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# Glenelg Ark 2005–2010: evidence of sustained control of foxes and benefits for native mammals

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# Glenelg Ark 2005–2010: 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 Weeds and Pests on Public Land Initiative, Department of Sustainability and Environment. The project aims to facilitate the recovery of native animal populations in far south-west Victoria by undertaking large-scale, continuous red fox (*Vulpes vulpes*) baiting using buried Foxoff® baits across 100 000 ha of state forest and national park.

The monitoring and evaluation of the program of Glenelg Ark measured the effect of the control operation on foxes as well as the response of a range of native species that are currently at risk from red fox predation. The monitoring program uses 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 number of foxes.

The Glenelg Ark monitoring program has achieved its aims of quantifying the response of foxes and targeted native species to broad scale continuous fox control. As the monitoring program receives further information, more certainty around the nature of these responses will be achieved.

Treatment Monitoring Areas (i.e. received fox control) and Non-Treatment Monitoring Areas (i.e. 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 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.

There was a significant and sustained decline in the index of abundance on the three treatment monitoring areas (1080 Foxoff baits) compared to the three non-treatment monitoring areas. However, the index of abundance (sand plot) did not conclusively indicate differences between treated and non-treated sites.

We investigated the effectiveness of supplementary fox control using Victor soft jaw leg-hold traps at Lower Glenelg National Park (LGNP) south. We also investigated the relative efficiency of live-trapping, remote cameras and hair tube surveys for Heath Mice (*Pseudomys shortridgei*) a species recorded rarely in hair tubes in the Glenelg Ark program. We also investigated the influence of digital camera orientation on the detection rates of small and medium-sized mammals.

No foxes were captured in LGNP south (area baited for foxes), and eight (1.98 captures/100 trap nights) were captured in LGNP north (area not baited for foxes). Twenty-nine feral cats were captured at LGNP south (7.18 captures/100 trap nights) and 10 feral cats were captured at LGNP north (2.47 captures/100 trap nights).

Overall, Reconyx cameras had higher detection rates than ScoutGuard cameras, and downward facing cameras generally had higher detection rates than outward facing cameras.

There was a significant positive effect of treatment on persistence and colonisation for Southern Brown Bandicoots and Long-nosed Potoroos each at a single but different site. There was a positive effect 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.

The results of the monitoring and evaluation program clearly demonstrate that foxes have been reduced and remain at relatively lower levels, and that the use of occupancy modelling as applied in this project is capable of measuring changes in native species.

Future efforts will be aimed at determining the relative effectiveness of alternative methods for assessing differences in fox abundance between treated and non-treated sites, and enhancing the

analysis by examining the influence of covariates such as rainfall, fire history and vegetation community on rates of persistence and colonisation.

# 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 *Environmental Protection and Biodiversity Conservation Act 1999* (Commonwealth). 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 impacted 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. The simplest approach to estimating occupancy is to derive a naïve 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 in an unequivocal manner. 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, monitoring and evaluation is a part of management and 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. received fox control) and Non-Treatment Monitoring Areas (NTMA, i.e. did not receive fox control) (Figure 1) are used to assess the effectiveness of the program. 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. Cobboboonee 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).

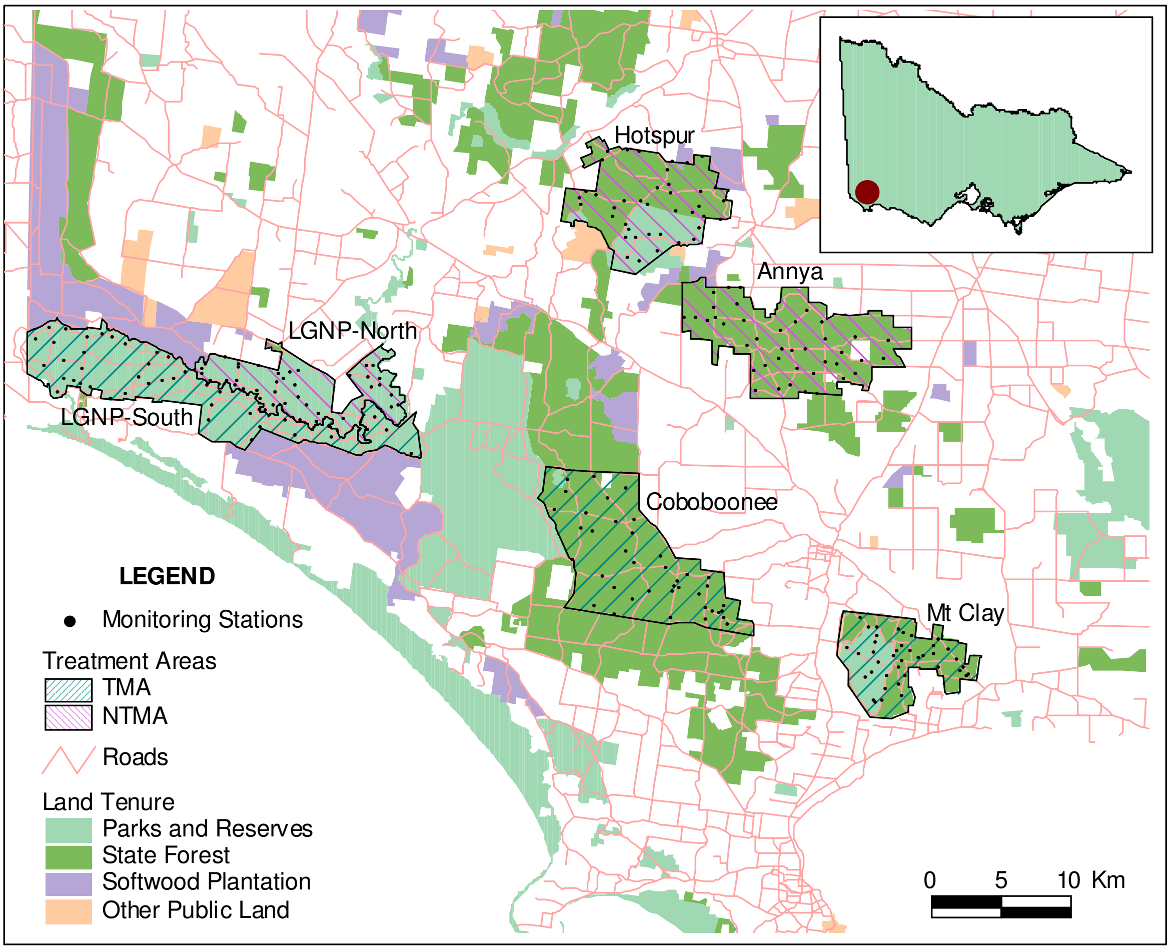

**Figure 1. Glenelg Ark monitoring and evaluation areas.**

## 2.2 Measuring changes in foxes

In the TMAs and NTMAs there had been little previous fox control prior to 2005. Each TMA bait stations 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 what 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 (see 2.2.2) 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.

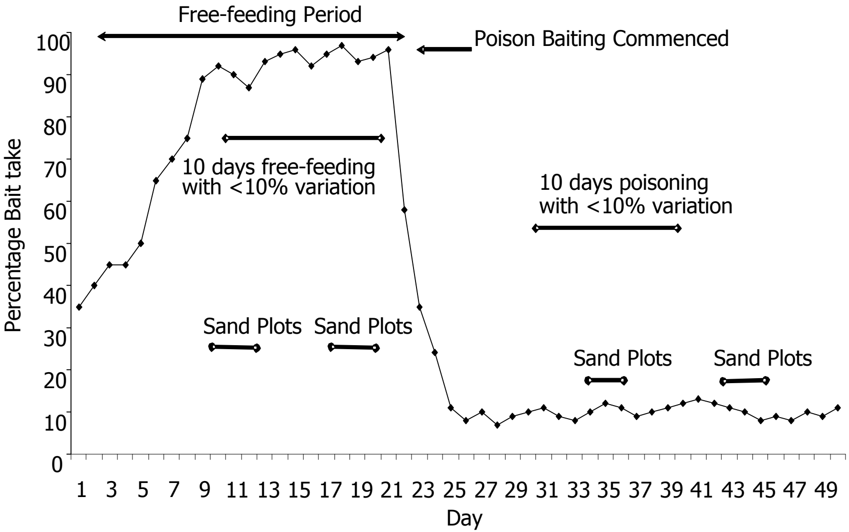

**Figure 2. Steps in monitoring the effectiveness of the initial fox reduction baiting, showing an example of data that might be obtained.**

### *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. Baits become unavailable to other foxes once bait has been removed bait; therefore, 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).

*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 treated sites was calculated as follows:

% Change = (pre-baiting index – post-baiting index) / pre-baiting index*100

### 2.2.2 Sustained reduction in the fox population

Once the initial knockdown of foxes had been established, continued monitoring of bait take and sand plot activity at two weekly intervals on TMAs and NTMAs provided evidence of any 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.

### 2.2.3 Supplementary fox control

Poison baiting for the control of foxes is practised widely for the protection of biodiversity and agricultural values. However, it is recognised that poison baiting alone cannot remove all foxes from a particular area (Saunders and McLeod 2007). To supplement the poison baiting program, leg-hold trapping was undertaken at Lower Glenelg National Park – south and north – for four weeks in spring 2010 using 20 Victor 1.5 soft jawed traps (Western Trapping Supplies, Qld). The details of any non-target feral cats (*Felix catus*) captured were also recorded. This program was instigated by DSE South-west Region and we report on the outcome here.

## 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 (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 Kendal 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 ecological vegetation class (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) with extra sampling in winter 2005 and 2006, and 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 ($\gamma_t$) 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 ($\phi_t$) is analogous to survival and is the probability that sites that were occupied in $t_1$ remain occupied at $t_2$. Extinction $\varepsilon_t$ 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 ($\psi$) in subsequent years results from the number of currently occupied sites multiplied by the probability of persistence ($\psi_t \phi_t$), and the number of currently unoccupied sites ($1 - \psi_t$) multiplied by the probability of colonisation ($\gamma_t$). 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 $\phi$, and unoccupied sites will be colonised with probability $\gamma$.

It was considered likely that the probability of detection using hair-tubing (as opposed to the probability of occupation) might have differed between years because of seasonal effects on foraging behaviour. Lumsden et al. (2007) previously documented significant levels of inter-annual variation in the detection probabilities of Long-footed Potoroos (*Potorous longipes*) 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.

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 (57 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 ($\gamma_t$) 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 as there are no sites available to persist.

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.

## 2.4 Assessing detection rates for Heath Mice using different survey techniques

Analysis of the hair tube data from 2005 to 2010 indicated that the detection rate for Heath Mice (*Pseudomys shortridgei*) is too low to allow meaningful conclusions to be drawn from the data, thus additional monitoring approaches are required to supplement the hair tube monitoring to enable the effectiveness of the fox control program to be measured. We compared the detection achieved by hair tube, remote camera and live-trapping surveys at Mt Clay and Hotspur sites.

*Extra hair-tube surveys*

Fifteen hair-tube monitoring stations from both Mt Clay and Hotspur were selected from the existing set of 40 monitoring stations based on either previous records of Heath Mice or likely habitat based on EVC. Hair-tube grids were run for 21 days in October-November 2010. Standard hair tubes comprise an inverted ‘U’ shaped wire bracket that holds plastic tube in place. This tube is normally 10–12 cm wide to allow medium-sized animals, e.g. Southern Brown Bandicoots, to enter. In the Heath Mice surveys, the brackets were closed to 3–5 cm to capture hair from small mammals. Tapes were collected and sent to Barbara Triggs (Genoa, Victoria) for analysis.

*Remote camera surveys*

Two Pix-controller DigitalEye cameras (Pix-controller Inc. PA, USA) were placed at each of the 15 monitoring sites used in the hair-tube surveys. Cameras were spaced at 100 m intervals, centred on the hair tube monitoring grid (i.e. 50 m either side of the grid). Cameras were set 1.5 m from a lure station and 30 cm from the ground facing the lure station. The lure station used a wire cage with a tea infuser suspended inside. Inside the tea infuser was an oil absorbent cloth soaked in linseed oil and pistachio essence. Cameras were run for 21 days concurrently with the hair tube monitoring.

*Elliott trapping*

Live trapping was conducted after the camera and hair-tube surveys. This was to avoid the possibility of altering individual animals’ behaviour that may increase or decrease their detection rates.

Two trap lines of 10 Elliott traps were deployed for up to seven nights at each of the 30 monitoring sites. Each monitoring area was trapped once. Traps were checked between 6 am and 11.30 am each morning. The traps were then closed, and re-set in the afternoon, so that no trap was open for greater than 18 hours. Traps had nesting material (sea-grass) provided. The Elliott traps were wrapped in plastic bags to prevent water entering the traps and animals becoming cold overnight. All new animals were measured and weighed at the site of capture. A patch of fur (no greater than 5 mm x 5 mm) was clipped from the rump of the animal using nail scissors to mark them as having been captured.

Species trapped other than Heath Mice were identified, and recorded as having been caught, before being released into suitable cover at the site of capture.

*Analysis*

To assess differences in detection probability for Heath Mice across all sites we used a state-space occupancy modelling approach (MacKenzie et al. 2003). Detection probability was allowed to vary between survey method (remote camera, hair tube and Elliott trap). Detection was held constant across treatment sites, i.e., with and without fox control. Occupancy was allowed to vary by treatment.

The probability of daily detection (assuming occupancy) was modelled on the logit scale. Dummy variables were used for categorical variables, where for survey method a 1 denoted hair tubes a 2 denoted remote cameras and a 3 denoted Elliott traps. For each site i, at time t:

logit(p[survey[i]]) = alpha + beta.survey*survey*survey[i] + beta.survey*survey*survey[t]

Models were specified in a Bayesian framework using OpenBUGS (3.1.1). Vague normal priors (N(mean=0, SD=100)) were specified for all intercept and coefficient parameters. Uniform priors were specified for occupancy on the treatment and non-treatment sites.

Chains were run for 50 000 iterations after discarding an initial burn-in of 10 000 iterations.

## 2.5 Assessing detection probability of native mammal species using different cameras and camera orientation

The ability of cameras to detect species is critical to the precise estimate of changes in occupancy. Camera orientation has been investigated by Melbourne University (J. Smith, pers. comm.). Initial investigations using Pix-controller cameras suggested that downward facing cameras had a higher probability of detecting a range of species (J. Smith, pers. comm.). To investigate the influence of camera orientation and camera model used in the Glenelg Ark monitoring program, we tested detection rates of both Reconyx RapidFire ProPC90 heat-in-motion activated digital cameras (Reconyx, LLP Wisconsin, USA) and ScoutGuard (SG550V; HCO, Norcross, Georgia, USA) cameras facing outward (current practice) and facing downwards.

Seven sites from each of the six monitoring areas were randomly selected, totalling 42 sites. At each site two cameras of the same type were established, one as per current standard procedures, i.e., 2 m from a lure station and 0.3 m from the ground facing the lure station. The second camera was positioned 1.5 m directly above the lure cage facing downwards (Figure 3). All cameras were set to take three images per trigger and were operated for 24 days. Images were time and date stamped allowing for estimates of detection to be calculated.

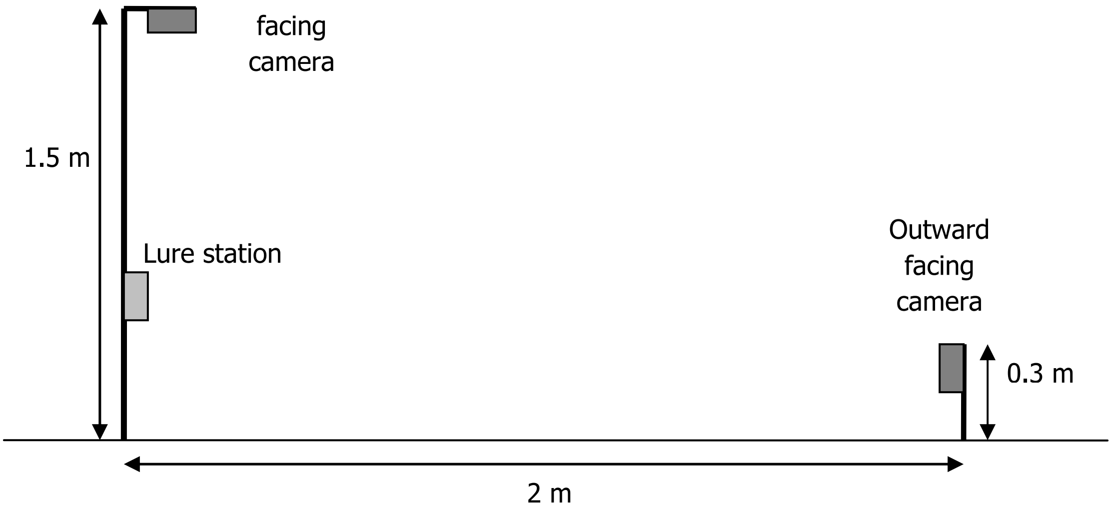

**Figure 3. Diagram showing layout of camera orientation study design.**

To assess differences in detection probability for species across all sites we used a state-space occupancy modelling approach (MacKenzie et al. 2003). Detection probability was allowed to vary between camera type (ScoutGuard and Reconyx) and camera orientation (downwards or outwards), as well as an interaction between camera type and camera orientation. Detection was held constant across treatment sites, i.e., with and without fox control. Occupancy was allowed to vary by treatment.

The probability of daily detection was modelled on the logit scale. Dummy variables were used for categorical variables, where for camera type a 0 denoted Reconyx and a 1 denoted ScoutGuard. Similarly, for camera orientation, a 0 denoted “Out” and a 1 denoted “Down”. For each site i, at time t:

> logit(p[camera[i],method[t]]) = alpha + beta.camera*camera[i] + beta.method*method[t] + beta.interaction*method[t]*camera[i]

Models were specified in a Bayesian framework using OpenBUGS (3.1.1). Vague normal priors (N(mean=0, SD=100)) were specified for all intercept and coefficient parameters. Uniform priors were specified for occupancy on the treatment and non-treatment sites.

Chains were run for 50 000 iterations after discarding an initial burn-in of 10 000 iterations.

# 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).

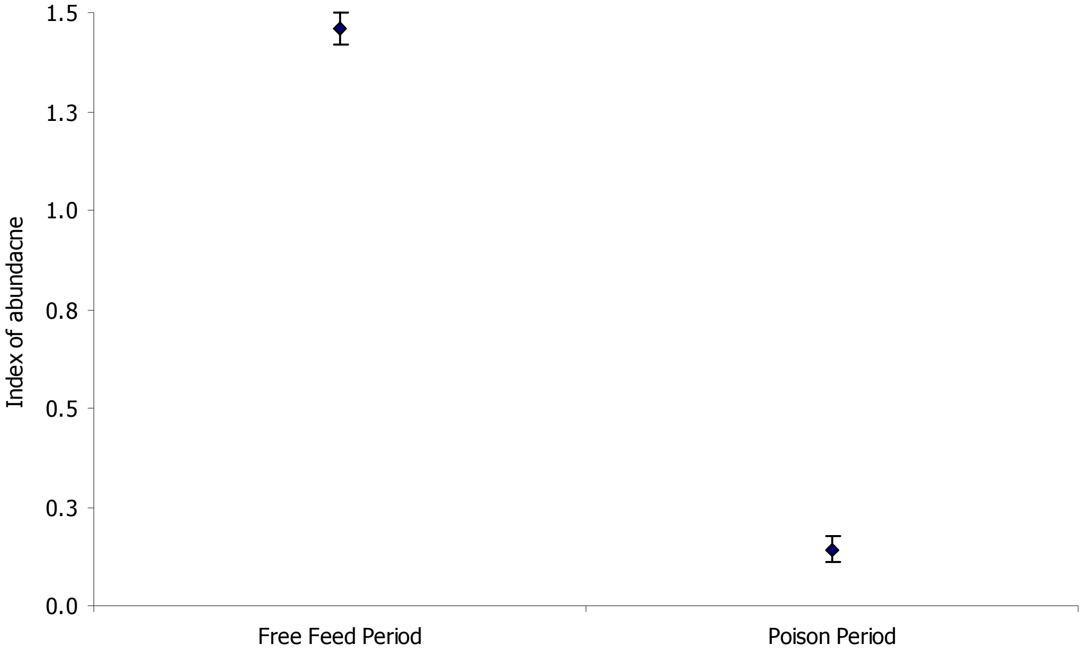

**Figure 4. Index of abundance for the 3 month free-feed period (July 2005–October 2005) and the 3 month poison period (October 2005–December 2005) at Lower Glenelg National Park south.** Bars are 95% binomial confidence intervals.

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 2010 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.

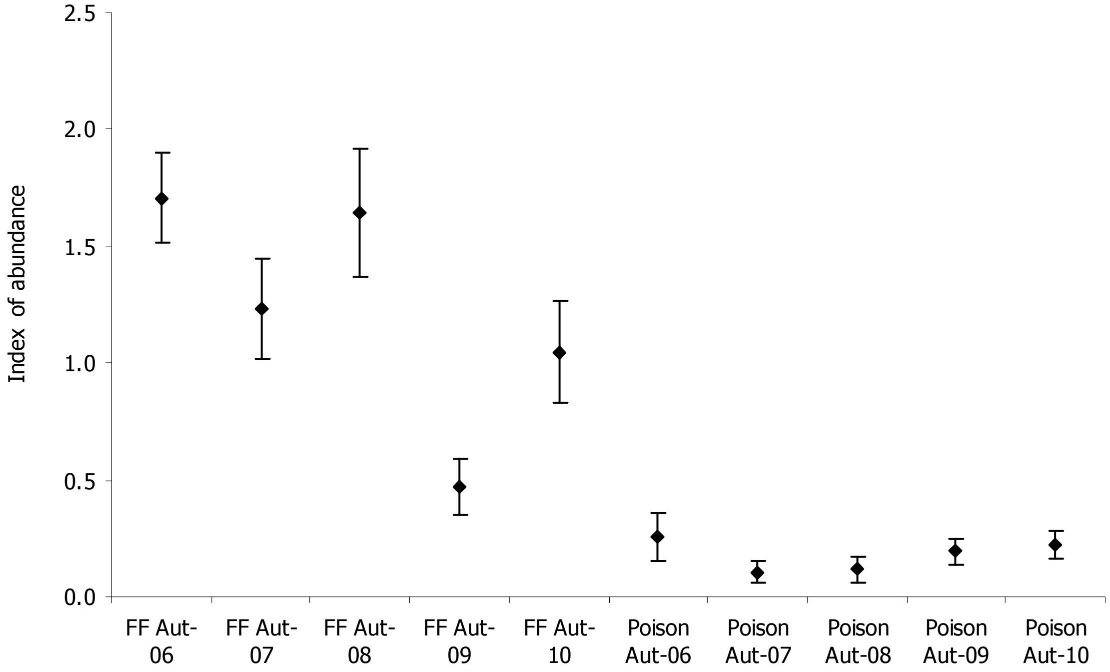

**Figure 5. Index of abundance during the poisoned and free-feed period on Lower Glenelg National Park.** Free-feeding occurred in autumn each year at LGNP north to compare with baited site in the south. Bars are 95% binomial confidence intervals.

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

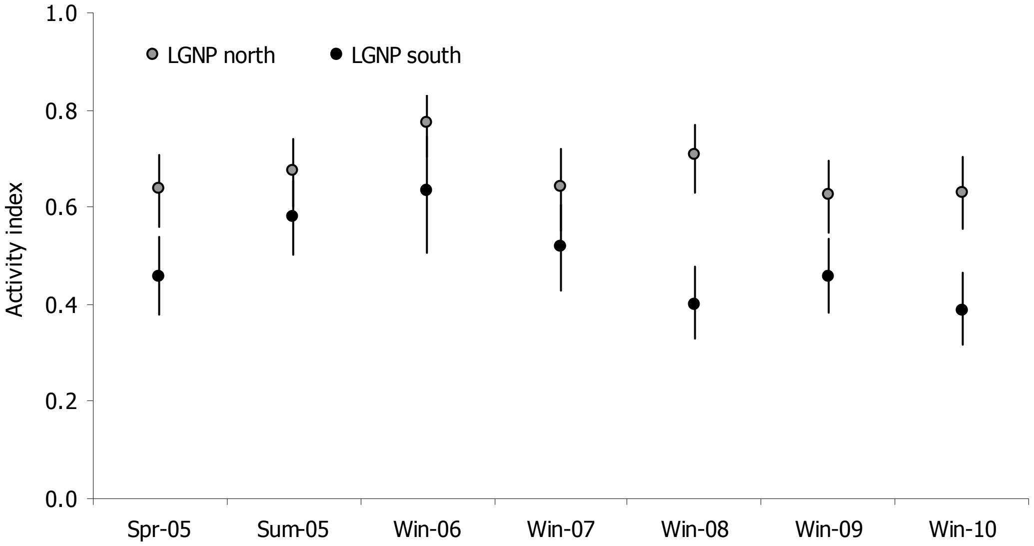

**Figure 6. Fox activity index for Lower Glenelg National Park south and north.** Fox baiting commenced in December 2005. Bars are 95% credible limits.

### 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 7).

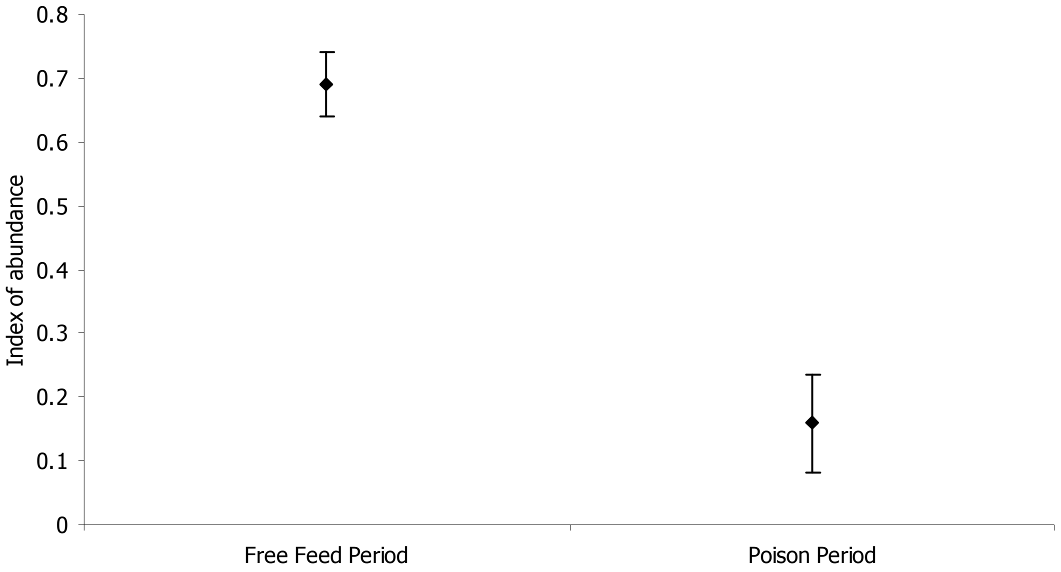

**Figure 7. Index of abundance during the free-feed period (July 2005–October 2005) and poison period (October 2005–December 2008) at Cobboboonee National Park.** Bars are 95% binomial confidence intervals.

The index of abundance was lower in Cobboboonee National Park in all years (2006–2010) compared to the same period in Annya State Forest (Figure 8).

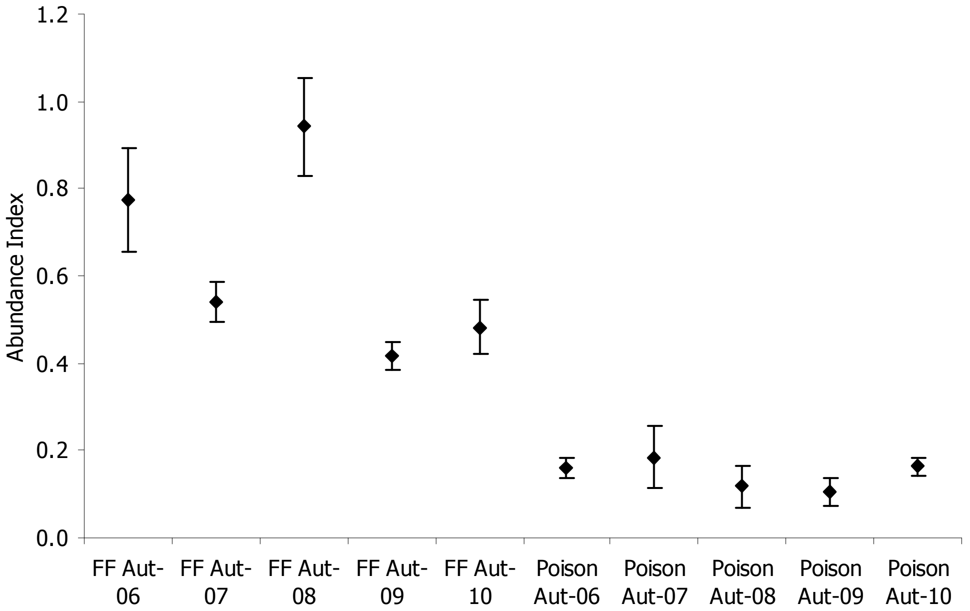

**Figure 8. Index of abundance during poisoned and free feed baiting at Cobboboonee National Park and Annya State Forest.** Free-feeding occurred in autumn each year at Annya State Forest. Bars are 95% binomial confidence intervals.

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 9).

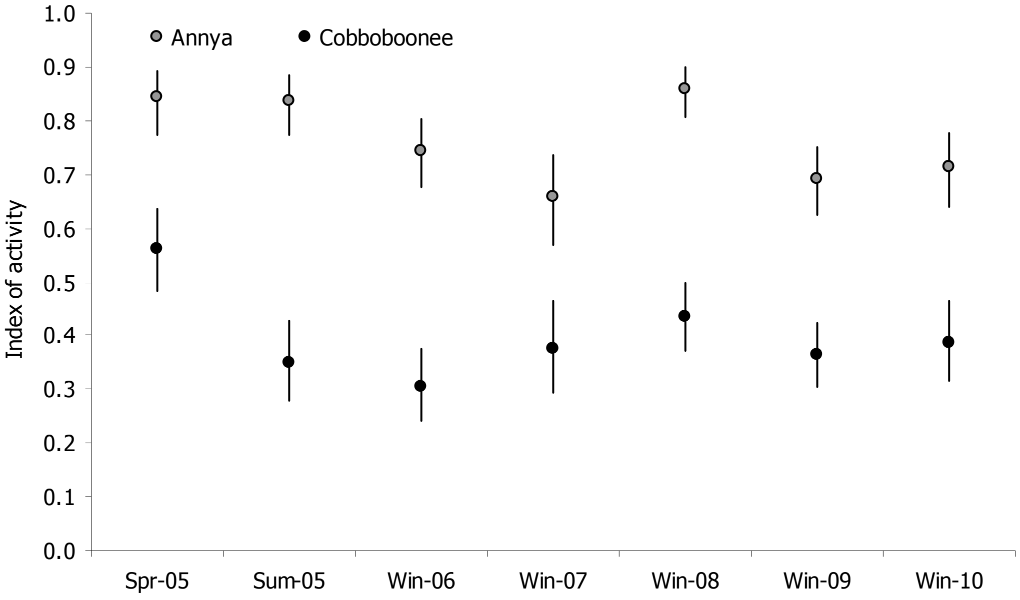

**Figure 9. Fox activity index for Annya State Forest and Cobboboonee National Park.** Fox baiting commenced in December 2005. Bars are 95% credible limits.

### 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 10).

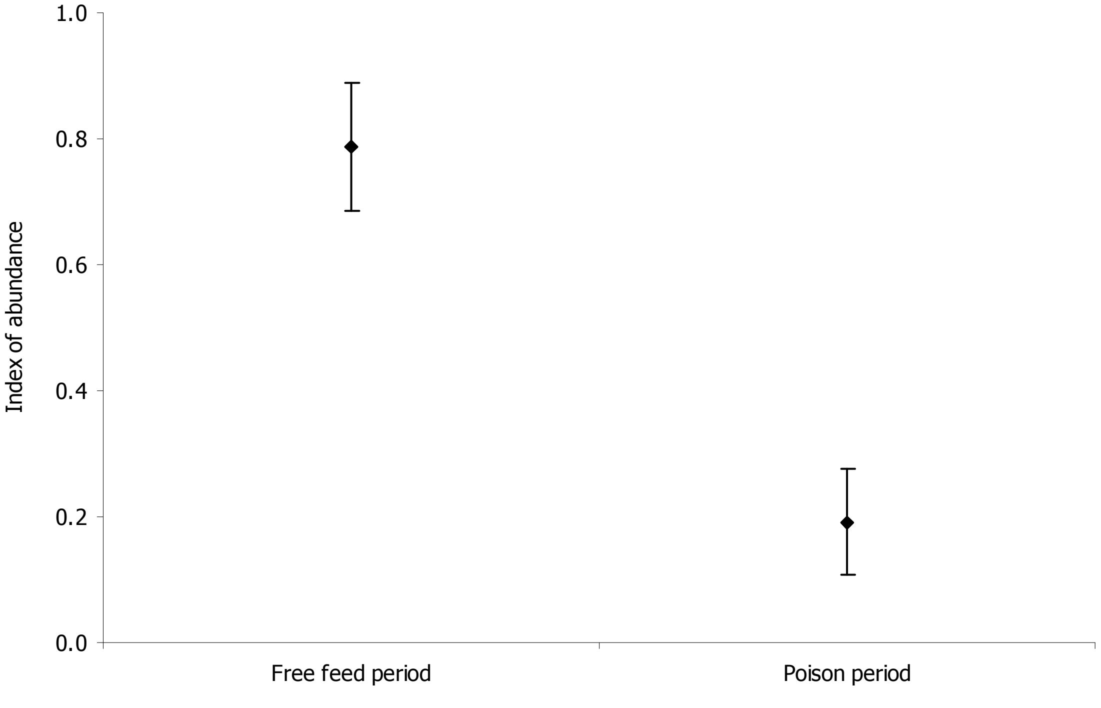

**Figure 10. Difference in the proportion bait taken for the free-feed period and poison period at Mount Clay State Forest.** Bars are 95% binomial confidence intervals.

The index of abundance was less on the TMA compared to the NTMA (Figure 11). A decision was made not to continue to collect free-feed data from this monitoring pair because of limited availability of resources, but free-feed will be undertaken in 2011 on the NTMA.

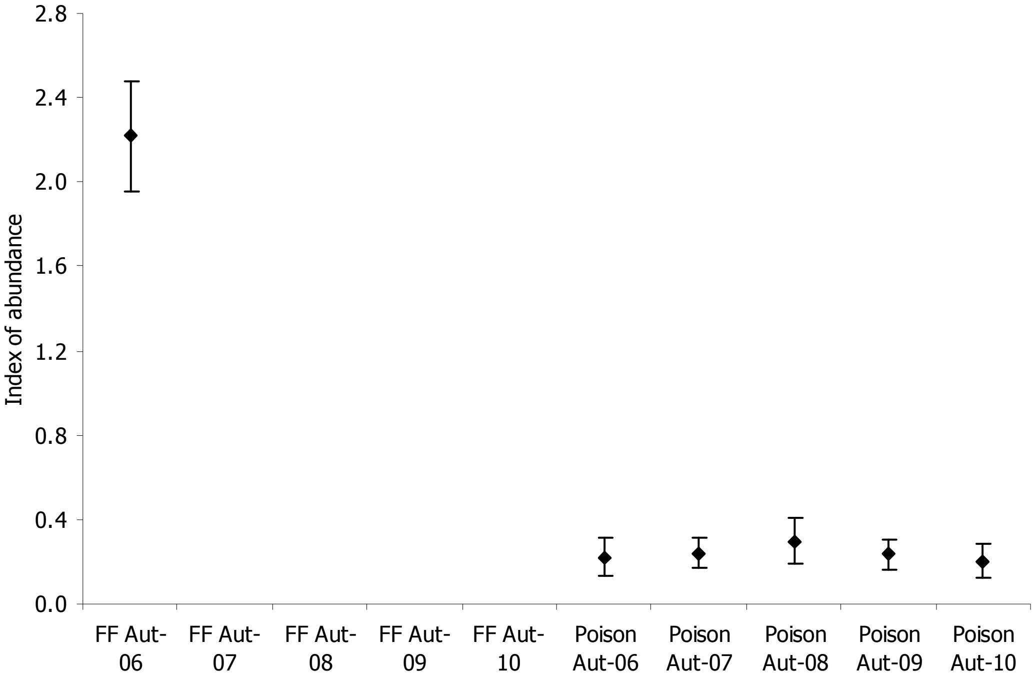

**Figure 11. Index of abundance between free-feed baiting at Hotspur State Forest and poisoning at Mount Clay State Forest.** Bars are 95% binomial confidence intervals.

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.

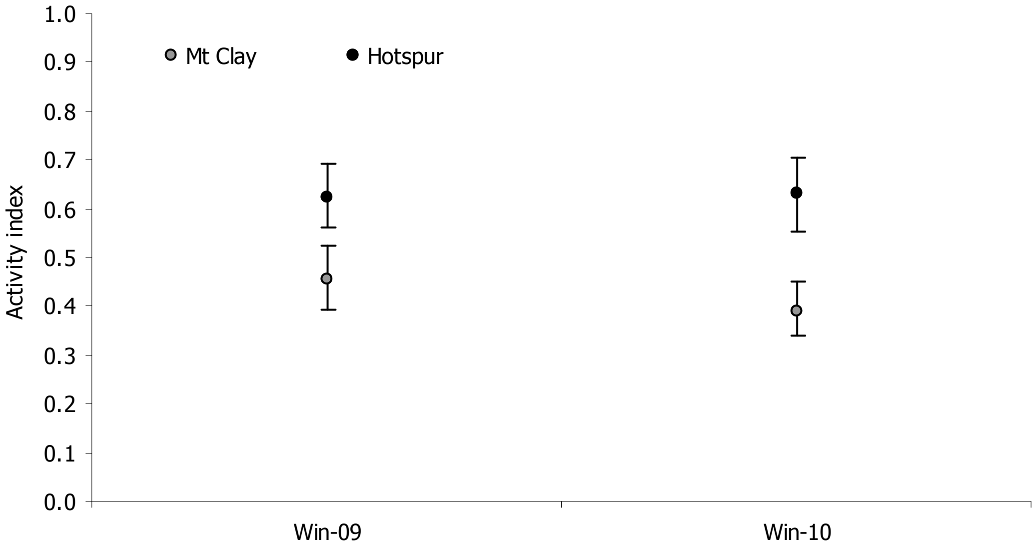

**Figure 12. Fox activity index for Mt Clay and Hotspur State Forest.** Fox baiting commenced in December 2005. Bars are 95% confidence intervals.

## 3.2 Supplementary fox control

Trapping for foxes using Victor soft jaw leg-hold traps 1.5was undertaken in LGNP south in August 2010. Trapping was conducted over four weeks and involved 404 trap nights.

Trapping was also undertaken in LGNP north in September 2010 using Victor soft jaw leg-hold traps. Trapping was conducted over four weeks and involved 404 trap nights.

No foxes were captured in LGNP south and eight (1.98 captures/100 trap nights) were captured in LGNP north (Table 1).

Feral cats were also captured during this exercise. Twenty-nine feral cats were captured at LGNP south (7.2 captures/100 trap nights) and 10 feral cats were captured at LGNP north (2.5 captures/100 trap nights). Of the cat captures in LGNP south, six were near the Nelson Tip which is located inside the LGNP south fox treatment area. Removing these captures reduces the capture rate to 5.7 captures/100 trap nights (Table 1).

**Table 1. Results of the supplementary fox trapping (captures/100 trap nights) at LGNP south and north.**

| Location | Week | Fox captures | Feral Cat captures | Trap nights | Fox (captures/100 trap nights) | Feral Cat (captures/100 trap nights) |
|---|---|---|---|---|---|---|
| **LGNP south** | | | | | | |
| | 1 | 0 | 12 | 96 | 0 | 12.5 |
| | 2 | 0 | 8 | 112 | 0 | 7.1 |
| | 3 | 0 | 5 | 100 | 0 | 5.0 |
| | 4 | 0 | 4 | 96 | 0 | 4.2 |
| **Total** | | **0** | **29** | **404** | **0** | **7.2** |
| **LGNP north** | | | | | | |
| | 1 | 1 | 3 | 120 | 0.8 | 2.5 |
| | 2 | 0 | 5 | 96 | 0.0 | 5.2 |
| | 3 | 3 | 0 | 92 | 3.3 | 0.0 |
| | 4 | 4 | 2 | 96 | 4.2 | 2.1 |
| **Total** | | **8** | **10** | **404** | **2.0** | **2.5** |

# 4 Monitoring changes in native species

In each of the sampling years the main species of interest (Southern Brown Bandicoot, Long-nosed Potoroo and Common Brushtail Possum) were detected. Figures in Appendix 3 and 4 show the monitoring station and year of detection for Southern Brown Bandicoots and Long-nosed Potoroos 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 Hotspur/Mt Clay (Figures 13, 14; Table 2).

The probability of a positive effect of treatment on persistence and colonisation was significant for potoroos at the Hotspur/Mt Clay pair. There is weak evidence of a trend in a positive effect for colonisation at Annya/Cobboboonee, while there is no indication of a treatment effect at LGNP south and north.

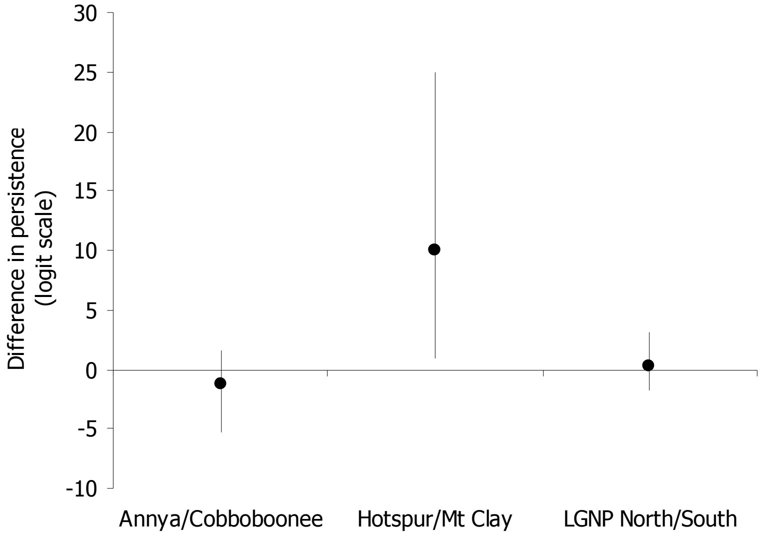

**Figure 13. Effect of treatment on persistence on Long-nosed Potoroos at each non-treatment/treatment pair.** Bars are 95% credible intervals.

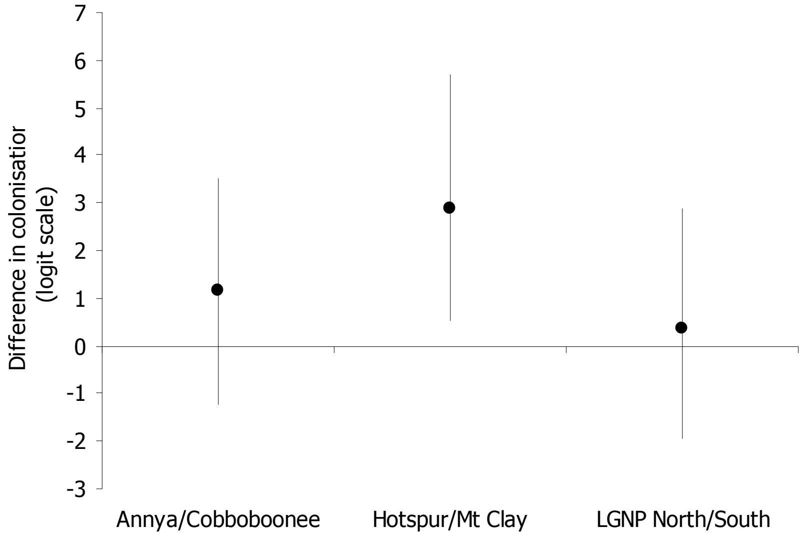

**Figure 14. Effect of treatment on colonisation on Long-nosed Potoroos at each non-treatment/treatment pair.** Bars are 95% credible intervals.

**Table 2. 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 | 0.21 | 0.99 | 0.60 |
| Colonisation | 0.84 | 0.99 | 0.62 |

These results indicate that at Mt Clay, Long-nosed Potoroos are slowly occupying new sites, and that the chance 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 (Figure 15 and 16).

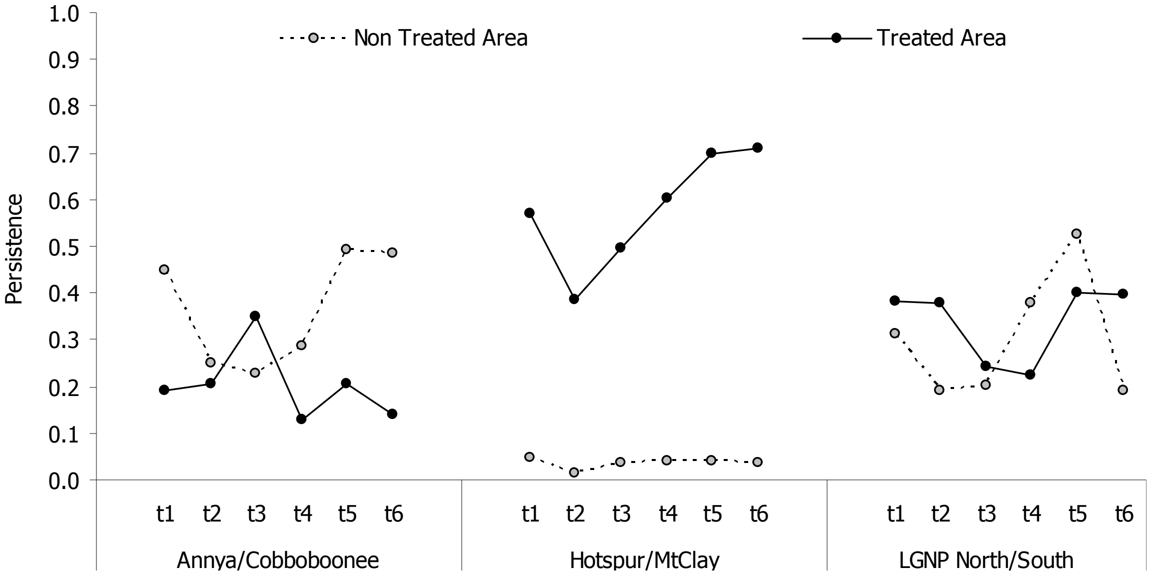

**Figure 15. Derived estimates of mean persistence for Long-nosed Potoroo at each non-treatment/treatment pair.**

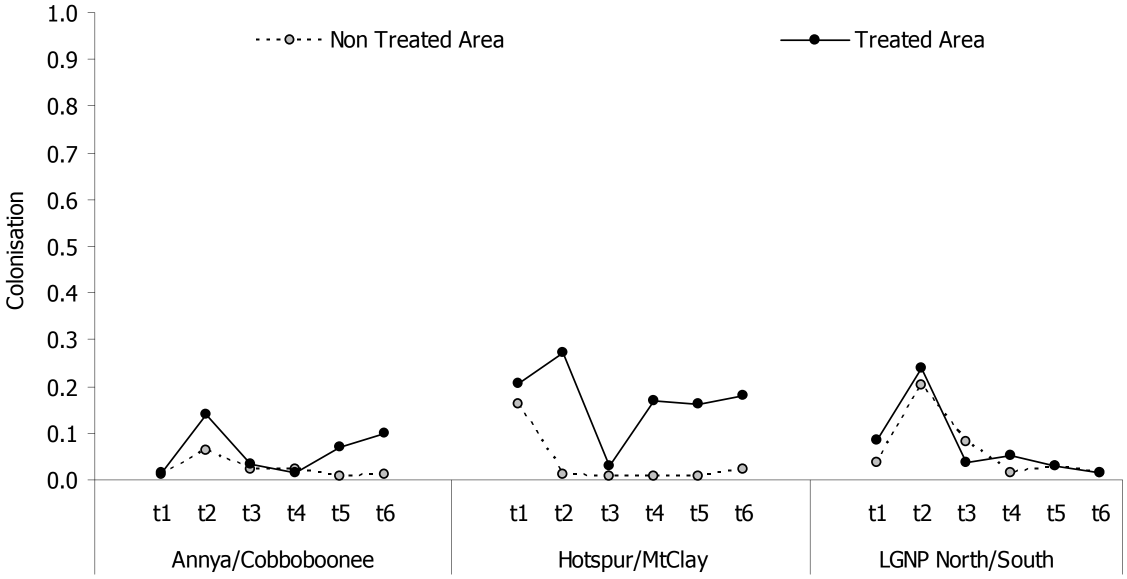

**Figure 16. Derived estimates of mean colonisation for Long-nosed Potoroo at each non-treatment/treatment pair.**

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 17). 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.

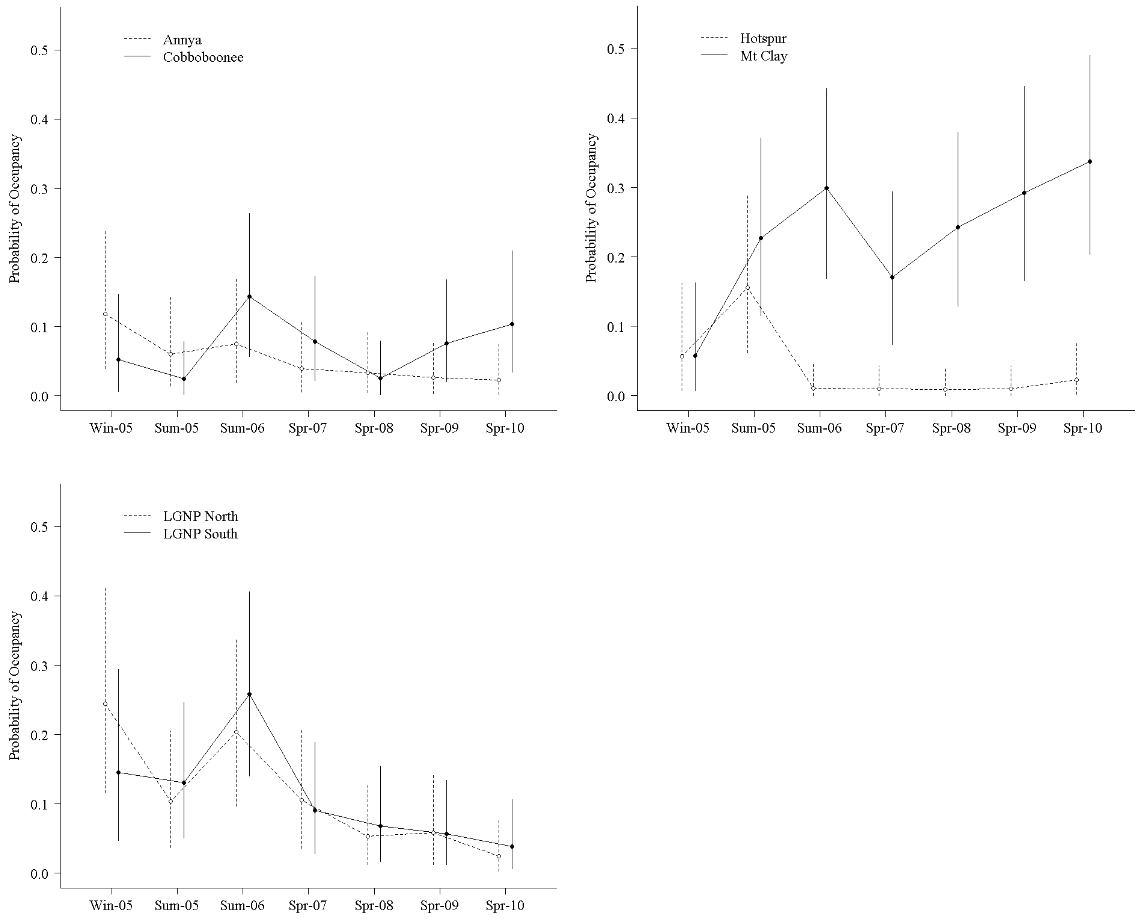

**Figure 17. Derived estimates of occupancy for Long-nosed Potoroo at each non-treatment/treatment pair.** Bars are 95% credible intervals.

## 4.2 Southern Brown Bandicoot

The effect of fox control on persistence and colonisation from year to year was significant at the Annya/Cobboboonee pair. There was a significant effect on persistence at the Mt Clay/Hotspur pair and on colonisation at the Lower Glenelg National Park pair (Figures 18, 19; Table 3).

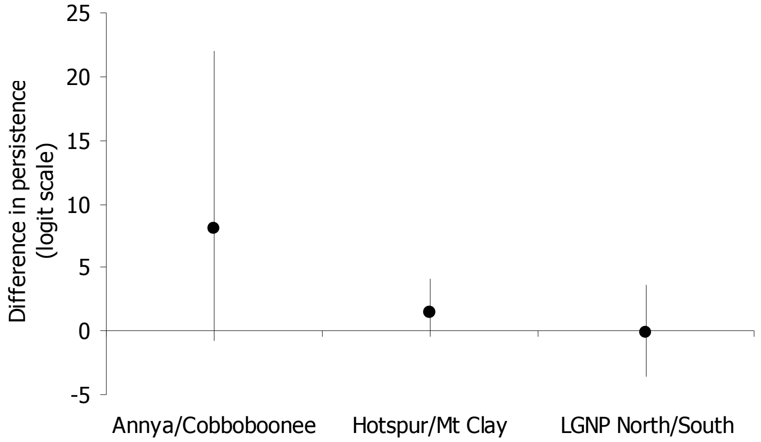

**Figure 18. Effect of treatment on persistence for Southern Brown Bandicoot at each non-treatment/treatment pair.** Bars are 95% credible intervals.

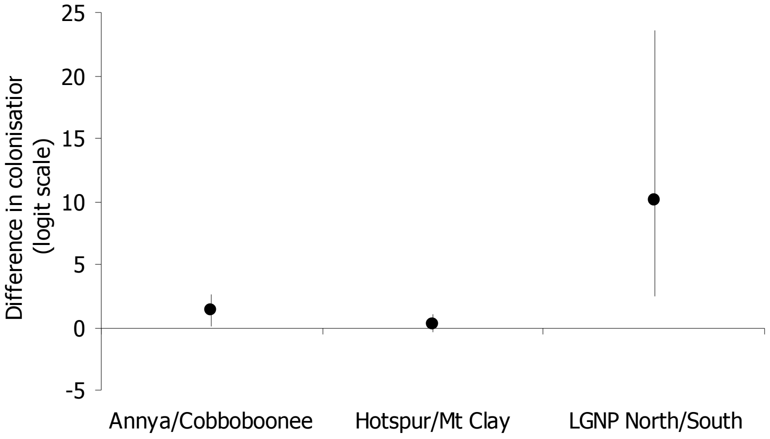

**Figure 19. Effect of treatment on colonisation for Southern Brown Bandicoot at each non-treatment/treatment pair.** Bars are 95% credible intervals.

**Table 3. 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 | 0.95 | 0.94 | 0.30 |
| Colonisation | 0.98 | 0.80 | 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 20, 21). 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.

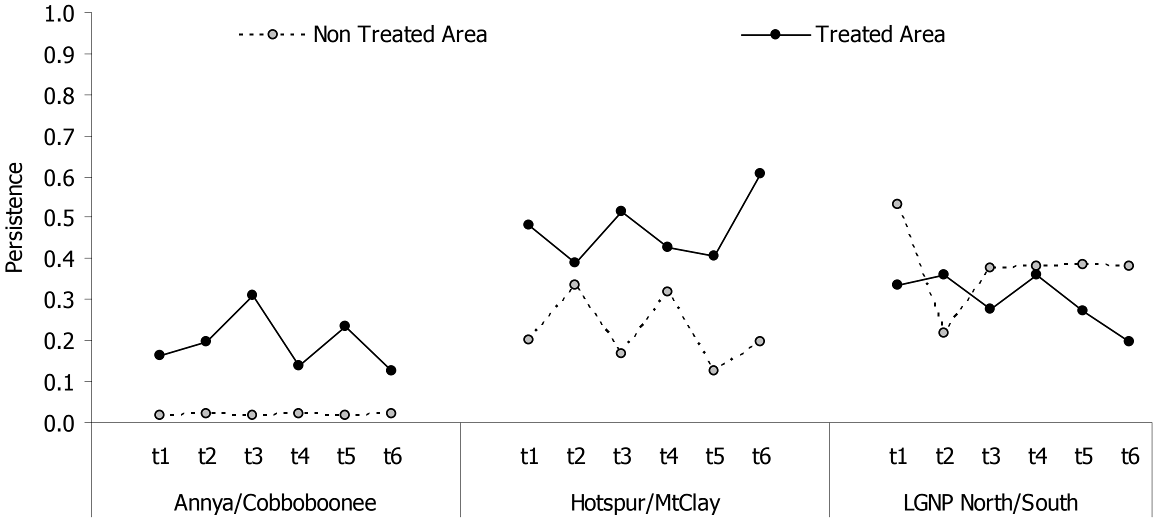

**Figure 20. Derived estimates of mean persistence for Southern Brown Bandicoot at each non-treatment/treatment pair.**

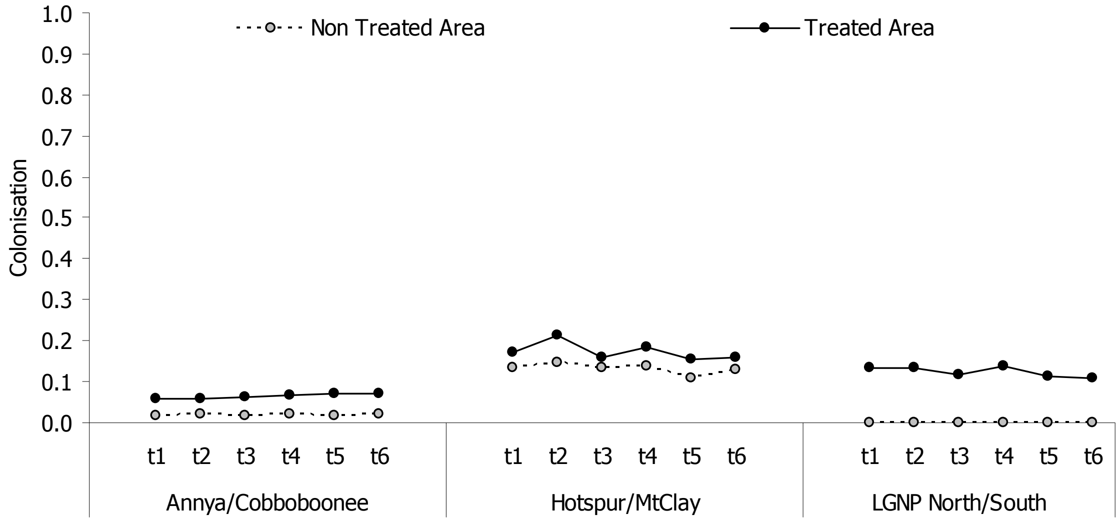

**Figure 21. Derived estimates of mean colonisation for Southern Brown Bandicoot at each non-treatment/treatment pair.**

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 22). 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 treated 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 untreated area (Annya). For the LGNP areas, the probability of colonisation at the treated 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 are uniform between 0 and 1 (i.e. equivalent to the prior).

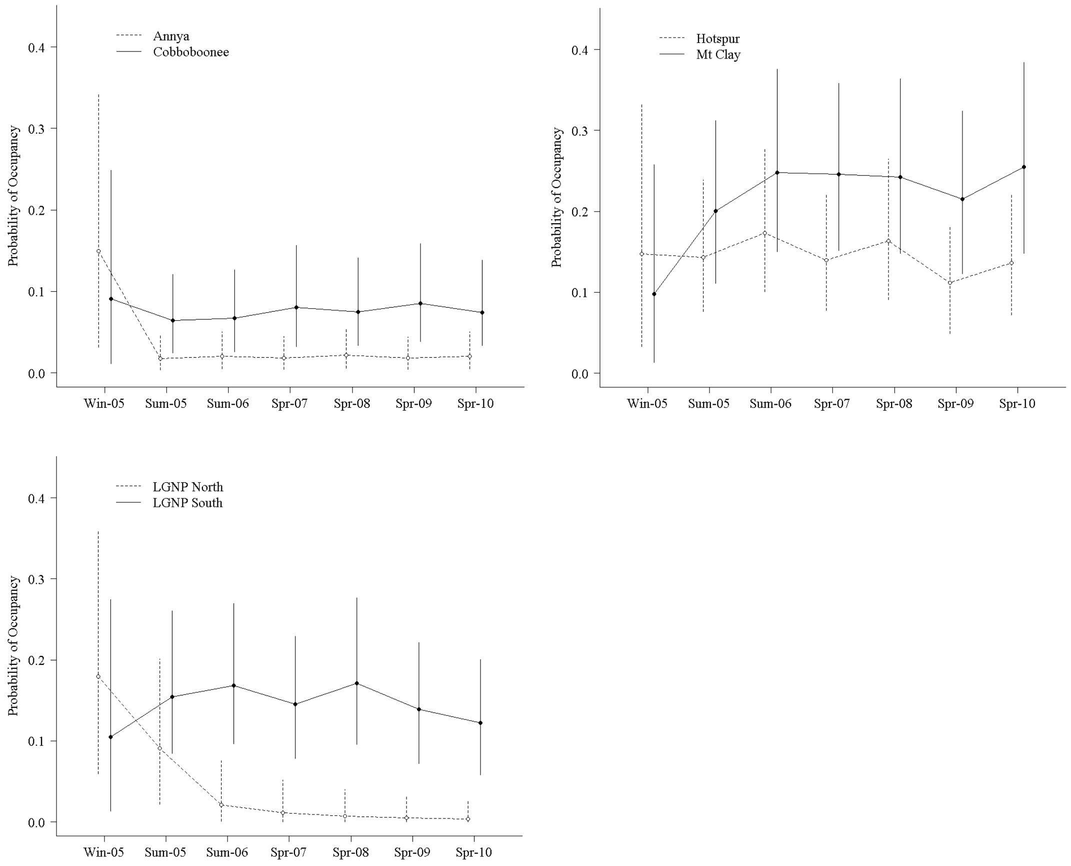

**Figure 22. Derived estimates of occupancy for Southern Brown Bandicoot at each non-treatment/treatment pair.** Bars are 95% credible intervals.

## 4.3 Common Brushtail Possum

There is strong evidence of a significant treatment effect on persistence (Table 4) and colonisation (Table 4) at the LGNP north and south pair of sites, and for colonisation at the Annya/Cobboboonee pair and no differences at the other site pairs (Figure 23). The treatment effect on colonisation is strongly negative at the Hotspur/Mt Clay pair (Figure 24; Table 4). These results are apparent in derived estimates of persistence and colonisation (Figures 25, 26). 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.

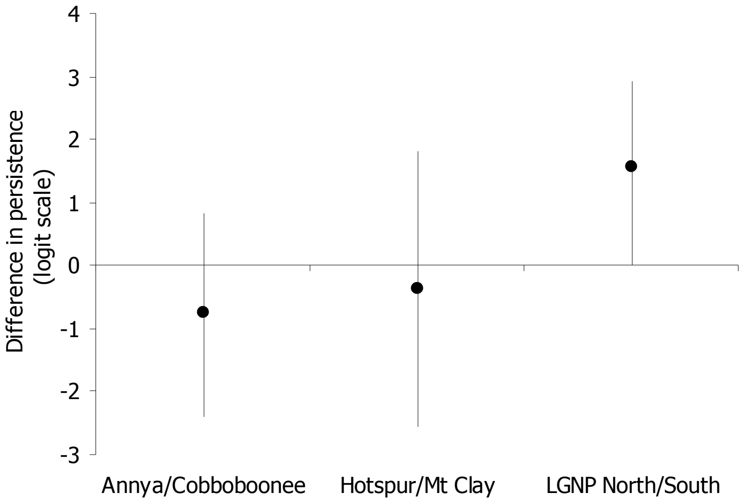

**Figure 23. Effect of treatment on persistence for Common Brushtail Possum at each non-treatment/treatment pair.** Bars are 95% credible intervals.

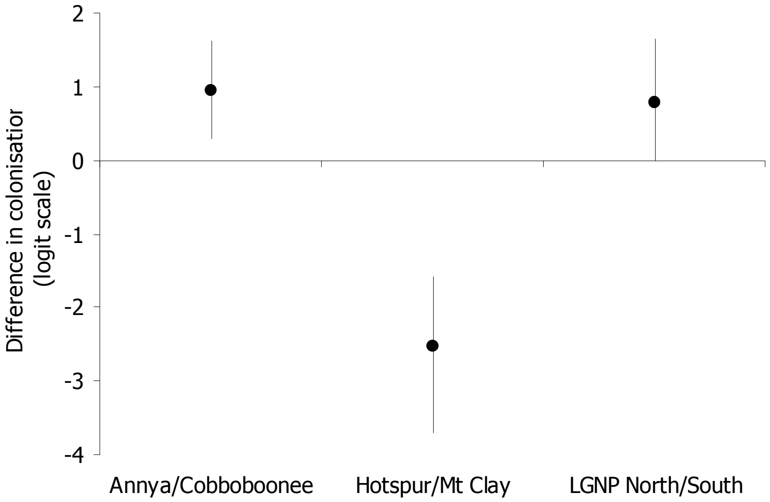

**Figure 24. Effect of treatment on colonisation for Common Brushtail Possum at non-treatment/treatment pair.** Bars are 95% credible intervals.

**Table 4. 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 | 0.18 | 0.36 | 0.98 |
| Colonisation | 0.99 | <0.001 | 0.97 |

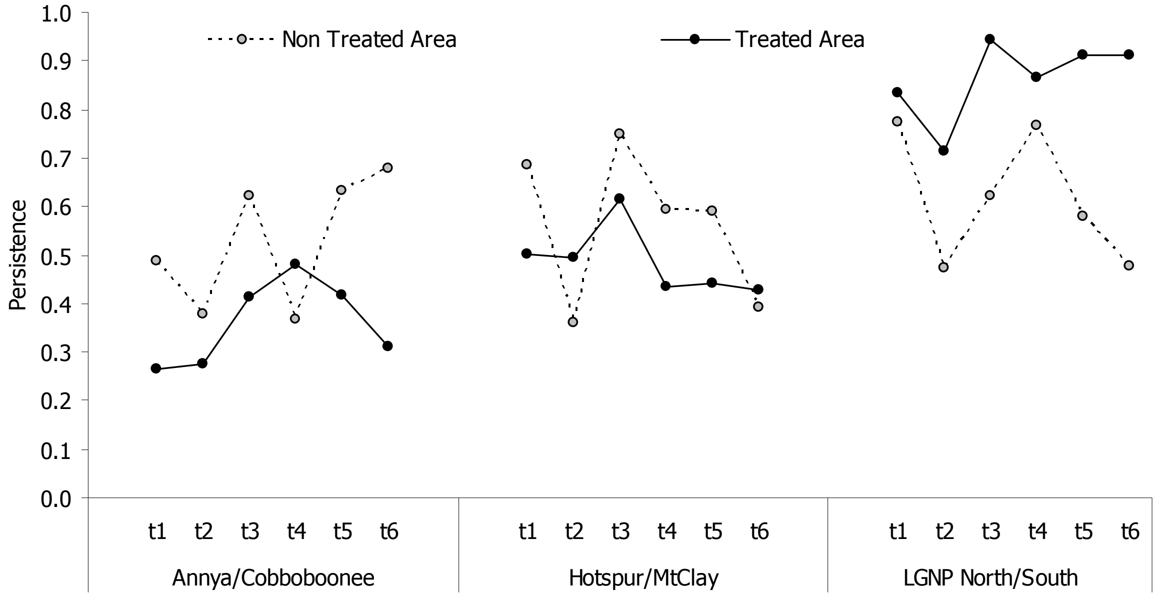

**Figure 25. Derived estimates of mean persistence for Common Brushtail Possum for each non-treatment/treatment pair.**

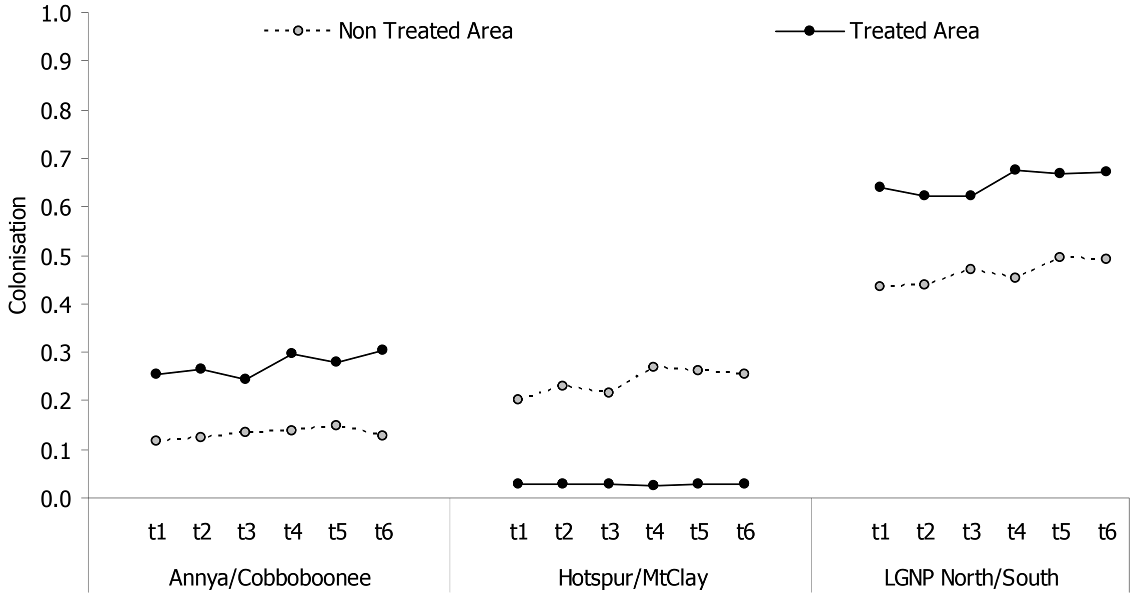

**Figure 26. Derived estimates of mean colonisation for Common Brushtail Possum for each non-treatment/treatment pair.**

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 Lower Glenelg National Park 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 25). 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).

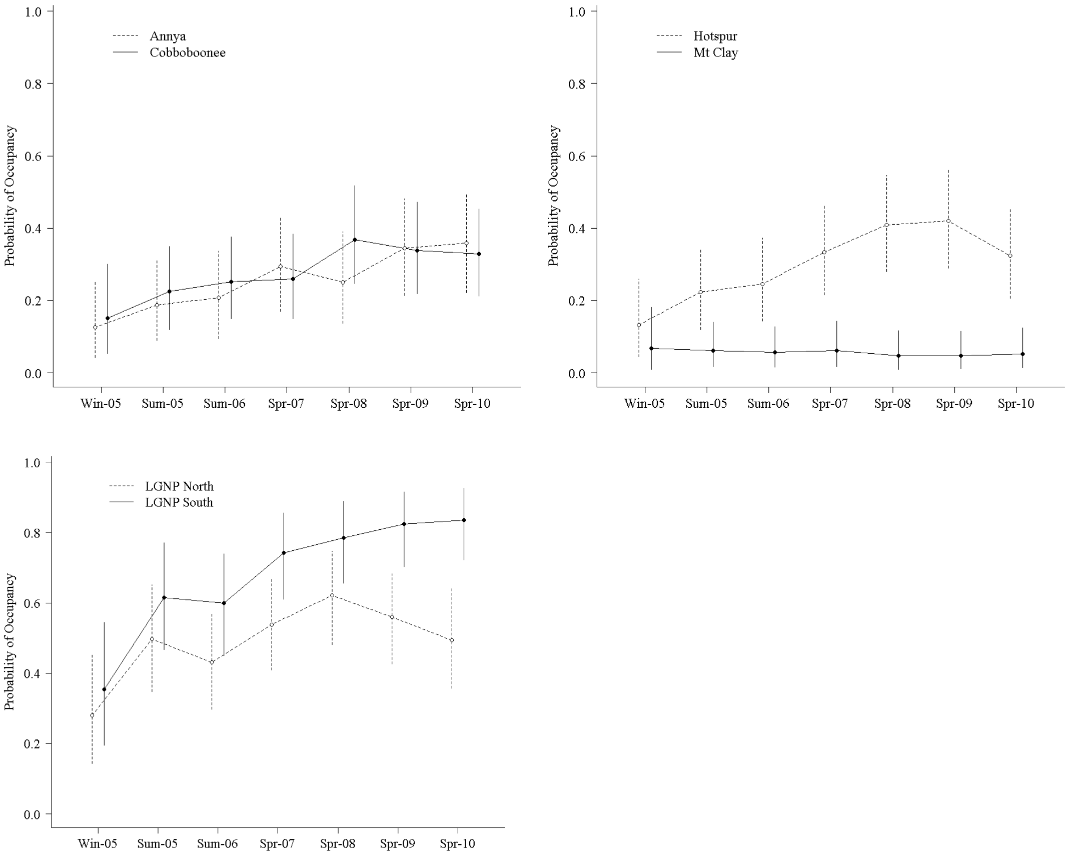

**Figure 27. Derived estimates of occupancy for Common Brushtail Possum for each non-treatment/treatment pair.** Bars are 95% credible intervals.

## 4.4 Heath Mice Surveys

No Heath Mice were detected at any site using any of the three survey methods investigated in this study. We analysed the differences in capture rates (captures/100 trap nights) between treated and non-treated sites for mammal species that were detected in each survey method.

### 4.4.1 Hair tube surveys

Hair tube surveys were conducted at Hotspur State Forest and Mt Clay State Forest. Seven mammal species were detected at Mt Clay and four at Hotspur (Table 5). *Rattus* spp. was the most commonly detected taxa on both sites, with Southern Brown Bandicoots (Mt Clay) and Common Brushtail Possum (Hotspur) the second most commonly detected species. Detections were higher (range 50–100%) on the treated sites compared to the non-treated sites.

**Table 5. Results of the hair-tube survey (detection/100 tube nights) at Hotspur State Forest and Mt Clay State Forest.**

| Common Name | Scientific Name | Hotspur<br>No. Sites | Hotspur<br>Detection Rate | Mt Clay<br>No. Sites | Mt Clay<br>Detection Rate | Percentage Difference |
|---|---|---|---|---|---|---|
| Agile Antechinus | *Antechinus agilis* | 1 | 6.2 | 5 | 31.3 | 80.0 |
| *Antechinus* sp. | *Antechinus* spp. | 0 | 0 | 5 | 31.3 | 100.0 |
| Common Brushtail Possum | *Trichosurus vulpecula* | 3 | 18.7 | 0 | 0.0 | -100.0 |
| Short-beaked Echidna | *Tachyglossus aculeatus* | 0 | 0 | 1 | 6.3 | 100.0 |
| Long-nosed Potoroo | *Potorous tridactylus* | 0 | 0 | 5 | 31.3 | 100.0 |
| *Rattus* spp. | *Rattus* spp. | 6 | 37.5 | 12 | 75.0 | 50.0 |
| Southern Brown Bandicoot | *Isoodon obesulus* | 1 | 6.2 | 3 | 18.8 | 66.7 |
| Black Wallaby | *Wallabia bicolor* | 0 | 0 | 2 | 12.5 | 100.0 |
| **Total** | | **11** | **68.6** | **33** | **175.2** | **60.8** |

### 4.4.2 Camera surveys

Camera surveys were only conducted at Hotspur State Forest and Mt Clay State Forest. At Hotspur, 15 Pix-controller cameras were established, however, four failed and two were stolen, resulting in 9 cameras operating for an average of 14 days (126 camera trap nights). At Mt Clay, 15 Pix-controller cameras were set-up, with only one failure, resulting in 14 cameras operating for an average of 14 days (210 camera trap nights).

At Hotspur, nine mammal species were detected (Table 6) including Southern Brown Bandicoot (0.7 captures/100 trap nights) and Long-nosed Potoroo (1.4). The most commonly detected species were Common Brushtail Possum and Black Wallaby.

Eleven mammal species were detected at Mt Clay, again including Southern Brown Bandicoot (6.7) and Long-nosed Potoroo (17.6 captures/100 trap nights). Common Brushtail Possums were also detected, but less commonly than at Hotspur.

**Table 6. Results of camera trapping (captures/100 trap nights) at Hotspur State Forest and Mt Clay State Forest.**

| Common Name | Scientific Name | Hotspur | Mt Clay | Percentage difference |
|---|---|---|---|---|
| Bush Rat | *Rattus fuscipes* | 1.4 | 23.8 | 94.0 |
| Common Brushtail Possum | *Trichosurus vulpecula* | 10.7 | 2.4 | -77.8 |
| Common Ring-tail Possum | *Pseudocheirus peregrinus* | 0.0 | 2.9 | 100.0 |
| Eastern Grey Kangaroo | *Macropus giganteus* | 0.0 | 0.5 | 100.0 |
| Eastern Pygmy Possum | *Cercartetus nanus* | 0.7 | 0.5 | -33.3 |
| Koala | *Phascolarctos cinereus* | 0.7 | 1.0 | 25.0 |
| Long-nosed Potoroo | *Potorous tridactylus* | 1.4 | 17.6 | 91.9 |
| Red-necked Wallaby | *Macropus rufogriseus* | 2.9 | 1.4 | -50.0 |
| Short-beaked Echidna | *Tachyglossus aculeatus* | 0.7 | 1.4 | 50.0 |
| Southern Brown Bandicoot | *Isoodon obesulus* | 0.7 | 6.7 | 89.3 |
| Black Wallaby | *Wallabia bicolor* | 8.6 | 7.1 | -16.7 |
| **Total** | | **27.9** | **65.2** | **57.3** |

### 4.4.3 Elliott trapping

Seven mammal species and one reptile species were recorded by traps at Mt Clay and Hotspur (Table 7). Capture rates for mammals were 77.8% higher at Mt Clay compared to Hotspur. Eastern Water Skinks, *Eulamprus quoyii* were only captured at Hotspur. Long-nosed Potoroo and Common Ring-tail Possum, *Pseudocheirus peregrinus* (0.7 and 0.3 captures/100 trap nights) were only captured at Mt Clay.

**Table 7. Results of Elliott trapping (captures/100 trap nights) at Hotspur State Forest and Mt Clay State Forest.**

| Common Name | Scientific Name | Hotspur | Mt Clay | Percentage Difference |
|---|---|---|---|---|
| Agile Antechinus | *Antechinus agilis* | 2.7 | 0.7 | -74.1 |
| Bush Rat | *Rattus fuscipes* | 10.3 | 33.3 | 69.1 |
| Common Ring-tail Possum | *Pseudocheirus peregrinus* | 0.0 | 0.3 | 100.0 |
| Dusky Antechinus | *Antechinus swainsonii* | 8.0 | 33.7 | 76.3 |
| Eastern Pygmy Possum | *Cercartetus nanus* | 0.0 | 1.7 | 100.0 |
| Long-nosed Potoroo | *Potorous tridactylus* | 0.0 | 0.7 | 100.0 |
| Swamp Rat | *Rattus lutreolus* | 0.3 | 25.7 | 98.8 |
| **Total Mammals** | | **21.3** | **96.0** | **77.8** |
| Eastern Water Skink | *Eulamprus quoyii* | 1.0 | 0.0 | 100.0 |
| **Total Reptiles** | | **1.0** | **0.0** | **100.0** |

Additional trapping was undertaken at Annya/Cobboboonee. Eight mammals and two reptile species were recorded (Table 8). Captures rates were 46.3% and 30.0% higher for mammals and reptiles at Cobboboonee compared to Annya. Long-nosed Potoroos were only captured at Cobboboonee. There were 88.9% more captures of Common Brushtail Possums and 57.1% more captures of Long-nosed Potoroos at Cobboboonee compared to Annya.

**Table 8. Results of Elliott trap surveys (captures/100 trap nights) at Annya State Forest and Cobboboonee National Park.**

| **Common Names** | **Scientific Names** | **Annya** | **Cobboboonee** | **Percentage Difference** |
|---|---|---|---|---|
| Agile Antechinus | *Antechinus agilis* | 11.4 | 26.7 | 57.3 |
| Bush Rat | *Rattus fuscipes* | 60.0 | 73.3 | 18.1 |
| Common Brushtail Possum | *Trichosurus vulpecula* | 1.0 | 8.6 | 88.4 |
| Eastern Pygmy Possum | *Cercartetus nanus* | 5.7 | 0.0 | -100.0 |
| Long-nosed Potoroo | *Potorous tridactylus* | 0.0 | 7.6 | 100.0 |
| Southern Brown Bandicoot | *Isoodon obesulus* | 2.9 | 6.7 | 56.7 |
| Swamp Antechinus | *Antechinus minimus* | 1.0 | 30.5 | 96.7 |
| Swamp Rat | *Rattus lutreolus* | 1.9 | 9.5 | 80.0 |
| **Total Mammals** | | **82.9** | **163.8** | **49.3** |
| Blotched Blue-tongue Lizard | *Tiliqua nigrolutea* | 7.6 | 5.7 | -25.0 |
| Skink sp. | Scincidae | 7.6 | 16.2 | 53.1 |
| **Total Reptiles** | | **15.2** | **21.9** | **30.6** |

# 5 Camera orientation

We established 82 camera stations across the six study sites in Glenelg Ark (Table 9).

**Table 9. Numbers of cameras, camera type and their orientation at each location during the camera orientation trials.**

| **Site** | **ScoutGuard** | | **Reconyx** | | **Total** |
|---|---|---|---|---|---|
| | **Down** | **Out** | **Down** | **Out** | |
| Annya State Forest | 4 | 4 | 3 | 3 | 14 |
| Cobboboonee NP | 4 | 4 | 3 | 3 | 14 |
| Hotspur State Forest | 3 | 3 | 3 | 3 | 12 |
| Mt Clay State Forest | 2 | 2 | 5 | 5 | 14 |
| Lower Glenelg NP north | 4 | 4 | 3 | 3 | 14 |
| Lower Glenelg NP south | 3 | 3 | 4 | 4 | 14 |
| **Total** | **20** | **20** | **21** | **21** | **82** |

We were able to assess the differences in detection probability of camera type and camera orientation for Common Brushtail Possum, medium-sized species (those between 100 mm and 300 mm head/body length) and small mammals (less than 100 mm head/body length). Although we detected a range of other species, detection rates were not sufficient to undertake analysis.

Common Brushtail Possums tended to be detected more frequently by downward facing Reconyx cameras than ScoutGuard downward facing cameras (Figure 28). Downward facing cameras had a significantly higher detection rate for medium-sized mammals (Figure 29) with no difference in camera type. Small mammals were detected more by downward facing cameras also; however, Reconyx had a significantly higher detection rate than ScoutGuard cameras (Figure 30).

Overall Reconyx had higher detection rates than ScoutGuard cameras and downward facing cameras generally had higher detection rates than outward facing cameras.

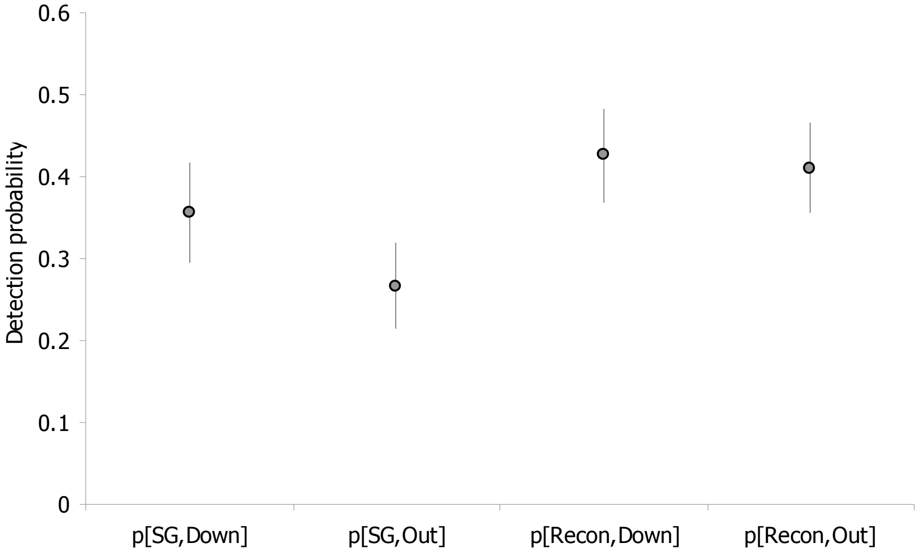

**Figure 28. Detection probabilities of Common Brushtail Possums for ScoutGuard (SG) and Reconyx (Recon) orientated downwards and outwards.**

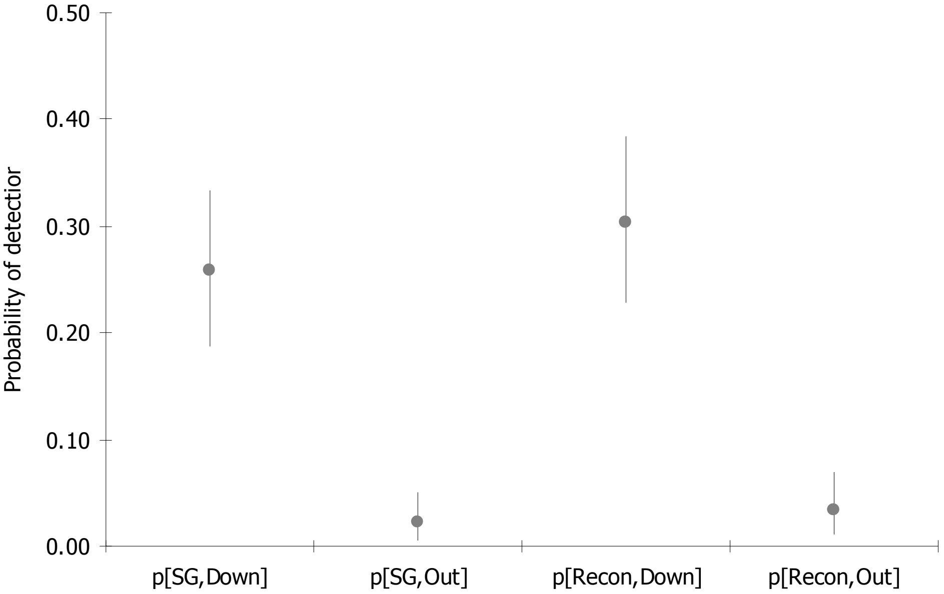

**Figure 29. Detection probabilities of medium-sized mammals (between 100 mm and 300 mm body length) for ScoutGuard (SG) and Reconyx (Recon) orientated downwards and outwards.**

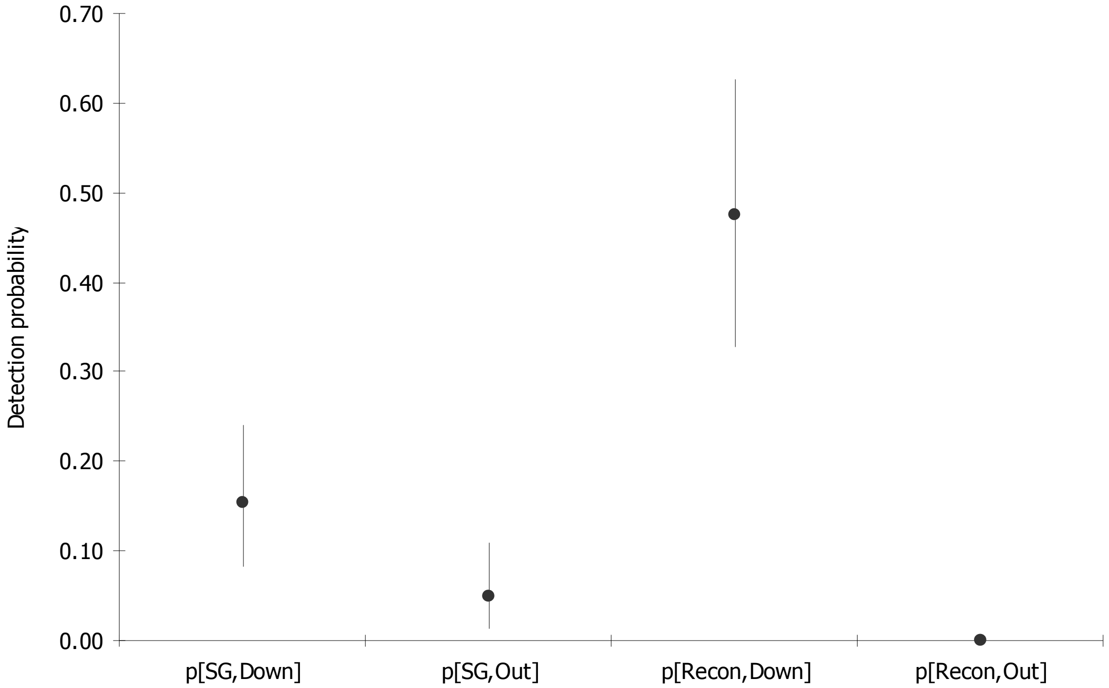

**Figure 30. Detection probabilities of small mammals (< 100 mm body length) for ScoutGuard (SG) and Reconyx (Recon) orientated downwards and outwards.**

# 6 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, more certainty around the nature of these responses will be achieved.

Indices of abundance derived from bait take data for foxes at all three treated areas (Lower Glenelg National Park south, Cobboboonee National Park and Mt Clay State Forest) clearly demonstrate a sustained reduction in fox populations. Data for fox activity at treatment and non-treatment sites using sand plots was highly variable and did not show a clear difference. Heat-in-motion camera data leant support for the sustained reduction of foxes at LGNP.

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.

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 Lower Glenelg National Park south, bandicoots were to be 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 treated 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-treated 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-treated site and increased marginally at the treated 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-treated site.

The probability of Southern Brown Bandicoots persisting and colonising sites at Cobboboonee National Park was significant, leading to higher derived estimates of site occupancy at Cobboboonee National Park compared to Annya State Forest. This suggests that Southern Brown Bandicoots are able to survive in some parts of the landscape and increase in the presence of low levels of predation, but not in areas were foxes are relatively higher. The overall derived estimate of occupancy suggests that bandicoots have remained stable on the treated site and declined on the non-treated site.

At Mt Clay, bandicoots were able to survive in parts of the landscape, but colonisation rates (births) were not significantly greater on the treated 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, however, the estimates are 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 not significantly dissimilar suggesting that at these sites Long-nosed Potoroos are regulated by something other than fox predation, or that the level of 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 treated 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 regulation, 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. Cat control and camera data indicate that feral cat populations on the treated site at LGNP were three times higher than on the non-treated site. While no pre-fox control data are available on cats at these sites, it suggests that cats may have replaced foxes to some degree as the apex mammalian predator.

It has been suggested (J. Smith, pers. comm.) that the orientation of cameras used to detect native species can influence the detection rate and subsequently estimates of occupancy. We investigated this possibility for the camera types used in Glenelg Ark. Downward facing Reconyx cameras tended to return a higher rate of detection for mammals in this study. Common Brushtail Possums and mammals between 100 mm and 300 mm body length were all detected at higher rates by Reconyx and by downward facing cameras. We did detect Southern Brown Bandicoots and Long-nosed Potoroos at Mt Clay and Long-nosed Potoroos at Cobboboonee but the detection rates were too low for any analysis. Neither species were detected at LGNP south nor were they detected at any of the non-treatment sites. A similar study at 18 sites at Mt Clay with two Pix-controller cameras one facing horizontally and one vertically at Mt Clay, found the detection probability of Long-nosed Potoroos and Southern Brown Bandicoots by vertical cameras to be 2 to 5 times higher than horizontal cameras. There was no significant difference between the two orientations with respect to the number of false triggers; however, ease of medium-sized species identification was increased by vertical cameras (J. Smith, pers. comm.). It is possible to set cameras so that the detection zone is aimed inaccurately with outward facing orientation, leading to missed photo opportunities. PixController cameras have a 10° cone-shaped, passive infrared detection area. Shifting the camera unit up or down could result in a gap between the detection area and the ground, allowing species to move past without triggering the camera (J. Smith, pers. comm.). Both Reconyx and ScoutGuard cameras have different detection shaped zones. The Reconyx manual indicates it has a field of view of $40^0$ and a sensing distance of up to 30 m, while there is no information provided in the ScoutGuard manual. Thus, it is possible that differences in these led to the differences in the detection rates recorded by cameras used in our study.

We also assessed the relative ability of different devices to detect the presence of the rare Heath Mouse, and other small mammals across sites in the Glenelg Ark study area. While medium-sized mammals have been the focus of a number of fox control programs, small mammals may well respond to the removal of foxes. While no Heath Mice were captured by the hair tube, camera or Elliott trap surveys, these surveys did provide additional valuable information on the relative occurrence of native species on sites with and without fox control. Results from the three survey methods indicated that Long-nosed Potoroos were more common at Mt Clay compared to Hotspur and the additional small mammal trapping with Elliott traps indicated that they were also more common at Cobboboonee National Park compared to Annya State Forest. Camera surveys and the extended hair-tube survey also indicated that Southern Brown Bandicoots were detected more often at Mt Clay than Hotspur. These two methods also supported the results from the occupancy modelling for Common Brushtail Possums, showing that this species is more common at Hotspur than Mt Clay. Overall, a range of mammal species were detected more frequently at Mt Clay than at Hotspur, supporting the inference that there is a positive association between sites with fox control in place and the relative occurrence of native mammals species.

The results of the monitoring and evaluation program clearly demonstrate that foxes have been reduced and remain at relatively lower 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 broad-scale 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;
- 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 will 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) and any increase in native species. This phase could also 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 regulating 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 broad scale continuous fox control through the rigorous implementation of the monitoring program. This report has highlighted several areas for future consideration. Consideration could also be given to how the monitoring of Glenelg Ark should continue into the future.

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# 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:

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

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:

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

Combining these gives time and treatment specific occupancy probabilities:

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

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:

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

Therefore a 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:

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

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:

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

Prior probabilities were placed on all parameters, specifically Inverse Gamma (0.001, 0.001) for the random effects parameters $\sigma_\phi$ and $\sigma_\gamma$, Normal(0, 100) for the mean site specific persistence and colonisation parameters $\alpha_{site}$ and $\beta_{site}$, 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 $\alpha_{\text{site}}$ and $\beta_{\text{site}}$ parameters. For example, in the case of the Annya/Cobboboonee pair, the overall effect of treatment on persistence is given by:

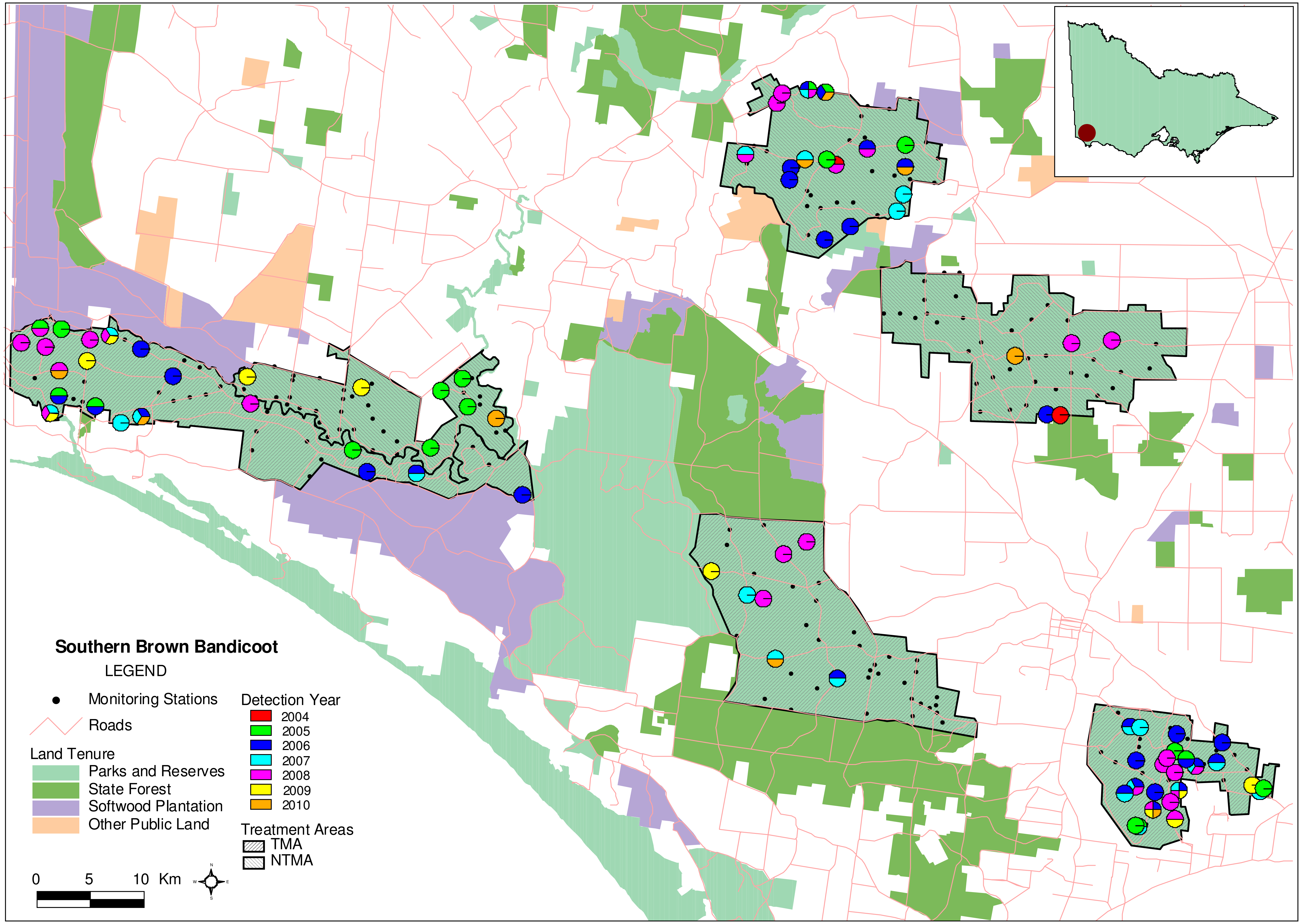
$$\alpha_{Cob-Ann} = \alpha_{Cob} - \alpha_{Ann}$$

and for colonisation is given by:

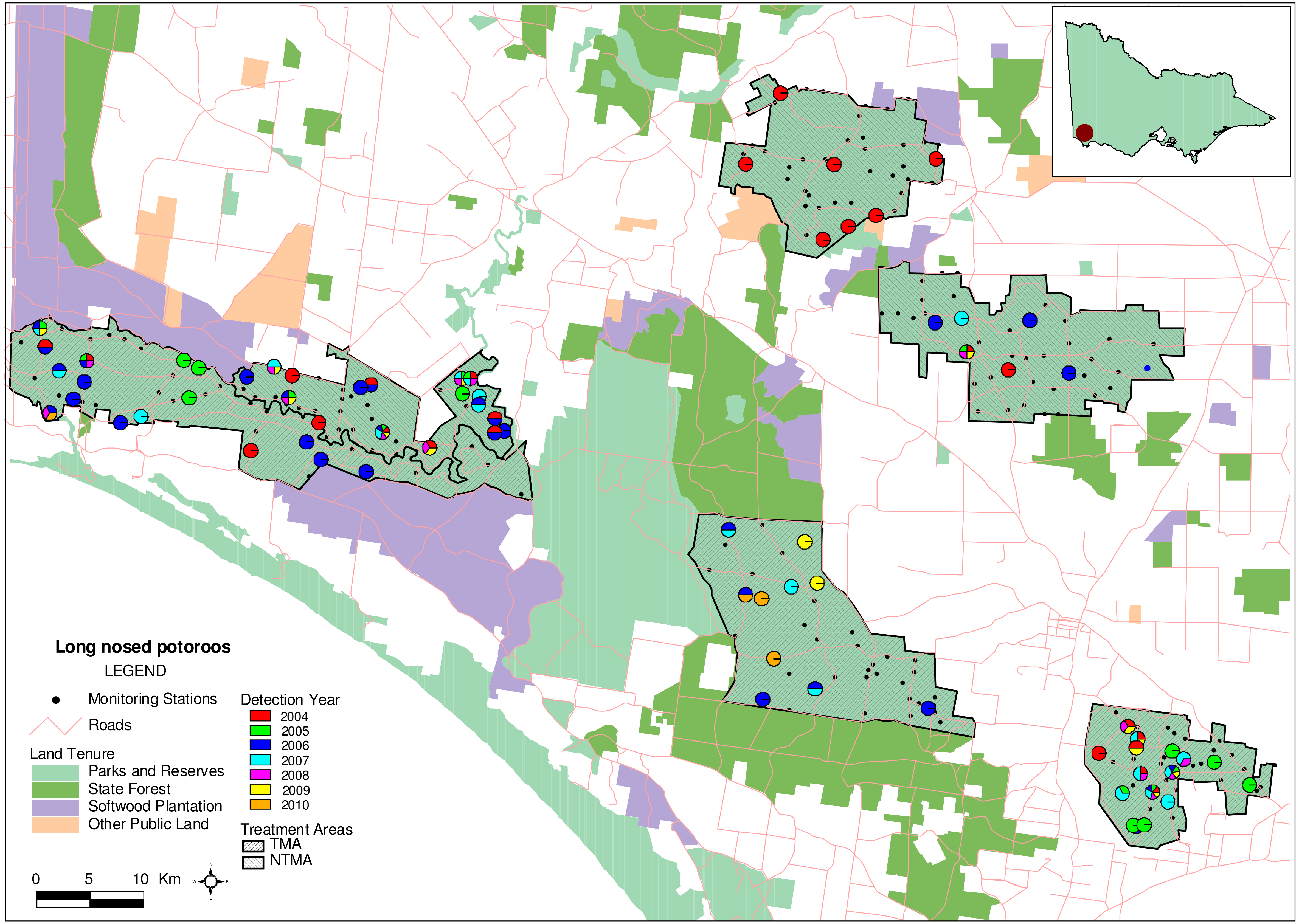
$$\beta_{Cob-Ann} = \beta_{Cob} - \beta_{Ann}$$

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 Southern Brown Bandicoots across the Glenelg Ark monitoring area



## Appendix 4. Location and year of detection for Long-nosed Potoroos across the Glenelg Ark monitoring area



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