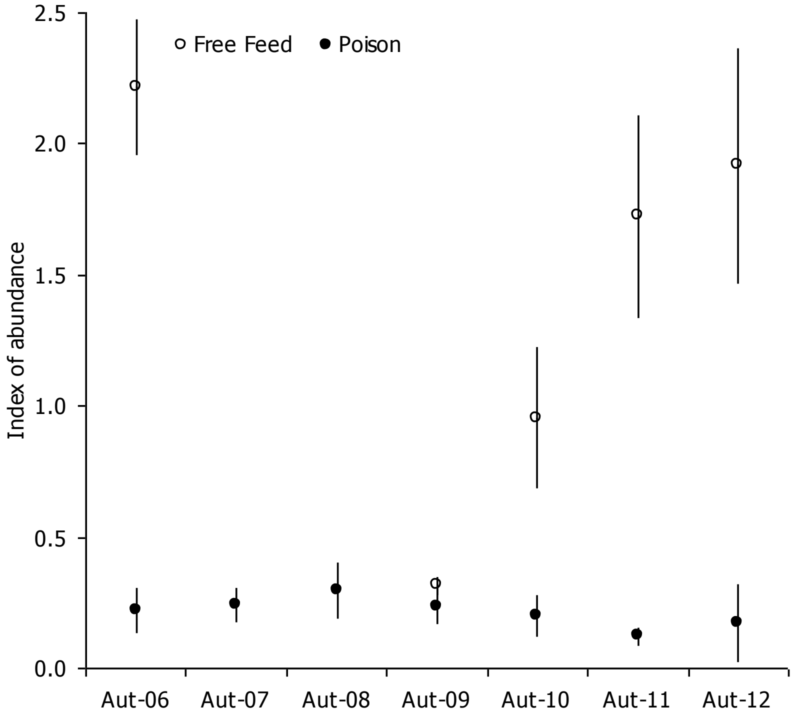


Figure 12. Index of abundance between free-feed baiting at Hotspur State Forest and poisoning at Mount Clay State Forest. Bars are 95% confidence intervals.

Sand plot activity monitoring commenced on the Mt Clay State Forest and Hotspur State Forest pair in May 2009 and was repeated in May 2010. Results indicate that fox activity was higher at Hotspur in both years (Figure 13).

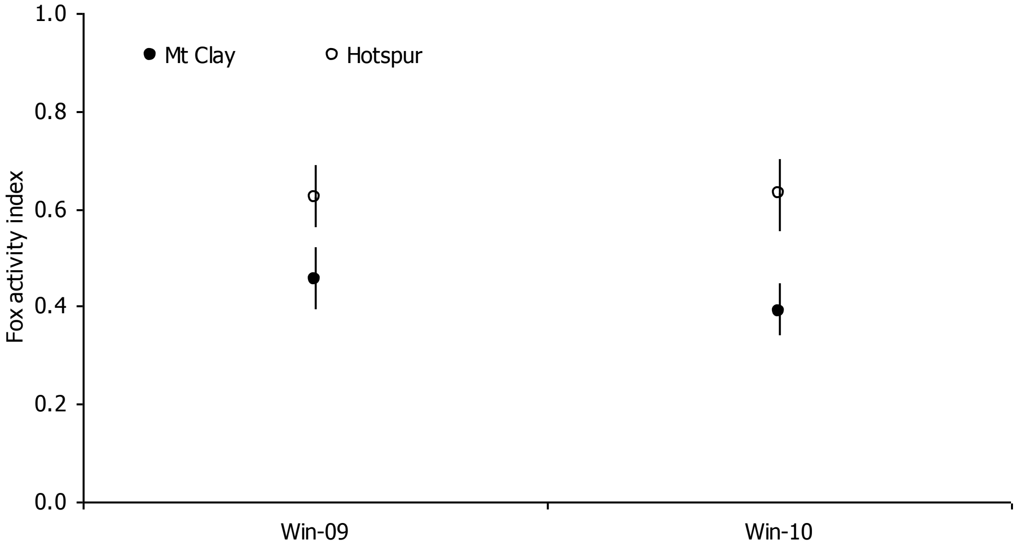


Figure 13. Fox activity index for Mt Clay and Hotspur State Forest. Fox baiting commenced in December 2005. Bars are 95% confidence intervals.

4 Monitoring changes in native species

In each of the sampling years the main species of interest (Long-nosed Potoroo, Southern Brown Bandicoot and Common Brushtail Possum) were detected. Figures in Appendices 3 and 4 show the monitoring station and year of detection for Long-nosed Potoroos and Southern Brown Bandicoots respectively.

4.1 Long-nosed Potoroos

There was strong evidence for an effect of fox control on persistence at sites occupied the previous year and on colonisation of new sites at Mt Clay (Figures 14, 15; Table 3). The probability of a positive effect of treatment on colonisation at Annya/Cobboboonee was not significant (0.85; Table 3); it is higher than the LGNP-south/north pair where there is no indication of a treatment effect.

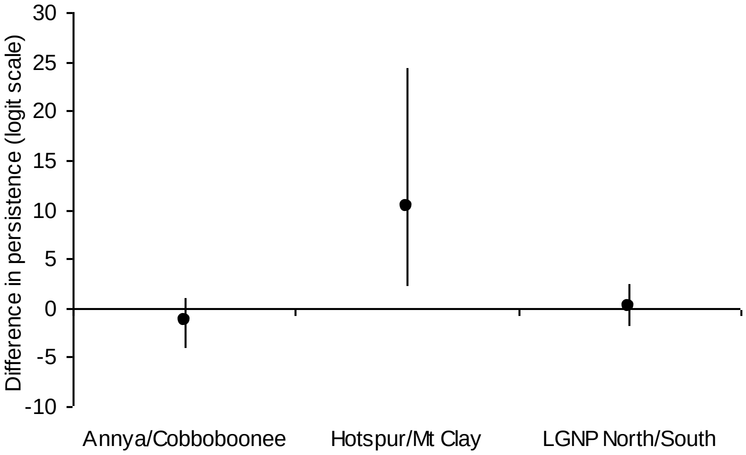


Figure 14. Effect of treatment on persistence of Long-nosed Potoroos at each non-treatment/treatment pair. Bars are 95% CI.

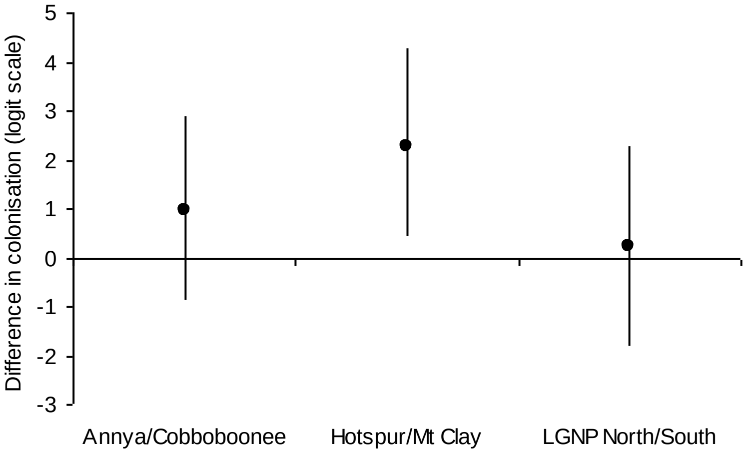


Figure 15. Effect of treatment on colonisation of Long-nosed Potoroos at each non-treatment/treatment pair. Bars are 95% CI.

Table 3. Probability of positive effect of treatment on Long-nosed Potoroo at each non-treatment/treatment pair.

Parameter	Annya/Cobboboonee	Hotspur/Mt Clay	LGNP north/south
Persistence	0.14	0.99	0.56
Colonisation	0.85	0.99	0.60

These results indicate that at Mt Clay, Long-nosed Potoroos are occupying new sites, and the probability they persist at those new sites is high. While persistence is greater at Cobboboonee relative to Annya, potoroos are not colonising new sites at any greater rate than on the non-treatment site. It may be that fox control is preventing potoroo decline (persistence) but is insufficient to allow colonisation of new sites, or colonisation is limited by other factors. At LGNP south and north, persistence and colonisation rates are not affected by treatment (Figures 16 and 17).

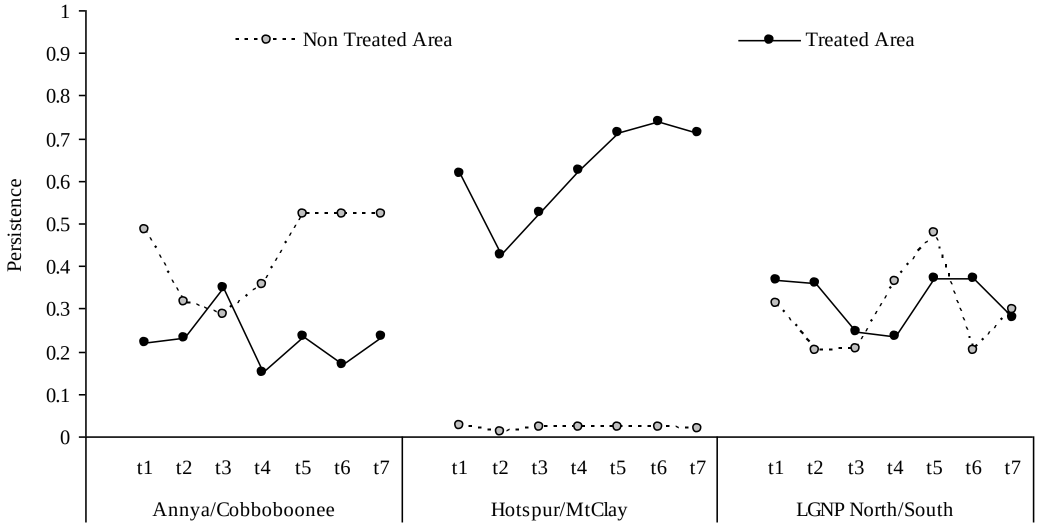


Figure 16. Derived estimates of mean persistence for Long-nosed Potoroo at each non-treatment/treatment pair.

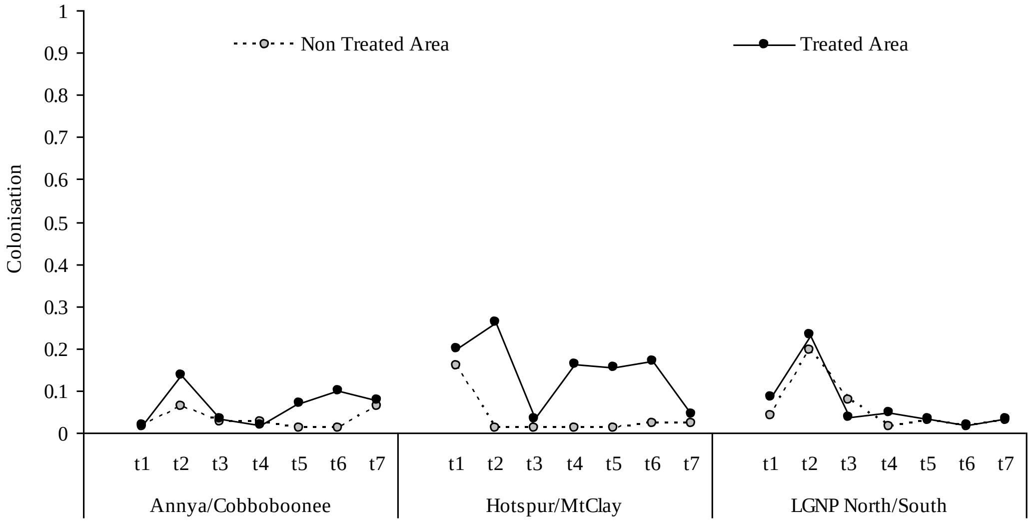


Figure 17. Derived estimates of mean colonisation for Long-nosed Potoroo at each non-treatment/treatment pair.

The derived estimates of occupancy, which takes into account the combined effects of persistence and colonisation, show that from summer 2006 onwards the probability of a site being occupied by Long-nosed Potoroos at Mt Clay was greater than that at Hotspur (Figure 18). Estimated probabilities at Annya/Cobboboonee suggest a trend towards increased occupancy rates over the past two years; however, this is not statistically significant. Rates of occupancy on both LGNP north and south have been in decline since summer 2006 at both the treatment and non-treatment sites.

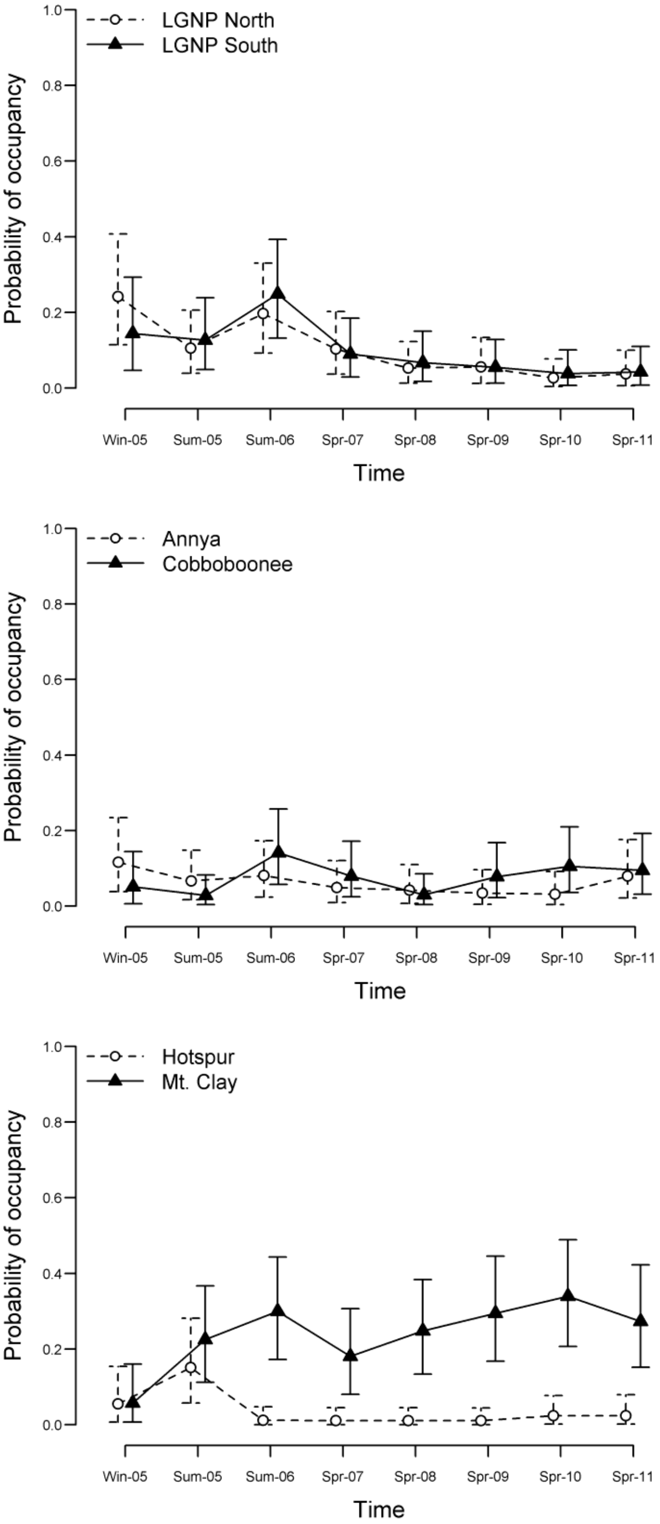


Figure 18. Derived estimates of occupancy for Long-nosed Potoroo at each non-treatment/treatment pair. Bars are 95% CI.

4.2 Southern Brown Bandicoot

The effect of fox control on Southern Brown Bandicoot persistence from year to year was significant at the Hotspur/Mt Clay pair, and the probability of colonisation was high (0.81) but not significant. There was a significant effect of fox control on colonisation at the Annya/Cobboboonee and Lower Glenelg National Park pair (Figures 19, 20; Table 4).

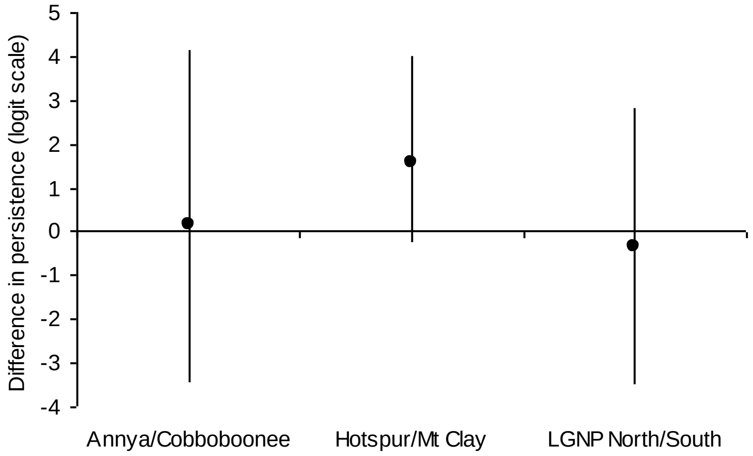


Figure 19. Effect of treatment on persistence for Southern Brown Bandicoot at each non-treatment/treatment pair. Bars are 95% credible intervals.

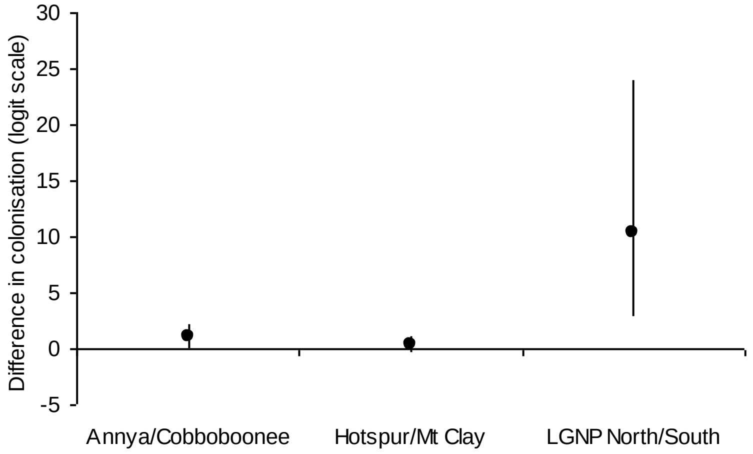


Figure 20. Effect of treatment on colonisation for Southern Brown Bandicoot at each non-treatment/treatment pair. Bars are 95% credible intervals.

Table 4. Probability of a positive effect of treatment for Southern Brown Bandicoot at each non-treatment/treatment pair.

Parameter	Annya/Cobboboonee	Hotspur/Mt Clay	LGNP north/south
Persistence	0.55	0.95	0.40
Colonisation	0.99	0.81	0.99

There is little difference in the derived estimates for persistence and colonisation through time between treatment/non-treatment pairs, with the probability of persistence and colonisation neither increasing nor decreasing with time (Figures 21, 22). Persistence and colonisation rates are consistently higher at LGNP south, suggesting that at the very least fox control has prevented a decline in these parameters. At Cobboboonee, the chances that Southern Brown Bandicoots persist at previously occupied sites was higher than at Annya, while there was no difference in the probability of colonisation between the pairs.

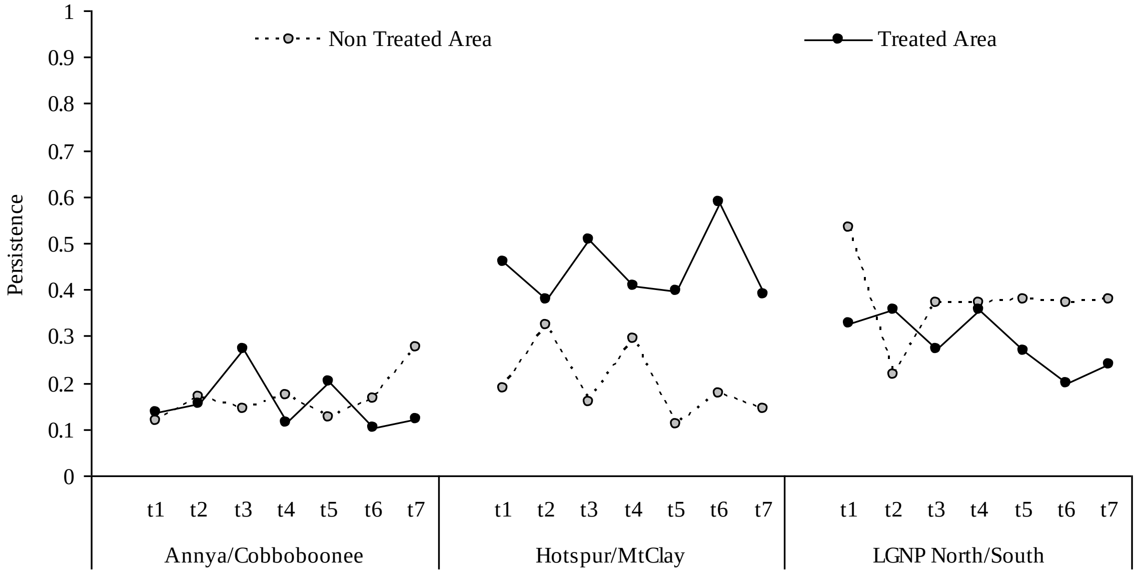


Figure 21. Derived estimates of mean persistence for Southern Brown Bandicoot at each non-treatment/treatment pair.

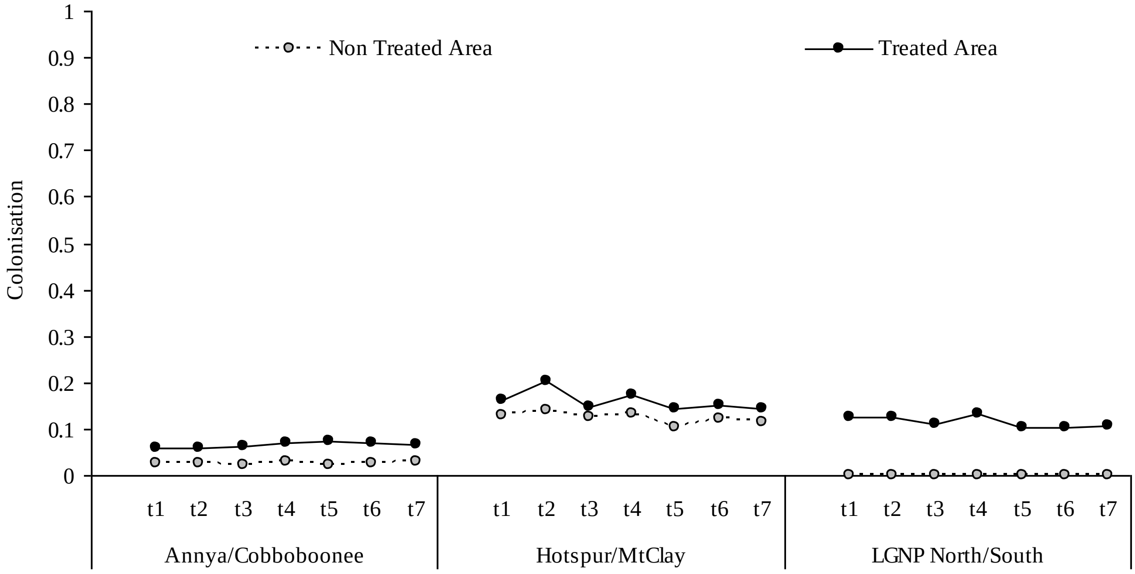


Figure 22. Derived estimates of mean colonisation for Southern Brown Bandicoot at each non-treatment/treatment pair.

The derived estimates of occupancy show that from summer 2006 onwards the probability of a site being occupied by Southern Brown Bandicoots was greater at LGNP south than that at LGNP north (Figure 23). Estimated probabilities of occupancy at Annya/Cobboboonee and Hotspur/Mt Clay are higher but the overlapping 95% credible intervals indicate these are not statistically different.

For the Annya/Cobboboonee pair, the probability of colonisation and persistence is significantly higher at the treatment area (Cobboboonee). The occupancy probability of Southern Brown Bandicoots, however, remains relatively low due to a combination of low colonisation and persistence rates. They are, however, both higher than at the non-treatment area (Annya). For the LGNP areas, the probability of colonisation at the treatment area (LGNP south) is significantly higher, although no difference is observed in persistence. This result occurs due to the zero detections of Southern Brown Bandicoot at LGNP north for the last few years resulting in estimates of persistence that is uniform between 0 and 1 (i.e. equivalent to the prior).

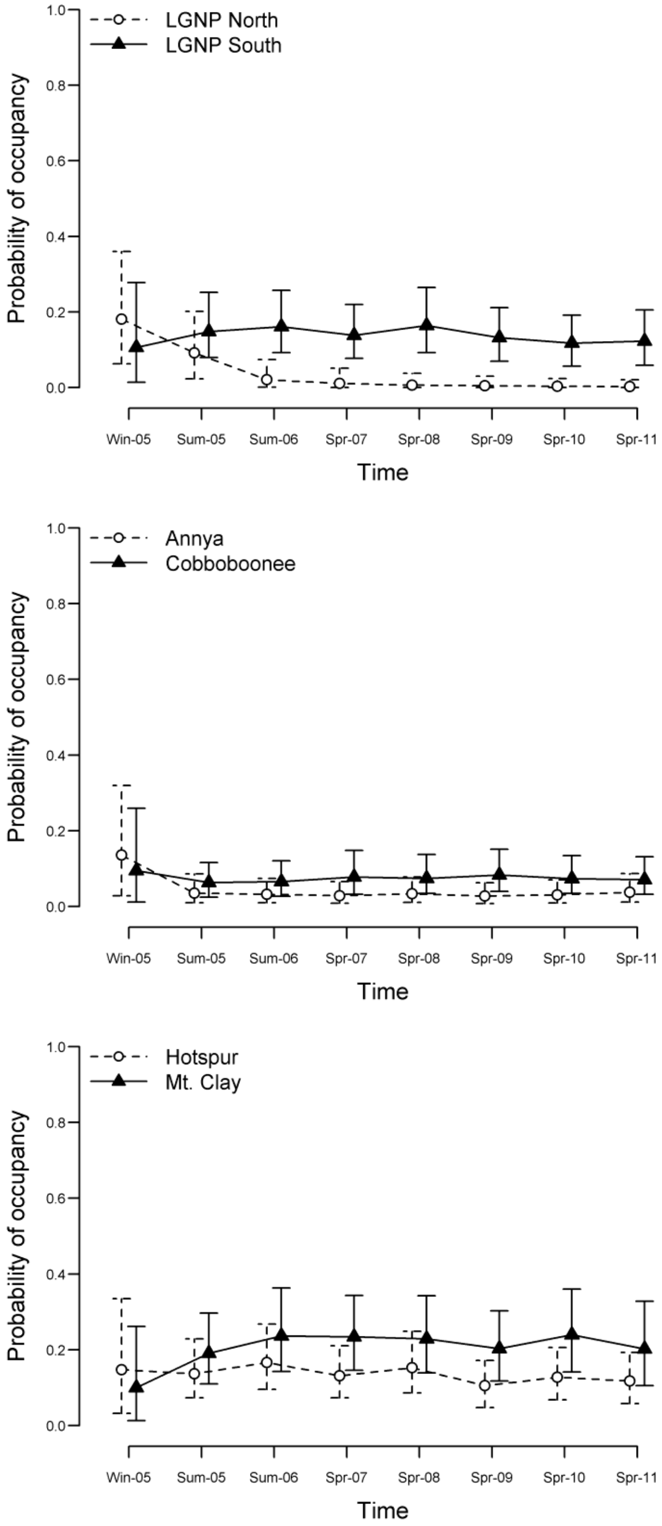


Figure 23. Derived estimates of occupancy for Southern Brown Bandicoot at each non-treatment/treatment pair. Bars are 95% credible intervals.

4.2.1 Additional analysis of Southern Brown Bandicoot data

Boxplots of the covariates (separated by presence) indicated that per cent lateral cover had the greatest difference, with coarse woody debris and soil approaching significance. Fitting the univariate models for each covariate suggested that, at least initially, percentage lateral cover squared, percentage vertical cover, litter depth and the number of trees may also be included in the model. The model included fox control, EVC, dominant tree, and per cent lateral coverage squared. The most parsimonious model estimates for the parameters in the best model are:

$$\begin{aligned}\operatorname{logit}\left(\hat{\pi}\right) &= 1.57\,FoxControl - 0.47\,EVC_2 - 0.80\,EVC_3 - 1.83\,EVC_{15} - 2.28 \\ &\quad + 1.27\,DominantTree_{\text{Manna}} + 0.94\,DominantTree_{\text{StringyBark}} + 0.0002\,LatCov^2\end{aligned}$$

The model provided an adequate fit to the data (Hosmer–Lemeshow test 0.2). There is insufficient evidence to suggest a lack of fit of the model to the data. For the model, using 0.5 as an intuitive cut-point, the sensitivity (correctly classifying presence) was 52.9% and the specificity (correctly classifying absence) was 79.9%. The area under the ROC curve was 0.79, which we consider to be an acceptable level of discrimination between sites with and without Southern Brown Bandicoots. These factors combined and no pattern in the residuals means that there is insufficient evidence to reject any of the assumptions underlying the model.

As can be seen from Table 5, fox control increased the odds ratio of the presence of Southern Brown Bandicoots by an estimated factor of 4.82 when compared to areas without fox control. Similarly, in EVC 15 the odds ratio is about one sixth (factor of 0.16) of the ‘Other’ EVCs surveyed. As the percentage lateral coverage increases, the odds ratio of Southern Brown Bandicoots increases. For instance, an increase of 10% in the percentage of lateral coverage is estimated to increase the odds ratio of Southern Brown Bandicoots by 2.5% (factor of 1.025) if all other variables remain fixed. The model predicts 84.5% likelihood of Southern Brown Bandicoots being present in an area with fox control, in an EVC classified as ‘Other’, with Manna Gums being a dominant tree and the lateral cover of 67.53% (the upper quartile). Finally, the model predicts 2.3% likelihood of Southern Brown Bandicoots being present in an area without fox control, in herb-rich woodlands, with the dominant tree not Manna Gums or Stringy Bark and the lateral cover of 39.65% (the lower quartile).

The dominant tree classes were predominantly Stringy Bark (111) and Manna Gum/Stringy Bark (69). We therefore collapsed the set into three groups: ‘Manna Gum Dominant’, ‘Stringy Bark Dominant’ (without Manna Gum) where the remaining dominant trees form the group ‘Other’.

Table 5. The odds ratios for each variable in the final model, including 95% confidence intervals.

Variable	Value	Odds Ratio	95% CI	
Fox Control	No	1.00		
	Yes	**4.82**	2.46	9.88
EVC	2	**0.62**	0.18	2.13
	3	0.45	0.16	1.25
	15	0.16	0.05	0.53
	'Other'	1.00		
Dominant Tree	Manna Gum	**3.57**	1.13	12.79
	Stringy Bark	2.56	0.86	8.90
Percentage Lateral Coverage	Quadratic	**1.0002**	1.00008	1.0042

4.3 Common Brushtail Possum

There is strong evidence of a significant treatment effect on persistence (Table 6) and colonisation (Table 6) at the LGNP north and south pair of sites, and for colonisation at the Annya/Cobboboonee pair but there were no differences at the other site pairs (Figure 24). The treatment effect on colonisation is strongly negative at the Hotspur/Mt Clay pair (Figure 25; Table 6). These results are apparent in derived estimates of persistence and colonisation (Figures 26, 27). Rates of persistence are similar between all pairs of sites, whereas rates of colonisation differ. The derived estimates of occupancy show a difference between all the pairs, although the difference between the Hotspur/Mt Clay pair is reversed due to the strong apparent negative effect of treatment on colonisation.

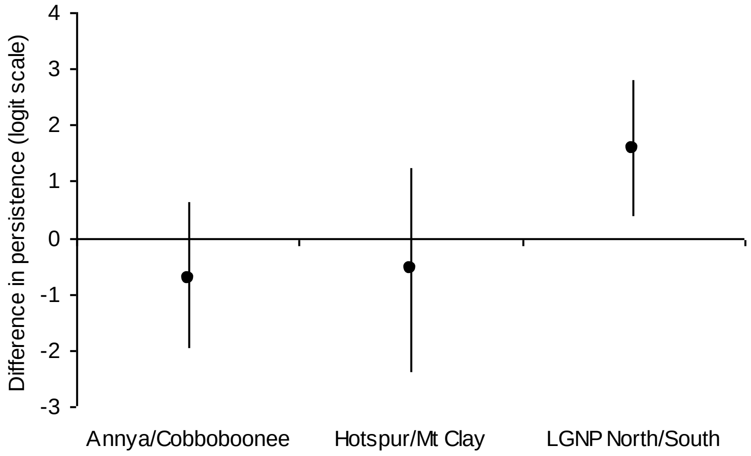


Figure 24. Effect of treatment on persistence for Common Brushtail Possum at each non-treatment/treatment pair. Bars are 95% credible intervals.

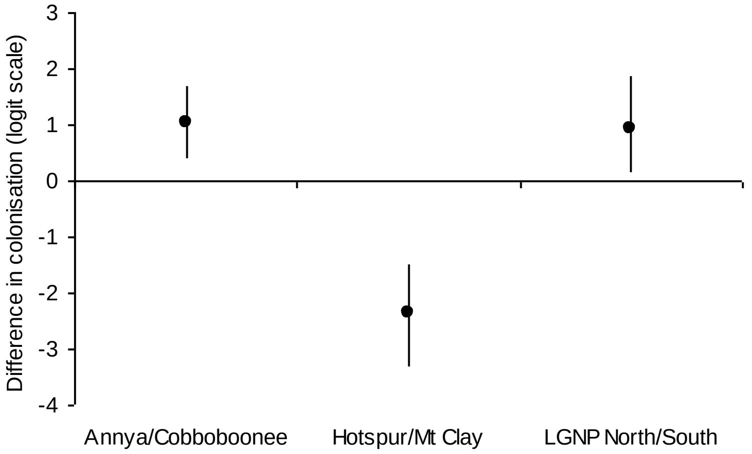


Figure 25. Effect of treatment on colonisation for Common Brushtail Possum at non-treatment/treatment pair. Bars are 95% credible intervals.

Table 6. Probability of a positive effect of treatment for Common Brushtail Possum at each non-treatment/treatment pair.

Parameter	Annya/Cobboboonee	Hotspur/Mt Clay	LGNP north/south
Persistence	0.12	0.27	0.99
Colonisation	0.99	<0.001	0.99

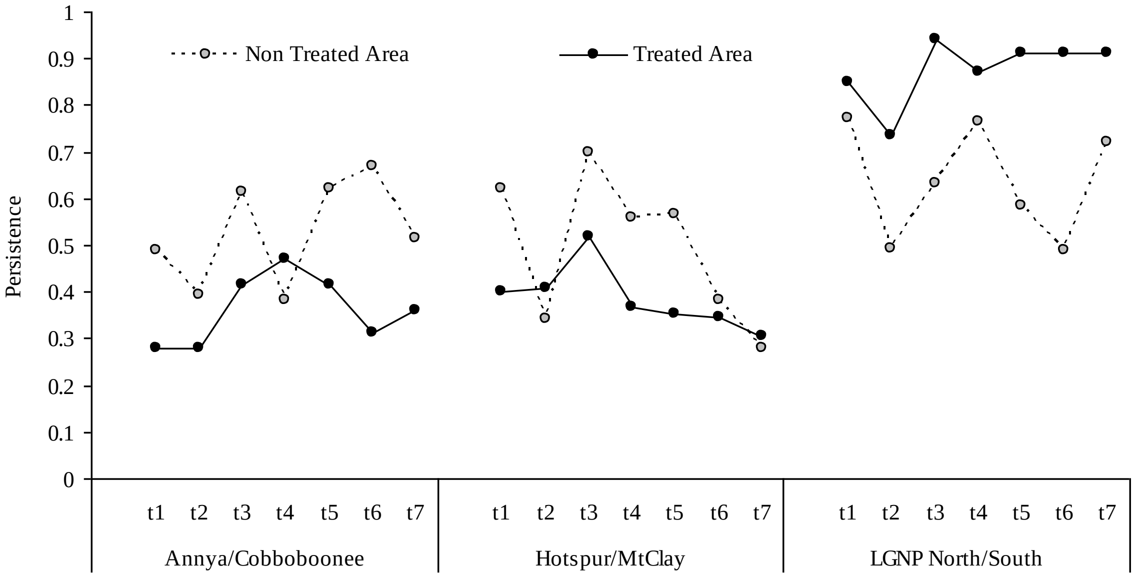


Figure 26. Derived estimates of mean persistence for Common Brushtail Possum for each non-treatment/treatment pair.

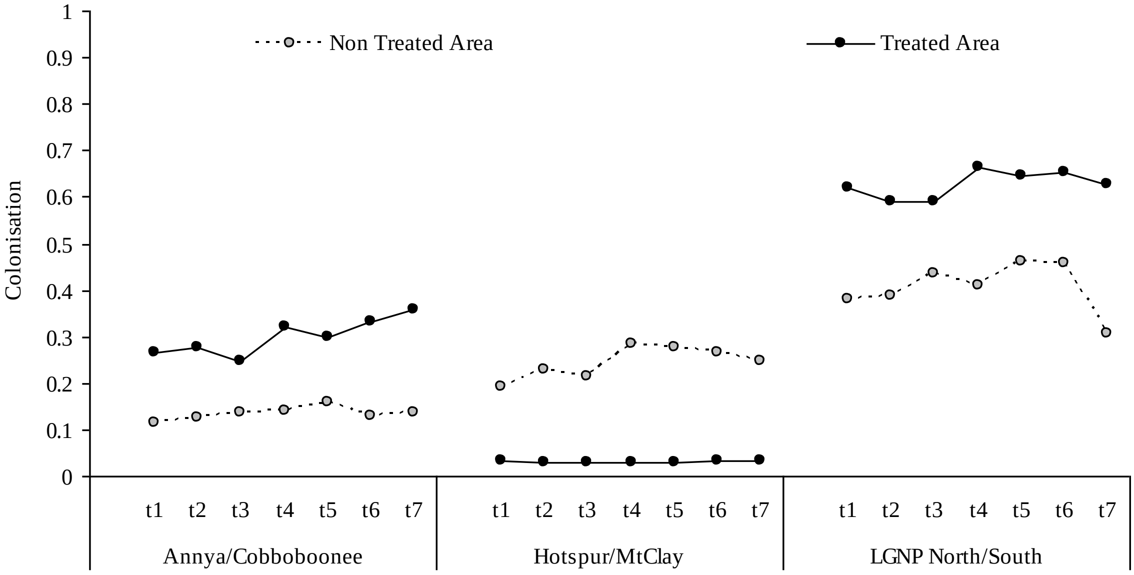
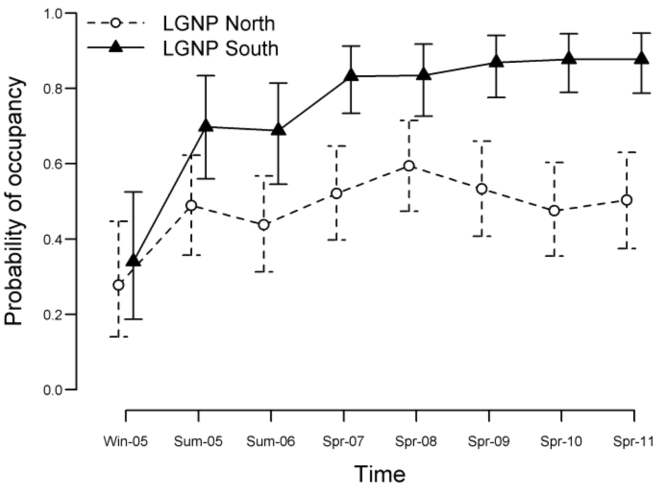


Figure 27. Derived estimates of mean colonisation for Common Brushtail Possum for each non-treatment/treatment pair.

The derived estimates of occupancy show no differences in the probability of a site being occupied by Common Brushtail Possums at Annya/Cobboboonee (Figure 27) despite a treatment effect on colonisation. This is possibly due to the lower probability of persistence at the non-treatment site (Cobboboonee). At LGNP north and south, divergence has occurred in the past two years due to a positive effect of treatment on both colonisation and persistence. At the Hotspur/Mt Clay pair, the result is reversed with very low probability of occupancy on the treatment site (Mt Clay) despite similar probabilities of persistence (Figure 28). The effect appears to be due to a higher probability of colonisation on the non-treatment site (Hotspur) and near zero probability on the treatment site (Mt Clay).



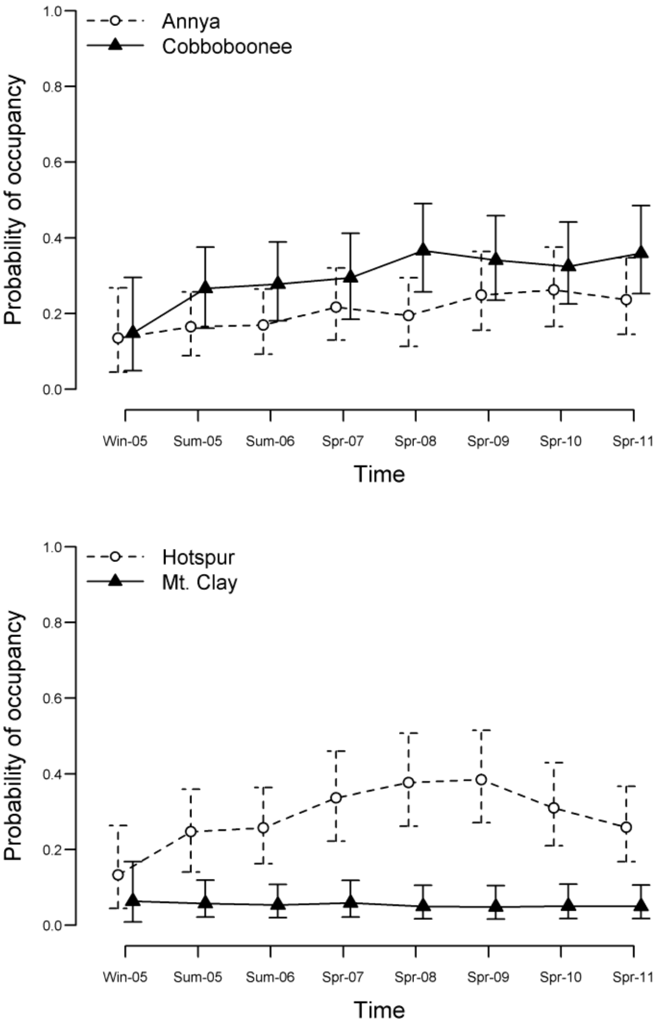


Figure 28. Derived estimates of occupancy for Common Brushtail Possum for each non-treatment/treatment pair. Bars are 95% credible intervals.

5 Discussion

The monitoring program for Glenelg Ark achieved its aims of quantifying the response of foxes to control actions and the response of targeted native species. As the monitoring program receives further information, certainty around the nature of these responses is expected to increase.

Indices of abundance derived from bait take data, sand plot data and cameras surveys (LGNP only) at all three treatment areas (LGNP south, Cobboboonee National Park and Mt Clay State Forest), demonstrate a sustained lower abundance of foxes.

Independence of the treatment and non-treatment sites is an important issue as we compare rates of change in foxes and native species between sites. At Glenelg Ark, the distance between treatment and non-treatment areas was more than 10 km. We saw no reduction in foxes on the non-treatment sites (as indexed by abundance indices) in Glenelg Ark, indicating that the treatment/non-treatment sites are independent from the control imposed on the treatment sites. This is particularly important for the Glenelg National Park sites, suggesting that the Glenelg River is an effective barrier to the movement of foxes and hence allows a clearer interpretation of results.

Two of the three targeted species had a greater probability of occupancy at sites with fox control: Southern Brown Bandicoots at Lower Glenelg National Park, and Long-nosed Potoroos at Mt Clay. There was no overall effect of treatment on occupancy rates for Common Brushtail Possums. Further evidence of an association between fox control and species' response was provided by the logistic regression on the 2011 Southern Brown Bandicoot data. While other factors (EVC and dominant tree species) were important, the probability of Southern Brown Bandicoots occurring at sites with fox control was nearly five times higher than on sites without. Three EVCs (2, 3 and 15) accounted for most of the sites, with the remaining 27 forming a group called 'Other'. It is possible that the selection of these groupings could be improved with smaller like EVCs combined, rather than pooled in an 'Other' category. Fire regime (time since fire, season of fire and number of fires) did not influence the prediction of presence for Southern Brown Bandicoots. However, the interval between fires may be a more important factor worth consideration in future analysis. Fire regime may better reflect the changes in habitat structure likely to influence the dynamics of native mammal species.

Colonisation and persistence can be considered as surrogates for birth and death rates (the inverse of persistence is extinction; MacKenzie et al. 2006). Birth and death rates are key functional aspects of population dynamics, and understanding which processes are being affected by fox predation is important to developing appropriate management strategies.

At LGNP south, bandicoots were able to survive at sites (persistence) but colonisation rates were not any greater than at the non-control site. This suggests that bandicoots are able to find refuge from predation at specific locations in the landscape in the treatment area but are currently unable to exploit new locations. Factors may include predation by foxes (despite the control program) and feral cats. Feral cats were in relatively higher abundance compared to the non-treatment site (Robley et al. 2010). This supports work by Risbey et al. (2000) who reported a positive response to fox control in populations of feral cats, which then impacted on the preferred prey species. It may also be possible that predation is regulating bandicoots and there is a limit to the availability of appropriate food or nesting locations. The overall derived estimate of occupancy (combined colonisation and persistence) indicates that bandicoots have declined markedly at the non-treatment site and increased marginally at the treatment site, this suggests that they have not significantly responded to the reduction in foxes, and fox predation or some other factor has precipitated the decline on the non-treatment site.

The probability of Southern Brown Bandicoots colonising sites at Cobboboonee National Park was significant, but persistence levels for this species at Cobboboonee were low at 55%. This suggests that Southern Brown Bandicoots are able to survive and occupy new sites in the presence of low levels of predation, but were unable to survive or establish in those new locations. The overall derived estimate of occupancy suggests that bandicoots have remained stable on the treatment site and declined on the non-treatment site.

At Mt Clay, bandicoots were able to persist in parts of the landscape, but colonisation rates were relatively high but not significantly greater on the treatment site, suggesting they were susceptible to predation or some other factor that prevented colonisation of new sites relative to Hotspur State Forest. The overall occupancy estimates indicate that bandicoots have increased slightly in the years following fox control, although the effect is not statistically significant.

Long-nosed Potoroos had higher rates of colonisation and persistence at Mt Clay than at Hotspur, suggesting that relative to Hotspur, potoroos were able to escape regulation by foxes. Colonisation and persistence rates at the remaining sites were dissimilar suggesting that at these sites Long-nosed Potoroos are limited by something other than fox predation, or that the level of fox reduction achieved is insufficient to allow this species to increase.

If foxes were the key limiting factor for bandicoots, potoroos and possums then the expectation would be that populations of these species would respond positively at all treatment sites, albeit at different rates. However, the data indicate that this is not the case, suggesting that:

a) the fox control has not reduced predation pressure sufficiently at some sites to release populations from limitation, or that the baiting and monitoring program has not run long enough;
b) predation by feral cats and/or other predators (e.g. raptors) is a more significant factor; or
c) resources are limiting at some sites (e.g. food and shelter) and predation is not a factor.

Changes to the composition of the predator assemblage can result in altered rates of predation on prey species. Supplementary fox control that captured feral cats as a non-target species and camera survey data indicate that feral cat populations on the treatment site at LGNP were three times higher than on the non-treatment site. While no pre-fox control data are available on cats at these sites, it suggests that cats may have at least partly replaced foxes as the apex mammalian predator.

Below long-term average rainfall was recorded across the general study area between 2005 and 2009, with only average rainfall recorded in 2010. Drought-breaking rain followed in 2011. The lower than average rainfall may have contributed to the responses in the three native mammal species. Fox control may have allowed populations to persist in areas within the treatment sites that provide suitable drought refuge, while on the non-treatment sites both fox control and drought combined to maintain or even lower existing populations. Some evidence for this can be seen in the results for Southern Brown Bandicoots at LGNP. Bandicoots on the non-treatment site decreased from 2005 to 2006 and have remained at low levels, while on the treatment site bandicoots showed signs of an increase over the same period. A similar pattern can be seen at the Annya/Cobboboonee pair. Significant rainfall in 2011 may result in increases in colonisation rates in the fox control areas leading to higher overall levels of occupancy for Southern Brown Bandicoots and Long-nosed Potoroos in 2012.

The results of the monitoring and evaluation program clearly demonstrate that foxes have been reduced and remain at relatively low levels, and that the use of occupancy modelling as applied in this project is capable of measuring changes in native species. However, the variable results suggest that blanket broadscale fox control may not be the most effective approach to recovery or protection of species threatened by fox predation, and that a greater understanding of both bottom-up and top-down processes may lead to more effective and targeted management of species that are rare in the landscape.

More complex models could be used to explore additional factors of occupancy and detection processes, including:

- interseasonal variation in detection probability due to weather or other factors. Lumsden et al. (2007) previously documented significant levels of inter-annual variation in the detection probabilities of Long-footed Potoroos (*Potorous tridactylus*) in a study conducted in north-eastern Victoria using hair tubes. Detection probability in our study was assumed to be constant across sites and within years

- differences in probability of occupancy among sites in an area due to habitat variables, such as vegetation, rainfall, temperature, exposure to solar radiation and fire
- other predictors of colonisation and extinction such as measured predator activity levels, spatial effects such as adjacency to known occupied or unoccupied sites, and the impact of fuel reduction burning and wildfires.

Developing multi-species dynamic occupancy models would explicitly link the probabilities of foxes and native species occupying areas, allowing for more direct estimates of the link between a reduction in fox (and potentially the presence of other predators) abundance and any increase in native species. These models could include site-specific covariates to help understand the non-uniform response of target species.

One limitation on the interpretation of the results from the current design is that we lack sufficient pre-baiting data to draw robust conclusions about changes in occupancy post-baiting. Non-manipulative studies have demonstrated that predation plays a role in limiting primary prey populations, but they cannot be used to unequivocally assess whether predation is a limiting factor because of the potential confounding effect of other factors. Pech et al. (1995) and Krebs et al. (2001) ascribed the possible manipulations of prey to test predator regulation, which include changes in prey density through reintroductions, altering food supply or the abundance of alternative prey species and predators. In the case of Glenelg Ark, a similar approach could be applied, including the reintroduction of species currently extinct or the translocation of species currently in low abundance, e.g., Long-nosed Potoroos. This approach would require careful design and implementation, strict monitoring and detailed analysis to ensure the outcome was properly interpreted and translated into management recommendations.

The Glenelg Ark monitoring program has achieved its aims of quantifying the response of foxes and targeted native species to broadscale continuous fox control through the rigorous implementation of the monitoring program. This report has highlighted several issues for further consideration. Consideration could also be given to how the monitoring of Glenelg Ark should continue into the future.

References

Agresti A. (2002) Categorical Data Analysis. Wiley Interscience, New York

Allen, L., Engerman, R. and Krupa, H. (1996). Evaluation of three relative abundance indices for assessing dingo populations. *Wildlife Research* **23**, 197–206

Burnham K. P. and Anderson D. R. (2010) Model selection and multi-model inference: A practical information-theoretic approach. Springer-Verlag, New York

Caughley, G. (1977) Analysis of vertebrate populations. John Wiley, London

Dexter, N. and Murray, A. (2008) The impact of fox control on the relative abundance of forest mammals in East Gippsland, Victoria. *Wildlife Research* **36**, 252–261

Efford, M. (1992) Comment – Revised estimates of the bias in the 'minimum number alive' estimator. *Canadian Journal of Zoology* **70**, 628–632

Fleming, P. (1997) Uptake of baits by red foxes (*Vulpes vulpes*): Implications for rabies contingency planning in Australia. *Wildlife Research* **24**, 335–346

Hosmer, D. W. and Lemeshow, S. (2000) Applied logistic regression. Wiley Interscience, New York

Jolly, G. M. and Dickson, J. M. (1983) The problem of unequal catchability in mark–recapture estimation of small mammal populations. *Canadian Journal of Zoology* **61**, 922–927

Krebs, C. J., Boutin, S. and Boonstra, R. (2001) *Ecosystem dynamics of the Boreal Forest: The Kluane Project.* Oxford University Press, New York

Lande, N., Ryan, L., Billard, L. and Farber, D. (1994) *Case studies in biometry*. John Wiley and Sons, New York

Lumsden, L. F., Scroggie, M. P., Chick, R., Grgat, L., Woodford, L., Bush, A., Alexander, J. and Robley, A. (2007) Assessing management strategies and wildfire impact on Long-footed Potoroos in north-eastern Victoria: 2nd progress report: Results from 2004/2005 and 2005/2006. Unpublished report to the Department of Sustainability and Environment. Arthur Rylah Institute for Environmental Research, Heidelberg

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

Lunney, D. (2001) Causes of the extinction of native mammals of the western division of New South Wales: an ecological interpretation of the nineteenth century historical records. *Rangelands Journal* **23(1)**, 44–70

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

MacKenzie, D. I., Nichols, J. D., Hines, J. E., Knutson, M. G. and Franklin, A. B. (2003) Estimating site occupancy, colonization, and local extinction when a species is detected imperfectly. *Ecology* **84**, 2200–2207

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

McLeod, L. J., Saunders, G. R. and Kabat, T. J. (2008) Do control interventions effectively reduce the impact of European Red Foxes on conservation values and agricultural production in Australia? Systematic Review No. 24. Collaboration for Environmental Evidence, www.environmentalevidence.org

Menkhorst, P. (1995) (Ed.) Mammals of Victoria, distribution, ecology and conservation. Oxford University Press, South Melbourne

Montgomery, W. I. (1987) The application of capture-mark-recapture methods to the enumeration of small mammal populations. *Symposium of the Zoological Society London* **58**, 25–57

Murray, A. (2005) A new low-cost hairtube design for the detection of the spotted-tailed quoll *Dasyurus maculatus* in south-eastern Australia. *Australian Mammalogy* **27**, 81–84

Nichols, J. D. and Pollock, K. H. (1983) Estimation methodology in contemporary small mammal capture-recapture studies. *Journal of Mammalogy* **64**, 253–260

Pech, R. P., Sinclair, A. R. E. and Newsome, A. E. (1995) Predation models for primary and secondary prey species. *Wildlife Research* **22**, 55–64

Pollock, K. H., Nichols, J. D., Brownie, C. and Hines, J. E. (1990) Statistical inference for capture-recapture experiments. *Wildlife Monographs* **107**, 1–97

Quinn, G. P. and Keough, M. J. (2002) *Experimental Design and Data Analysis for Biologists*. Cambridge University Press, Cambridge

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

Risbey, D. A., Calver, M. C., Short, J., Bradley, J. S. and Wright, I. W. (2000) The impact of cats and foxes on the small vertebrate fauna of Heirisson Prong, Western Australia. II. A field experiment. *Wildlife Research* **27**, 223–235

Saunders, G. and McLeod, L. (2007) *Improving fox managements strategies in Australia*. Australian Government Publishing Service, Canberra

Seber, G. A. F. (1986) A review of estimating animal abundance. *Biometrics* **42**, 267–292.

Thompson, J. A. and Fleming, P. (1994) Evaluation of the efficacy of 1080 poisoning of red foxes using visitation to non-toxic baits as an index of fox abundance. *Wildlife Research* **21**, 27–40

Tyre, A. J., Possingham, H. P. and Lindenmayer, D. B. (2001) Inferring process from pattern: can territory occupancy provide information about life history parameters. *Ecological Applications* **11**, 1722–1737

Zuur, A. F., Ieno E. N. and Smith G. M. (2007) Analysing Ecological Data. Springer, New York

Appendix 1. Ecological Vegetation Classes within each treatment and non-treatment area

Monitoring Area	Ecological Vegetation Class	Area (ha)	%
Mt Clay State Forest (treatment)	Heathy Woodland/Damp Heathy Woodland/Damp Heathland Mosaic	1597	35.6
	Herb-rich Foothill Forest	847	18.9
	Lowland Forest	1950	43.4
Hotspur State Forest (non-treatment)	Heathy Woodland	2235	32.9
	Lowland Forest	3097	45.6
	Wet Heathland	493	7.3
LGNP south (treatment)	Damp Sands Herb-rich Woodland	1319	14.9
	Damp Sands Herb-rich Woodland/Heathy Woodland Mosaic	2855	32.2
	Damp Sands Herb-rich Woodland/Heathy Woodland/Sand Heathland Mosaic	972	11.0
	Heathy Woodland/Limestone Woodland Mosaic	2827	31.9
LGNP north (non-treatment)	Damp Sands Herb-rich Woodland	2021	43.5
	Damp Sands Herb-rich Woodland/Heathy Woodland Mosaic	417	9.0
	Wet Heathland/Heathy Woodland Mosaic	2041	43.9
Cobboboonee National Park (treatment)	Lowland Forest	7557	79.9
	Wet Heathland/Heathy Woodland Mosaic	1035	11.0
Annya State Forest (non-treatment)	Damp Sands Herb-rich Woodland	1106	13.5
	Lowland Forest	5704	69.8

Appendix 2. Occupancy model specifications

Persistence for each TMA and NTMA at each time period *t* and each site *i* was modelled as:

$$\text{logit}\left(\phi_{i,t}\right) = \alpha_i + \varepsilon^{(\phi)}_{i,t},$$

Where $\varepsilon^{(\phi)}_{i,t}$ is a random effect term to account for additional unexplained variation in persistence of sites within each TMA and NTMA at each time period.

Similarly, TMA and NTMA time specific colonisation was modelled as:

$$\text{logit}\left(\gamma_{i,t}\right) = \beta_i + \varepsilon^{(\gamma)}_{i,t}$$

Combining these gives time and treatment specific occupancy probabilities,

$$\psi_{i,t+1} = \psi_{i,t}\phi_{i,t} + \left(1 - \psi_{i,t}\right)\gamma_{i,t}$$

The data were modelled in a state-space modelling framework, which separates the process and the observation components of the model. The process component models the process of occupancy from one year to the next. We can define $z_{i,t}$ as the true, yet unknown, state of each site at each year, where $z_{i,t} = 1$ indicates site *i* is occupied at time *t*, and 0 indicates unoccupied. The state at time *t* for each site within a monitoring area is modelled by:

$$z_{i,t} \sim Bern\left(\psi_{i,t}\right)$$

Therefore, site *i* is occupied at time *t* with the probability $\psi_{i,t}$.

The observation component of the model involves our observed detection history data Y, conditional on the true state *z* modelled previously, given by

$$y_{i,t,d} \sim Bern\left(z_{i,t}\, p_t\right)$$

where p_t is the daily detection probability for year *t*. The term $y_{i,t,d}$ indicates the observed occupancy at site *i*, year *t*, on day *d*. Therefore for truly occupied sites, the observation $y = 1$ is given by probability *p*.

The model was specified in WinBUGS 1.4 (Lunn et al. 2000) Random effects terms for persistence and colonisation were modelled by:

$$\varepsilon^{(\phi)}_{i,t} \sim Normal\left(0, \sigma_\phi\right) \quad \text{and} \quad \varepsilon^{(\gamma)}_{i,t} \sim Normal\left(0, \sigma_\gamma\right)$$

Prior probabilities were placed on all parameters, specifically Inverse Gamma (0.001, 0.001) for the random effects parameters σ_ϕ and σ_γ, Normal(0, 100) for the mean site-specific persistence and colonisation parameters α_{site} and β_{site}, and Uniform(0, 1) for the daily detection probabilities p_t. The Markov Chain Monte Carlo updater was run for 10 000 iterations to tune the algorithm, and ensure the chain had reached a stable distribution. These 'burn-in' samples were discarded and the algorithm run for a further 40 000 samples, which comprised the posterior distribution for each parameter.

Derived estimates of the overall treatment effect on persistence and colonisation for each pair was determined from the posterior distribution for the α_{site} and β_{site} parameters. For example, in the case of the Annya/Cobboboonee pair, the overall effect of treatment on persistence is given by

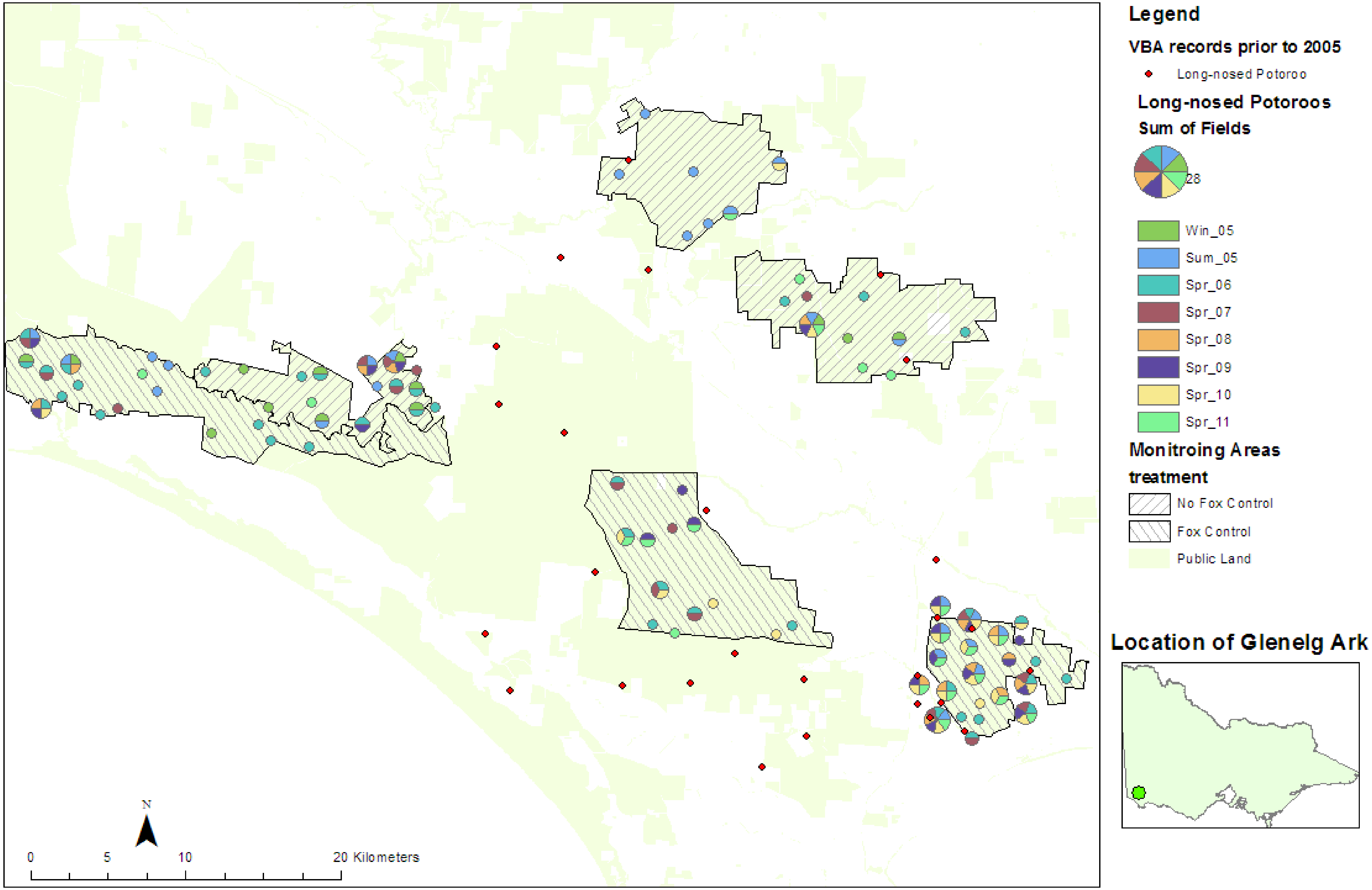
$$\alpha_{Cob-Ann} = \alpha_{Cob} - \alpha_{Ann}$$

and for colonisation is given by

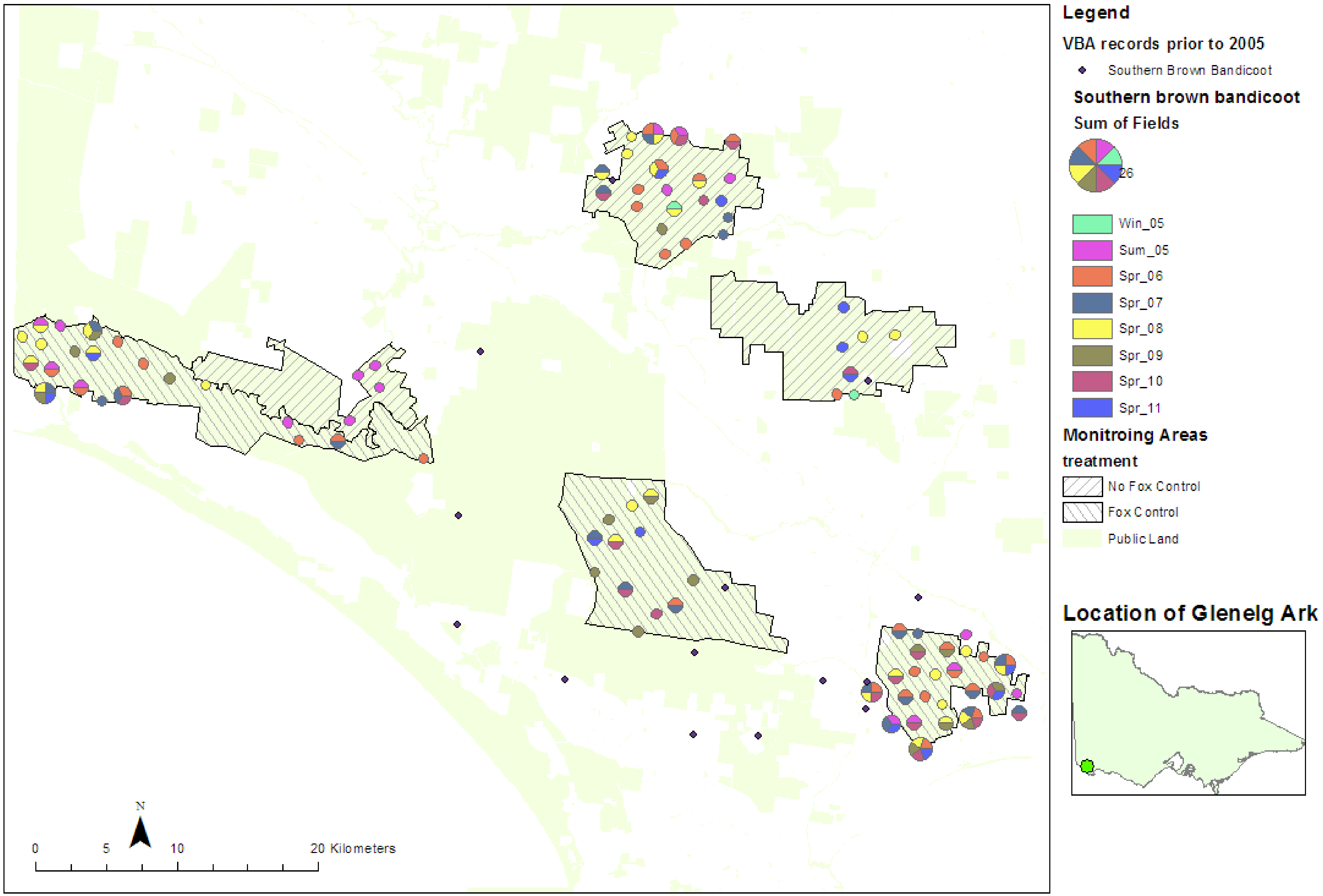
$$\beta_{Cob-Ann} = \beta_{Cob} - \beta_{Ann}$$

where values greater than zero indicate a positive effect of treatment. Treatment effects on other pairs are given in the same way. The proportion of the derived distribution greater than zero can be interpreted as the probability that the treatment effect has had a positive effect on persistence and colonisation respectively.

Appendix 3. Location and year of detection for Long-nosed Potoroos across the Glenelg Ark monitoring area



Appendix 4. Location and year of detection for Southern Brown Bandicoots across the Glenelg Ark monitoring area



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