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Front cover photos (clockwise from top left): Red Fox inspecting bait station; feral cat with native species prey; burnt forest; forest fire; monitoring native species response using camera traps (photographer: DELWP).

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The response of native species to the 2019–20 bushfires and introduced predators in far East Gippsland

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Context:

The 2019–20 bushfires had a major impact on the forests of eastern Victoria. The impact on native species would have been 2-fold: first, through the direct mortality caused by the large-scale and intense fires; second, by the significant removal of vegetation caused by fire, which can impact the survival of native species by reducing sites for nesting, breeding and shelter, and the availability of food. In addition, the removal of protective cover can allow for greater hunting efficiency of predators, also increasing the risk of predation; for example, by the red fox and feral cat.

The Southern Ark project has implemented year-round fox control across nearly 1 million hectares in eastern Victoria since 2005 with the main aim of protecting a range of native species at risk from fox predation. In 2016–17, 2019 (pre-fire) and in 2020 (6 months post-fire), the Southern Ark project carried out a broadscale camera-trap monitoring program. Data from this surveillance program provides an opportunity to investigate the short-term impact of the 2019–20 fires on the occurrence of the threatened Long-footed Potoroo and a range of other native species that are at risk from introduced predators.

Aim:

The aim of this project was to investigate the impact of the 2019–20 bushfires and the occurrence of introduced predators on native species; in particular, the threatened Long-footed Potoroo. This will support the Department of Environment, Land, Water and Planning's Gippsland Natural Environment Program and help to achieve the Southern Ark project aims and objectives.

Methods:

The Southern Ark monitoring design comprised 240, 5 km^2 cells, each with 3 camera trapping points. Each point was surveyed at intervals of 1 month with mammal bait as a lure. The third point was resurveyed with a predator lure for an additional month. We used an occupancy modelling approach to analyse the camera-trap data collected in 2016–17, 2019 (both pre-bushfire) and 2020 (6-months post bushfire) to assess the response of native species, including the Common Brush-tailed Possum, Lace Monitor, Long-footed Potoroo, Long-nosed Bandicoot and Long-nosed Potoroo, to the 2019–20 bushfires and the presence of predators – feral cats, foxes and wild dogs (which includes Dingoes, *Canis familiaris*, and their domestic dog hybrids).

Data analysis:

To investigate the impact of predators and fire on native species, we used a Bayesian approach to model occupancy while accounting for imperfect detection. First, we generated predictions of occupancy for feral cats and wild dogs that were recorded in 2016–17, 2019 and 2020. For foxes, we used an existing model of fox density. These site- and season-specific model predictions were then used as covariates within a native species occupancy model. We used model selection on frequentist mixed-effects models of predator and native species occupancies. The top model (lowest Akaike Information Criterion) from model selection was then used in a Bayesian occupancy model, which accounted for imperfect detection due to the inclusion of a detection submodel. We also included a range of environmental covariates, including fire severity, to explore their effect on occupancy and detection.

Results:

There was no significant change in mean site occupancy post-fire for any of the monitored native species. There was a tendency for mean site occupancy to be lower for Lace Monitors, Long-nosed Bandicoot and Long-nosed Potoroo, while Long-footed Potoroos showed very little difference in occupancy post-fire. Common Brush-tailed Possum showed a very slight (negligible) increase in occupancy when comparing pre-fire and post-fire data.

Occupancy in 2020 was influenced by the severity of the fire in the surrounding landscape: the magnitude of the fire severity significantly (95% CIs not overlapping zero) reduced occupancy for 4 of the native species. Lace Monitors had the largest estimated change with fire severity (while accounting for temperature

differences between sampling seasons), followed by the Long-nosed Potoroo, Long-nosed Bandicoot and Long-footed Potoroo.

Fox density was a significant variable in explaining the occurrence of Long-footed Potoroo, with the probability of Long-footed Potoroos occurring at a site being zero when fox density was higher than 0.5/km^2. There was a weaker positive relationship between Long-footed Potoroo occupancy and feral cat occupancy; indicating that feral cats were likely to occupy sites that were also occupied by Long-footed Potoroos. However, this relationship is confounded by both feral cats and Long-footed Potoroos tending to occupy sites with lower mean annual temperature. It is possible that climatic or environmental factors drive this association. For all other species, there was no significant relationship with predator density or occurrence.

There is moderate evidence that wild dog and feral cat mean estimated occupancies were impacted by fire severity. However, this trend is somewhat uncertain depending on whether we compare occupancy across high and low fire intensity percentiles or the continuous fire severity gradient which shows that the upper 95% confidence interval for wild dogs just overlaps zero (0.098). There is no evidence that foxes were impacted by the 2019–20 fires.

Conclusions and implications:

This study suggests that the 2019–20 bushfires had minimal impact on the immediate (6-months) post-fire occurrence of all assessed native species, including Long-footed Potoroos, in far East Gippsland. However, there was evidence that the severity of the fire did impact site occupancy in 2020. Lace Monitors appear to be the most impacted by the fires with both Long-nosed Potoroos and Long-nosed Bandicoots also being detected at fewer sites post-fire. Long-footed Potoroos were more likely to occur at sites with low fox densities. There was no relationship between the presence of feral cats or wild dogs and the occurrence of any of the monitored species.

Despite varying spatial and temporal sampling effort between 2016–17, 2019 and 2020, there was enough data to investigate the differences in occupancy of Long-footed Potoroos before and after the 2019–20 bushfires at sites where they were previously detected. For other species, such as the Long-nosed Potoroo, sampling only at the sites at which Long-footed Potoroos had previously been detected limits the ability to estimate the role of predation and other environmental variables and to make comparisons more broadly across the Southern Ark area.

Future work

There are several possible activities that could build on the current camera monitoring program to improve our understanding of the response of native species in severely fire-affected landscapes and the possible role that introduced and native predators might play in that recovery:

- To reduce issues arising from the variable sampling effort, re-cast future camera surveys to sample across a range of fire-severity categories and predator densities.
- To assess the effectiveness of fox control, develop a long-term monitoring protocol for a selected range of species of conservation interest. This could use spatially explicit occupancy models to explore the changes in species distribution pre-baiting (constructing distributions from historical data) to post-baiting and incorporating the impact of the 2019–20 fires to predict likely future distribution.
- To improve model predictions and to assess the response of native species if feral cat control were introduced into Southern Ark in the future, robustly estimate fox and feral cat densities.
- Update the power analysis undertaken by Bluff (2016) using current data to determine what level of changes could be detected related to different levels of fox density and to explore options to improve the design.
- Include the 2021 camera monitoring data into the analysis undertaken in this report when it becomes available.

1 Introduction

In 2019–20, bushfires impacted 1.5 million hectares of eastern Victoria. It has been estimated that 244 plant and animal species have more than 50% of their modelled habitat in the burnt areas, including 215 rare or threatened species; and 43 species had more than 50% of their modelled habitat impacted by high-severity fires, including 42 rare or threatened species (DELWP 2020).

Fire is a complex phenomenon that can have positive and negative effects on the functioning of ecosystems, including predator–prey and competitive interactions between species (Bond and Keeley 2005). Fire can reduce vegetation cover that would normally provide protection from predation (Radford 2012; Lentic et al. 2013; Hradsky 2020) and can lead to higher hunting success for predators (Hradsky 2020), directly impacting vulnerable native species such as bandicoots and potoroos (Hradsky et al. 2019a). This is because post-fire, the absence of the cover normally provided by vegetation and woody debris can make it easier for predators to detect and capture prey (e.g. Mandelik et al. 2003).

Fire can also have both positive and negative impacts on food availability. For example, habitats simplified by burning may be important for foraging – including for Long-nosed Potoroos (*Potorous tridactylus*) that exploit the presence of hypogeal fungi following a fire (Bennett 1993; Vernes et al. 2004; Norton et al. 2015) – despite the increased risk of fox predation (Norton et al. 2015). Conversely, fire can decrease the availability of food resources for some species by reducing the composition and structure of the vegetation (Woinarski et al. 2004b). Some research has indicated that small- to medium-sized ground-dwelling mammals, such as antechinus, potoroos and bandicoots, require a complex understorey with a diverse shrub layer and a ground cover of logs, leaf litter and woody debris for optimal foraging (Paull and Date 1999).

The introduced fox (red fox, *Vulpes vulpes*) spread rapidly across Australia following its introduction in 1870 and is now widespread and common across the continent (Saunders and McLeod 2007). Between the 1980s and the early 2000s, a body of evidence was generated on the impacts of foxes on native species in Australia (e.g. Burbidge and McKenzie 1989; Kinnear et al. 2002). The evidence indicated that foxes are omnivorous hunters that prey on small native mammals, birds, reptiles, amphibians and insects as well as eating fruits (Triggs et al. 1984). Fox predation has been implicated as the main factor in the complete or regional extinction of a range of critical weight range (35–5,500 g) native mammals (Burbidge and McKenzie 1989; Short and Smith 1994), as well as small reptiles (Olsson et al. 2005; Stobo-Wilson et al. 2021) and ground-nesting birds (Dickman 1996).

Along with foxes, feral cats (*Felix catus*) have been implicated in many native animal extinctions and declines since European settlement (Short and Smith 1994; Woinarski et al. 2011, 2015; IUCN 2019). Over the past few decades, research has demonstrated the significant impact of feral cats on native wildlife through direct predation (e.g. Nogales et al. 2004; Marlow et al. 2015; Jones et al. 2016). It has been shown that feral cats preferentially select mammals as prey (less than 10 g to 3–4 kg) (Paton 1993) and that some individual feral cats can be disproportionately responsible for predation on populations of native species (Moseby et al. 2015). There is evidence that feral cats also become the main predator of medium-sized mammals (0.7–2.0 kg) following the sustained control of foxes (Marlow et al. 2015). In northern Australia, feral cats had a strong preference for recently burnt areas (1–2 months old) but avoided areas burnt 3 months or more ago, and cats were found to travel long distances to recently burnt areas (McGregor et al. 2017).

In 2005, the Victorian Government initiated large-scale fox control projects, known as Arks. In Gippsland, the Southern Ark project (covering nearly 1 million hectares), managed by the Department of Environment, Land, Water and Planning (DELWP) and Parks Victoria, makes a significant contribution to the delivery of fox control across the conservation network in eastern Victoria.

Although Southern Ark has significantly reduced fox density in its operational area, foxes are still broadly distributed across the area, albeit at a low density (Francis et al. 2019). Few environmental factors limit their distribution, and there is evidence that localised fox activity can increase following fire (Hradsky et al. 2019a). The 2019–20 bushfires impacted 65–75% of the Southern Ark operations area. Threatened native mammal species in this area that are particularly at risk from fox (and potentially feral cat) predation and that had their habitat impacted by these fires are:

- Long-footed Potoroo (*Potorous longipes*) – 79% of its modelled habitat in Victoria was within the fire extent, and 51% of its modelled habitat was impacted by high-severity fire
- Southern Brown Bandicoot (*Isoodon obesulus*) – 25% of its modelled habitat in Victoria was within the fire extent, and 19% of its modelled habitat was impacted by high-severity fire.

Other species of concern that may also have been impacted by fox predation following the bushfires include the Broad-toothed Rat (*Mastacomys fuscus mordicus*), Long-nosed Bandicoot (*Perameles nasuta*), Long-nosed Potoroo, Spotted-tail Quoll (*Dasyurus maculatus*) and Smoky Mouse (*Pseudomys fumeus*).

The Long-footed Potoroo is a medium-sized terrestrial rat-kangaroo that inhabits forests in south-eastern Australia. It is currently known from 3 apparently disjunct populations in East Gippsland, the Victorian Great Dividing Range and south-eastern New South Wales. The Long-footed Potoroo’s poor conservation status – listed as Endangered under the Victorian *Flora and Fauna Guarantee Act 1988* – is based on its restricted and fragmented distribution, low population density and vulnerability to predation by wild dogs (*Canis familiaris,* Dingoes and their feral domestic dog hybrids) (Jackson et al. 2019[1]), foxes and feral cats. Inappropriate fire regimes and climate change are also thought to be threatening processes for the Long-footed Potoroo.

A 5-year study of the Great Dividing Range population of Long-footed Potoroos (Lumsden et al. in press) indicates that Long-footed Potoroo populations can survive severe broadscale fire events and, at least in the short term, utilise areas of burnt habitat. Although the long-term impacts of fire on this species are unknown, fire appears to have had less influence on the distribution of the species in the Great Dividing Range area than foxes. Initial results from a post-fire camera survey in 2020 across the Southern Ark area suggests that Long-footed Potoroos may not have been severely impacted by the fires in the short term (A. Murray pers. obs.).

The Southern Ark project includes a camera-trap program monitoring the presence of animals at 720 sites. The objective of the program is to measure changes in occupancy by native species and introduced predators. The design may also be able to address factors that might influence the presence of native and introduced species. Monitoring was undertaken in 2016–17, 2019 (pre-fire) and 2020 (6 months post-fire). While not specifically designed to investigate the impact of bushfires, the broad array of camera-trap sites was considered suitable to assess of the impact of the 2019–20 bushfires and the interaction with the presence of predators. This will further our understanding of the short-term impact that large-scale fires can have on native species at risk from introduced predators. This knowledge can help direct post-fire recovery efforts in the future.

We aimed to use the data collected from this monitoring program to investigate the impact of the 2019–20 bushfires on the occurrence of Common Brush-tailed Possums (*Trichosurus vulpecula*), Lace Monitors (*Varanus varius*), Long-footed Potoroos, Long-nosed Bandicoots, Long-nosed Potoroos and Southern Brown Bandicoots that are threatened by predation, and to explore the role of remotely sensed environmental variables.

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[1] In this report, wild dog includes Dingo, since they are difficult to distinguish by observation alone.

2 Methods

2.1 Study area

The study area was within the Southern Ark project area, encompassing the eastern corner of Victoria, from the Snowy River valley to Cape Howe. It assists the recovery of multiple species across nearly 1 million hectares of state forests, national parks and private land.

Southern Ark operates a year-round fox-baiting program along a network of roads and tracks throughout its project area (Figure 1). Baits are spaced on average 690 m apart (range 3–2,144 m) and are replaced every 6 weeks. Parks Victoria also has a year-round wild dog baiting program in the Southern Ark region, involving year-round baiting every 2 weeks (for perishable baits) or 4 weeks (for shelf-stable baits), with about a 40:60 ratio of perishable to shelf-stable baits.

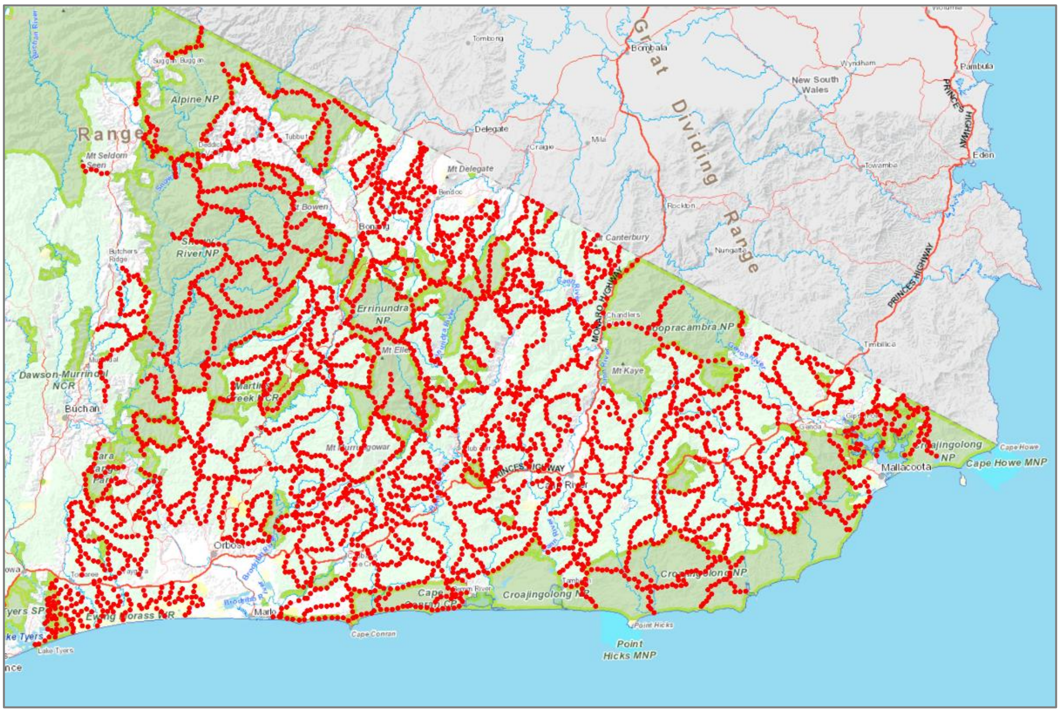

Figure 1. The Southern Ark operations area in East Gippsland and location of bait Stations (red dots) for the control of foxes

A significant proportion of the Southern Ark operations area was impacted by the 2019–20 bushfires (Figure 2). Over 150 bushfires began in November 2019 and burnt approximately 1.1 million hectares in East Gippsland before the fires were contained in February 2020.

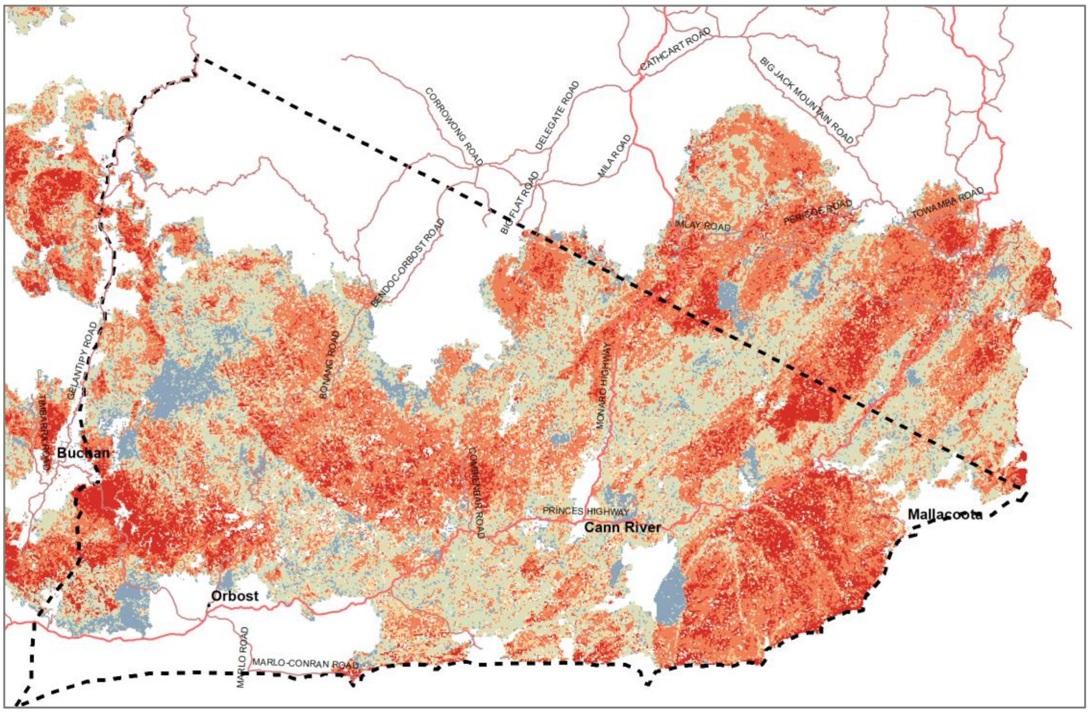

Figure 2. The extent of the 2019–20 bushfires and the level of fire severity within the Southern Ark area (dashed line)

Colours indicate fire severity classes:

red	canopy burnt	>20% canopy foliage consumed
orange	high canopy scorch	>80% of canopy foliage scorched
pale tan	medium canopy scorch	canopy a mosaic of unburnt and scorched foliage, 20–80%
light green	low canopy scorch	canopy foliage largely unaffected (<20% scorched), understorey burnt
grey/blue	unburnt	canopy and understorey foliage largely intact (>90%)
white	unburnt	no evidence of fire and no data (e.g. due to obscuration by cloud, cloud-shadow and/or smoke and haze).

2.2 Camera-trap sampling design

Camera surveys were undertaken by the Southern Ark operations team in 2016–17, 2019 and 2020 to collect information on the presence of native species and introduced predators across the Southern Ark operations area.

For these surveys, the Southern Ark area was divided into 5 km^2 cells. Some cells could not be included because of constraints on access (e.g. freehold land, no vehicle access). This resulted in 240 monitoring cells.

Within each cell, 3 camera-trap sites (designated as Green, Yellow and Red) were selected, resulting in 720 camera sites overall (Figure 3). Sites were spaced 1 km apart and were at least 1 km from any camera trap in an adjacent cell. This allowed for spatial independence in the detection of small- to medium-sized mammals. In each cell, a camera was deployed at the Yellow site for 35 days, then the Green site for 35 days, and then the Red site for 35 days. All cameras had a lure for attracting mammals installed 2 m away, consisting of rolled oats, peanut butter, golden syrup and pistachio essence inside a tea infuser. A fourth survey was conducted at Red camera sites for 35 days but using a predator lure of chicken wings secured in a small wire cage. The initial design relied on all Yellow, then all Green then all Red camera sites being deployed at the same time. This avoided any spatial-temporal confounding.

Heat-in-motion-activated digital cameras (PC900, Reconyx Holmen, WI, USA) were set 50 m off tracks and roads. Cameras were attached to either a wooden stake or a suitable tree, 30 cm above the ground. An area of approximately 1 m × 2 m was cleared of vegetation in front of each camera. Cameras were set to take a sequence of 3 images per trigger.

By June 2016, 97 cells (291 sites) had been established, and cameras were deployed in 88 cells in the northern part of the study area in that month.[2] Cameras were moved between the 3 sites in these cells between June and December 2016. Cameras were then deployed in the remaining 141 cells (423 sites) in the southern part of the study area from mid-December 2016 to early May 2017. The deployment of cameras in each round did not occur across the entire Southern Ark area at the same time as per the initial design.

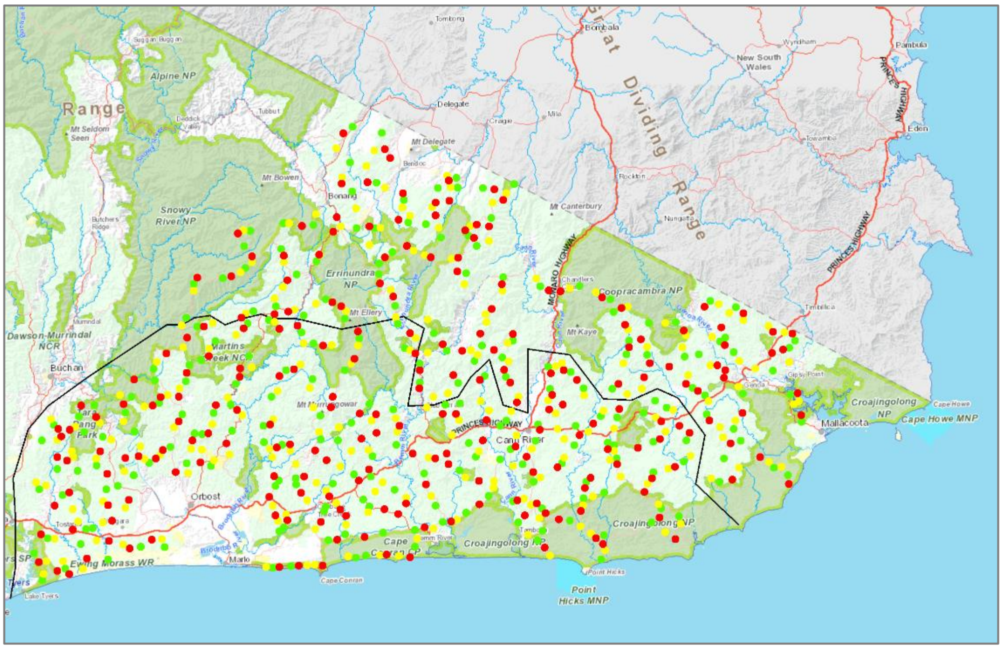

Figure 3. Locations of camera sites surveyed in 2016–17 within the Southern Ark area (coloured dots). The black line indicates the boundary between the initial northern and later southern deployment areas

All Green camera sites in the Southern Ark area were surveyed in October 2019 (Figure 4), and cameras were moved to the Yellow sites in November. Bushfires, which began in early November, then destroyed 70% of Yellow cameras before they could be retrieved and were therefore not included in this analysis (No cameras were deployed at Red sites in 2019).

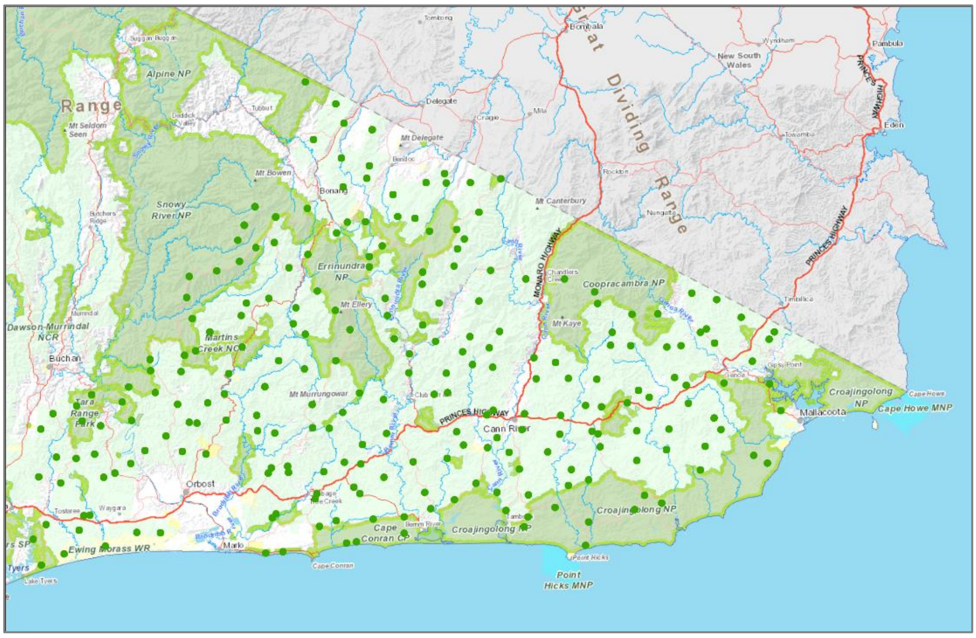

Figure 4. Pre-fire locations of Green camera sites in October 2019

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[2] Cameras were not deployed in 11 other cells due to access issues.

In 2020, cameras were deployed in only 85 cells at the 3 camera sites (Green – April 2020, Yellow – May 2020, Red – June 2020) that in 2016–17 and 2019 had detected Long-footed Potoroos (Figure 5).

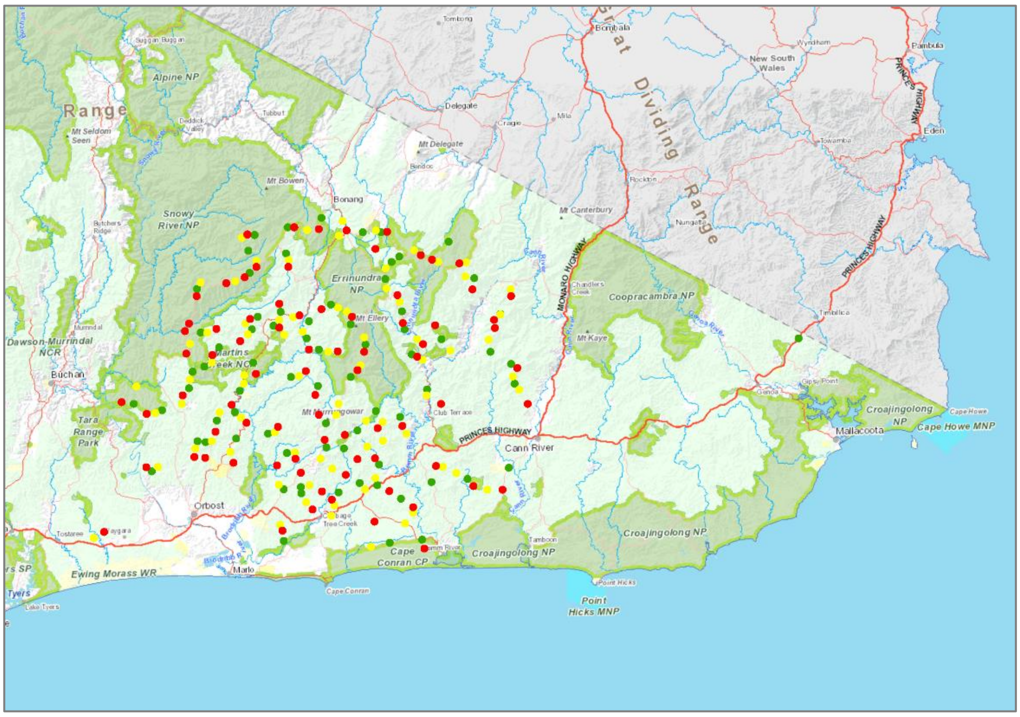

Figure 5. Location of Southern Ark cameras surveyed in April (Green), May (Yellow) and June (Red) 2020 at sites where Long-footed Potoroos had been detected in 2016–17 and 2019

2.3 Image management and species identification

Camera images were inspected and those with identifiable species were tagged using the image management software ExifPro Image Viewer 2.1 (Kowalski 2013). A full description of the methodology can be found in Bluff (2016). Briefly, camera station folders were created, and all images were copied from the SD cards to the corresponding camera station folder. A backup copy of all images was also made. Images were then tagged using Exifpro. Each tag is a 3-letter code (see Appendix 1 in Bluff 2016) for a species ID or other important information related to an individual image. Importantly, tags are stored in the EXIF data within each JPEG file. If the species identity was unclear, an expert reviewed the images and tagged them appropriately. Images that could not be assigned a species tag were identified by a specific 3-letter code.

2.4 Data analysis

2.4.1 Environmental and ecological covariates

The data used in this analysis comprised (a) covariate data for various environmental and ecological variables (mean temperature, annual precipitation, seasonal precipitation, landscape fire severity, site and location) for given sites and for given time periods (years); (b) species presence data, which was used in the estimation of detection probability; and (c) presence–absence data for each of the species across sites and over time.

For the landscape variables of fire severity, we used a distance and severity weighted calculation adapted from Lindenmayer et al. (2021). This method allows for the effects of disturbance at the camera-trap location and surrounding area (250 m radius) to be included as a continuous variable, reducing model complexity and degrees of freedom. Because the fire severity data is composed of 5 severity classes related to metrics like the percentage of crown scorch (DELWP 2021), the fire severity at a site was weighted according to: (a) the proximity of the site to a given burnt grid cell; and (b) the severity of fire in that grid cell. This method allows for a smoothing of fire severity scores around a site, which leads to higher landscape fire severity scores for sites with severe burns at and surrounding the site. However, it also reduces the fire severity score of a site that is severely burnt but is adjacent to areas that were not as badly burnt and may have acted as fire refugia. The calculation of the fire severity score weighting for each raster pixel within a 250 m radius of the site follows the following gaussian kernel estimate:

$$site\ fire\ \ severity = \sum_{i=1}^{n} e^{-\phi_1 d_n^2} \times \left(1 - e^{-\phi_2 s_n^2}\right)$$

Where:

- ϕ_1 is the distance scale parameter (73). The distance scale parameter was chosen by considering the distances between camera stations (at least 1 km) and the home ranges of the species being investigated. Given that multiple species were being assessed, we chose a conservative (high) scale parameter, meaning the fire rating at a site closely reflects the severity of fire at the centre of the site (camera location). For instance, the scale parameter of 73 ensures that the weighting of a raster cell 100 m from the camera is weighted approximately half the amount of a raster at the camera; after 200 m this weighting reduces to approximately 0.05. Thus, scores from this calculation will strongly reflect the fire severity at the camera location with some impact of the immediate neighbouring area. A visual representation of this weighting is available in Appendix 1
- d_n is the distance of the raster cell n from the site (in km)
- ϕ_2 is the fire severity scale parameter (0.0865). The fire severity scale parameter was selected so that a high severity fire (class 5: >80% canopy scorch) at a camera location had approximately a 10-fold weighting compared with a largely unburnt section (<10% canopy burn). Thus, this scale parameter roughly reflects the percentage of canopy burnt.
- s_n is the fire severity of the raster cell n.

Bioclimatic variables were obtained for each camera site from CHELSA (Climatologies at high resolution for the earth’s land surface areas) (Karger et al. 2017). A temperature-based variable (BIO01) and 2 precipitation-based variables (BIO12 and BIO15) were used. These variables represent annual mean temperature (BIO01), annual precipitation (BIO12) and precipitation seasonality (BIO15).

2.4.2 Occupancy models

To investigate the impact of fire and predation on native species occupancy, we used a Bayesian approach to model species occupancy (Kellner 2021) while accounting for imperfect detection. This approach involved determining the probability that a species will be present at some sites but will go undetected, and then explicitly incorporating that probability into the overall presence estimates. Occupancy models for both predator and native species are composed of 2 interacting submodels. A ‘detection’ submodel estimates the probability of detecting a species at any given site (provided they are actually present). This probability of detection (p) is then used in combination with a ‘observation’ submodel that incorporates variables affecting the occupancy of the species, which ultimately allows for the estimation of the true state of occupancy at a given site. The mathematical expression of the zero-inflated binomial models can be expressed as:

$$z_i \sim Bernoulli(\psi_i)$$

$$y_{ij} \mid z_i \sim Bernoulli(z_i \times p_{ij})$$

Where z_i is the true occupancy state of site i based on the probability of occupancy (ψ). Therefore, the detection of a species at a given site during an observation period (y_{ij}) is dependent upon the occupancy state at the site and the probability of occupancy at that site and observation period (p_{ij}).

Using the ubms R package (Kellner 2021), we fitted a ‘stacked’ model, which is a single-season occupancy model with pseudoreplication (i.e. sites being repeated in the data over years). The effects of pseudoreplication were controlled by having group (n = 240) and season (n = 3) as random variables. This stacked approach was chosen over a multi-season dynamic occupancy model (MacKenzie et al. 2002, 2003) mainly due to data limitations. Specifically, given the inconsistency in measuring occupancy at sites over the study period (partly due to fire destroying cameras), only some of the data would have more than 2 seasons’ worth of records. Therefore, a multi-season occupancy model would allow only a subset of the data from the Southern Ark study time and area to be used; the stacked approach used here utilises all available data.

We generated 2 classes of occupancy models: a predator occupancy model (with interaction terms for feral cats, foxes and wild dogs) and a native species occupancy model (with interaction terms for Common Brush-tailed Possums, Lace Monitors, Long-footed Potoroos, Long-nosed Bandicoots and Long-nosed Potoroos). First, we generated predictions of occupancy for feral cats and wild dogs that were recorded in 2016–17, 2019 and 2020. Second, for foxes, we used a more informative, spatially explicit, individually based population model (FoxNet; Hradsky et al. 2019b) of predicted fox densities across Southern Ark (Francis et al. 2019). ‘FoxNet’ incorporates parameters on dispersal and home ranges as a function of resource availability, and customised survival and reproductive parameters. In essence, these variables are both proxy measures for the effect of fox predation on native species, but fox density is more informative than fox occupancy. The final selected covariates in the predator occupancy model were determined using model selection (discussed below). The best-fitting predator model then had predictions generated for feral cats and wild dog at each site in each year. These predictions were used as covariates in the native species model (Figure 5).

Model selection

We conducted model selection on the observation submodels. For efficient (fast) model selection we used generalised linear mixed models (GLMMs), with combinations of covariates and the occupancy data condensed into presence (1) or absence (0). We selected the best model – the model with the lowest Akaike Information Criterion (AIC) – using the MuMIn R package (Bates et al. 2015) and incorporated this with the predetermined detection submodel in a Bayesian occupancy model to account for imperfect detection rates.

Occupancy model structure

All model covariates were fixed with a species interaction effect (Species) so that the impacts of the covariates could be investigated at the species-specific level. In the predator species occupancy model, the best model included interaction terms for species with mean temperature, annual precipitation and precipitation seasonality. For native species, we used the same occupancy modelling approach and included interaction terms for species and mean temperature, annual precipitation and predator presence.

Because stations were deployed in clusters of 3, we used this clustering (Group) as a random effect, because the inclusion of an individual station variable resulted in poor model convergence. The Season (2016–17, 2019 or 2020) was included as a second random effect.

In both predator and native species models, our detection submodel included an interaction between the lure type (predator lure or native mammal lure) and days since deployment (1 to 35). Species was also included as a fixed effect in the detection submodel because there was evidence that detectability varied between species and that species existed at different densities (which meant that the frequency of observations would differ even when the species was not present). For the native species model, an interaction term was fixed between species and daily maximum temperature (NOAA PSL 2021). Figure 6 is a diagrammatic representation of the occupancy modelling approach used in the analysis.

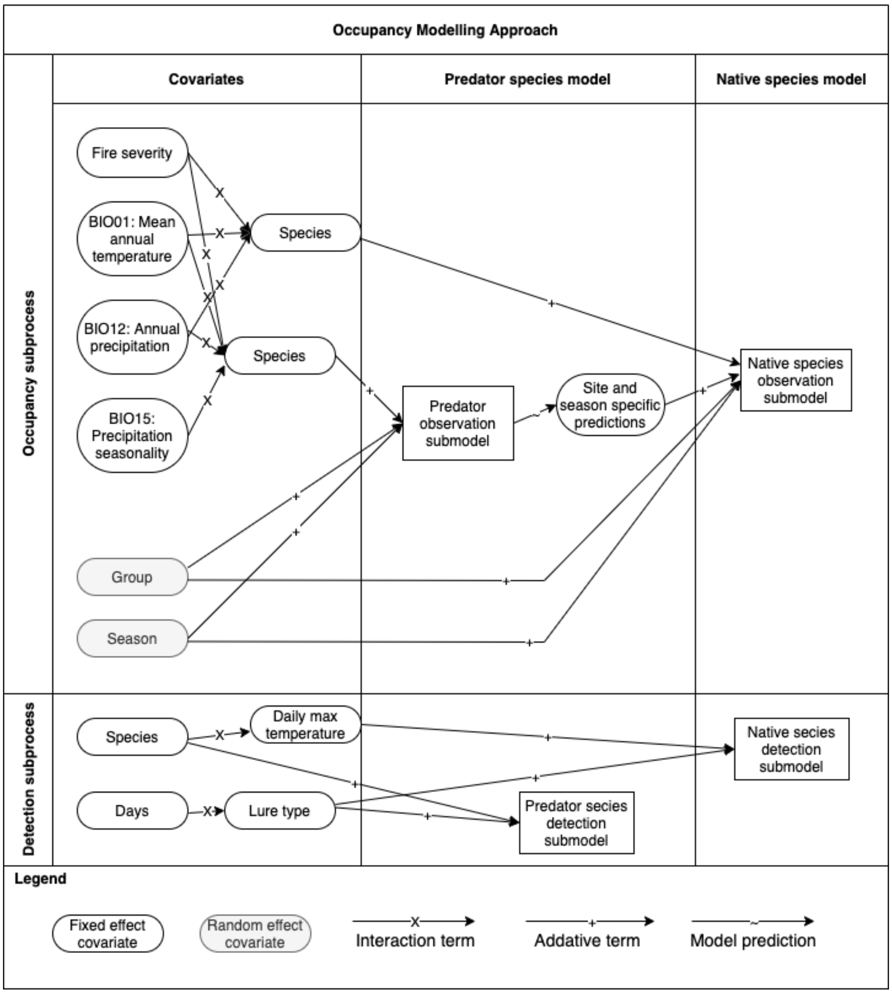

Figure 6. Diagrammatic representation of the modelling approach used to assess occupancy. GLMM was used to assess the best covariates (lowest AIC) to use in the Bayesian occupancy model for each native species

Model fit and diagnostics

We ran Bayesian occupancy models in parallel across 4 chains for 4,000 iterations. We investigated the model fit of our occupancy models through several diagnostic and posterior predictive tests to assess model convergence, mixing of chains, model fits and accurate posterior predictive power (see the Supplementary material for extensive checks and relevant R code). Overall, models were considered appropriately converged if all parameters had effective sample sizes above 200 and Rhat values <1.05. Model fits of the top-ranked GLMM were assessed using the area under the curve (AUC) of the response to the operating curve (ROC). Bayesian posterior predictive tests assessed the occupancy submodel by comparing the true proportion of stations that detected each species against the proportion of stations that detected each species when the posterior states (presence or absence) were simulated 250 times. The detection submodel was assessed using posterior predictive tests that compared the true and simulated values for the proportion of days during deployment where species were recorded. Additionally, goodness-of-fit tests using the Mackenzie–Bailey chi-squared test was used to assess the model's ability to predict the true observation and occupancy patterns (MacKenzie and Bailey 2004).

We investigated possible reasons for Long-footed Potoroo distribution patterns, based on the models, by conducting a post hoc analysis on the model predictions. We compared whether predicted occupancy rates in fire-affected areas might be lower for some species or at some stations due to the likelihood of fire refugia or landscape topography that may have allowed greater persistence post-fire. We did this by determining the standardised mean change (Cohen's *d*) in occupancy at sites before and after fire (Cohen 2013) against the terrain ruggedness index, which is a measure of the variability (ruggedness) of the elevation surrounding a given point (Riley et al.1999).

3 Results

3.1 Species detections across years

A total of 10 species were identified from the camera traps in all 3 survey periods (2016–17, 2019 and 2020; Appendix 2). Eight were native species (Common Brush-tailed Possum, Eastern Ring-tailed Possum, Lace Monitor, Long-footed Potoroo, Long-nosed Bandicoot, Long-nosed Potoroo, Southern Brown Bandicoot, and wild dog). Two were introduced predators (feral cat, fox).

In 2019, 27 sites detected a potoroo species (Potoroo sp. in Appendix 2). Because those records could not be identified to a species level, the presence of Long-nosed Potoroo and Long-footed Potoroo during that season may be biased in that records are underestimated. To correct this, the ‘Potoroo sp.’ records were reclassed as either Long-nosed Potoroo or Long-footed Potoroo based on (a) previous site detections of either species or, if no records were recorded in 2016–17, (b) the mean detection rates of Long-footed Potoroo versus Long-nosed Potoroo across all sites (see the Supplementary material for the R code used to perform this imputation). There was insufficient data to include Eastern Ring-tailed Possum and Southern Brown Bandicoot in any further analysis.

3.2 Change in occupancy

3.2.1 Effect of the 2019–20 bushfires on native species site occupancy

Overall, there was little evidence of changes in mean site occupancy following the 2019–20 bushfires. There was some weak evidence that Lace Monitor and Long-nosed Bandicoot experienced a decline, while Long-footed Potoroo and Long-nosed Potoroo showed very little difference (Table 1). Common Brush-tailed Possum showed a very slight (negligible) increase in occupancy when comparing pre-fire and post-fire data. However, the 95% confidence intervals on the change in occupancy overlapped zero for all these predictions.

Table 1. Change in native species occupancy for sites surveyed pre-fire and post-fire.

Species	Mean occupancy pre-fire [95% CI][1]	Mean occupancy post-fire [95% CI][1]	Proportional change in occupancy [95% CI]
Common Brush-tailed Possum	0.21 [0.03, 0.57]	0.26 [0.04, 0.65]	0.04 [−0.03, 0.15]
Lace Monitors	0.24 [0.01, 0.59]	0.02 [0.00, 0.48]	−0.18 [−0.48, 0.05]
Long-footed Potoroo	0.55 [0.13, 0.9]	0.49 [0.11, 0.92]	−0.04 [−0.27, 0.21]
Long-nosed Bandicoot	0.24 [0.07, 0.5]	0.13 [0.02, 0.44]	−0.1 [−0.3, 0.12]
Long-nosed Potoroo	0.02 [0, 0.1]	0.01 [0.00, 0.08]	−0.01 [−0.07, 0.02]

[1]95% confidence intervals (CIs) are taken from the 2.5% and 97.5% percentiles of posterior occupancy predictions for the 236 sites that had pre-fire and post-fire surveys.

3.2.2 Effect of fire severity, predators and other covariates on native species site occupancy

Native species post-fire occupancy in 2020 was influenced by the severity of the fire in the surrounding landscape: the magnitude of the fire severity significantly (95% CIs not overlapping zero) influenced occupancy for 4 of the native species (Figure 7c). Lace Monitors had the largest estimated change with fire severity (while accounting for temperature differences between monitoring seasons), followed by the Long-nosed Potoroo, Long-nosed Bandicoot and Long-footed Potoroo. Appendix 3 provides the model estimates showing the effects of all the covariates on the probability of occupancy (logit-scale) for native species.

Effect of introduced predators on native species occupancy

The Long-footed Potoroo had a strong negative relationship with predicted fox density (Figure 7d). There was a sharp decline in Long-footed Potoroo occupancy when fox density was greater than zero; the predicted occupancy was estimated at zero when fox densities were above 0.5/km^2. For all other native species, relationships between occupancy and fox density had 95% CI overlapping zero (Figure 7d and Appendix 3). There was no relationship between Long-footed Potoroo occupancy and feral cat occupancy (non-overlapping 95%CI, Figure 7b). However, this relationship is problematic because both feral cats and Long-footed Potoroos tend to occupy areas with lower mean annual temperatures (Figure 7e and Appendix 5). It is also possible that foxes suppress feral cats, although fox density is considered low across the study area. Neither feral cat (Figure 7b) nor wild dogs (Figure 7f) were predicted to have any influence on native species occupancy.

The effect of other covariates on native species site occupancy

Annual precipitation influenced site occupancy for Common Brush-tailed Possum (decreased with lower levels of precipitation and Long-nosed Potoroo (increased with higher levels of precipitation) (Figure 7a).

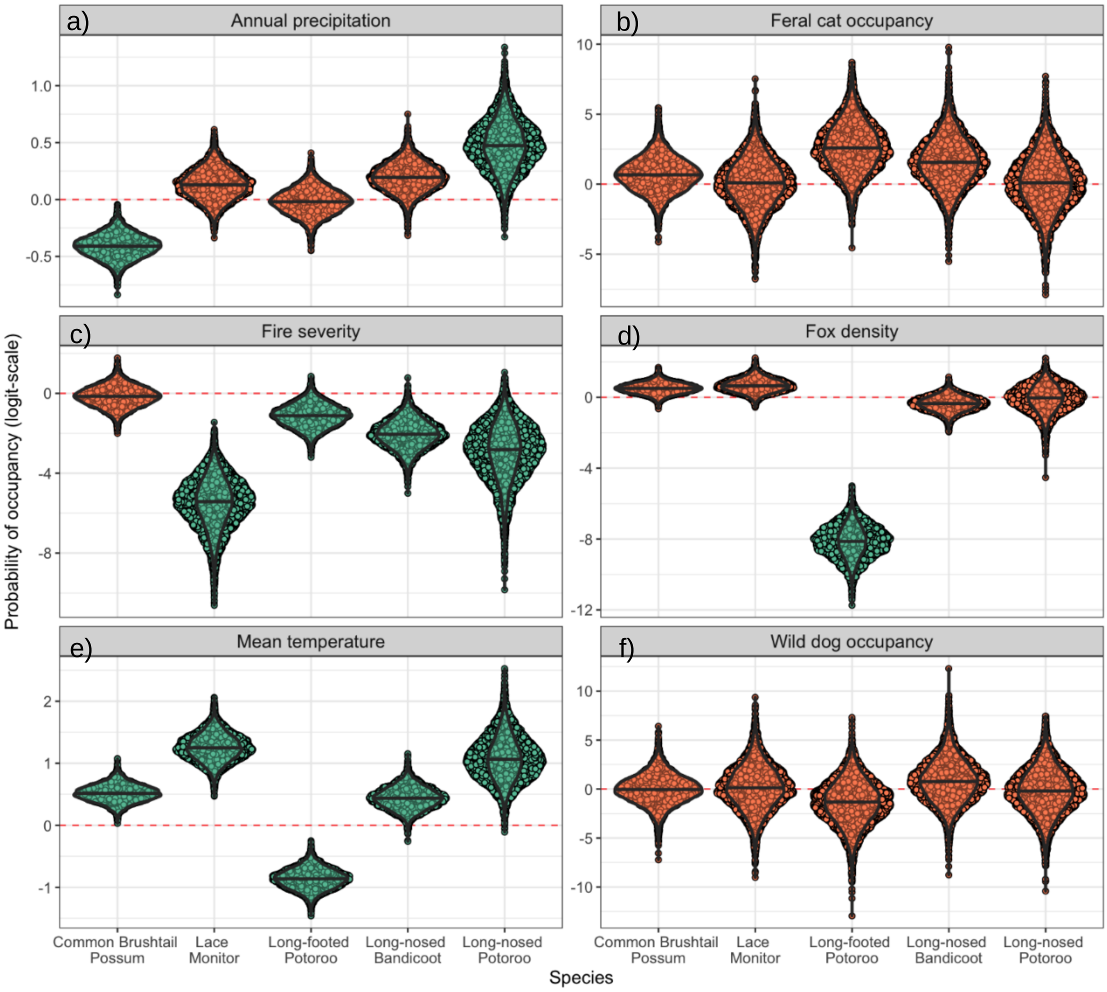

Figure 7. Conditional effects from model estimates showing the effects of various covariates on occupancy (logit-scale) for native species. Green – 95% CIs not overlapping zero, orange – 95% CIs overlapping zero. Horizontal line is the median value, tips cover the whole range of data. Spread in points shows predictive certainty; a wider spread indicates a less certain prediction

The impact of fire severity on occupancy varied spatially for some species (Figure 8). For example, sites with high rates of predicted occupancy for Long-footed Potoroo were in areas to the north that were not impacted by fire. This suggests that, although they can survive fire, Long-footed Potoroos fare better at sites not impacted by fire. Long-nosed Bandicoots appeared to have had a stronger negative response to fire severity, as they fared better in unburnt (or low-severity) sites in southern and northern pockets of the study area. The terrain ruggedness index – which may be a proxy for fire refugia (Krawchuk et al. 2016) – had no influence on occupancy for any species and is therefore not considered further.

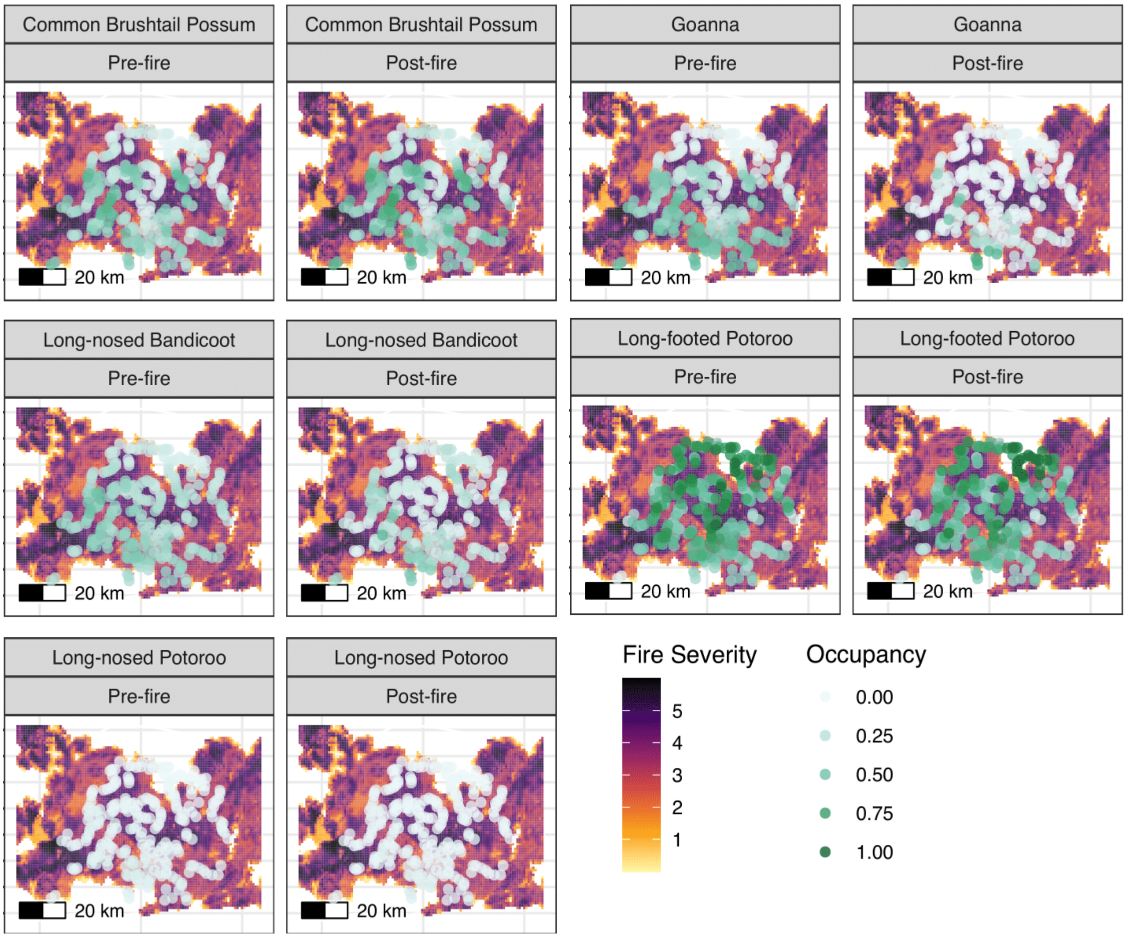

Figure 8. The spatial impact of the 2019–20 bushfires on pre-fire and post-fire occupancy of native species

3.2.3 Effect of the 2019–20 bushfires and other covariates on predator occupancy

There is moderate evidence that wild dog and feral cat mean estimated occupancy was impacted by fire intensity (Table 2) and fire severity (Figure 9 and Appendix 4). However, this trend is somewhat uncertain depending on whether we compare occupancy across high and low fire intensity percentiles (Table 2) or the continuous fire severity gradient (Figure 9 and Appendix 4), which shows that the upper 95% CIs for wild dogs just overlaps zero (0.098). There is no evidence that foxes were impacted by the 2019–20 fires.

Table 2. Difference in predator species occupancy at low and high fire severity impacted sites in 2020.

Species	Mean occupancy low fire severity [95% CI]	Mean occupancy high fire severity [95% CI]	Proportional change in occupancy [95% CI]
Feral cat	0.45 [0.36, 0.58]	0.33 [0.22, 0.46]	−0.12 [-0.25, 0.00]
Fox	0.07 [0.05, 0.13]	0.06 [0.02, 0.12]	−0.02 [-0.06, 0.03]
Wild dog	0.11 [0.06, 0.22]	0.05 [0.01, 0.15]	−0.06 [−0.14, −0.01]

The effect of mean temperature and annual and seasonal precipitation on predator occupancy is also given in Figure 9 (and Appendix 4). Feral cat occupancy was affected by mean temperature, with lower rates of occupancy under colder conditions. The same effect was found for the fox. Fox occupancy also declined with declines in annual precipitation and was predicted to be higher with increases in seasonality of precipitation.

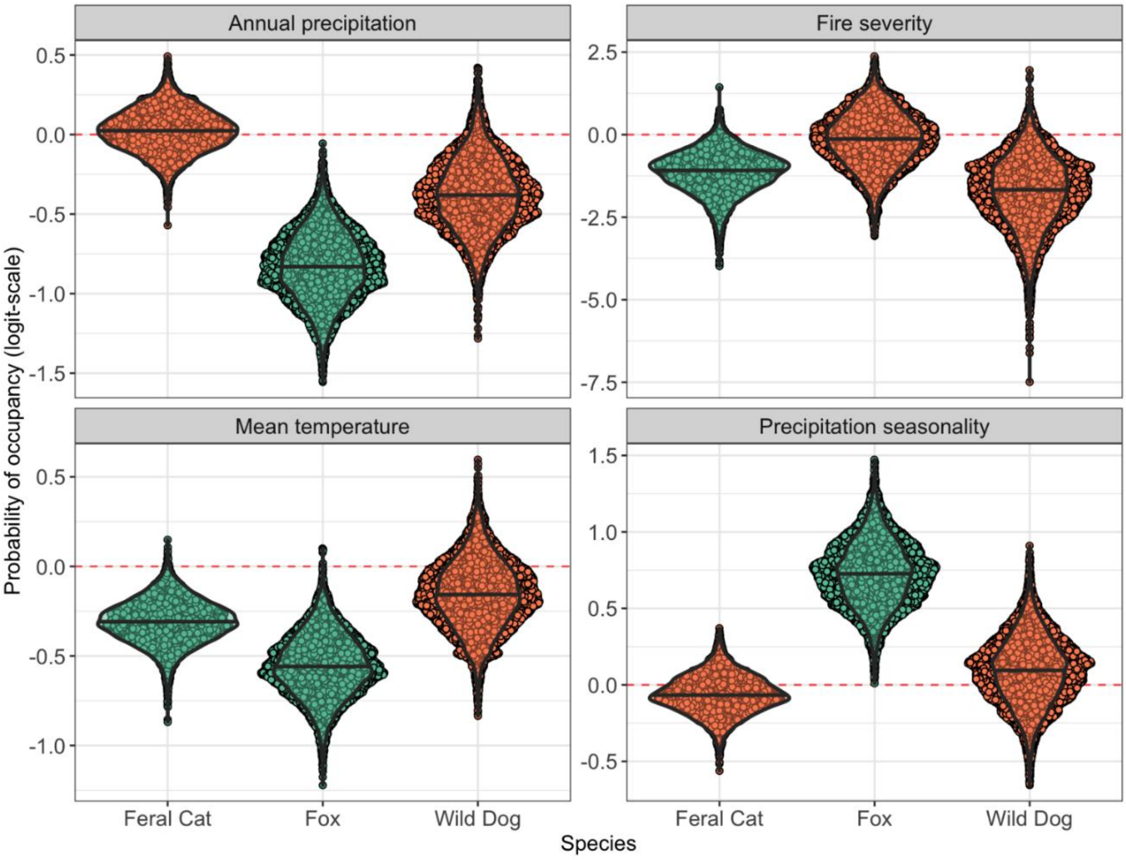

Figure 9. Conditional effects from model estimates showing the effects of various covariates on occupancy (logit-scale) for predator species. Green – 95% CIs not overlapping zero (red dashed line), orange – 95% CIs overlapping zero. Horizontal line is median value, tips cover the whole range of data. Spread in points shows predictive certainty; a wider spread indicates a less certain prediction

3.3 Detection probabilities for predators and native mammals

3.3.1 Attractiveness of lure type

Detection probabilities varied for predator species and native mammal species. Both the type of lure used and the number of days since the lure was put in place predicted the detection probability for a given species.

Predators were initially detected at higher rates by the predator lure (e.g. Day 1 feral cat detection predator lure = 0.043 [95% CI = 0.032, 0.057], Day 1 feral cat detection native lure = 0.018 [95% CI = 0.014, 0.022]). However, the detection probability for the predator lure decreased over time, with predators being detected by both lure types at similar rates after 25–30 days (Figure 10).

For the native species, we found (a) an effect of lure type, (b) a decrease in detection over time and (c) no discernible interaction between lure type and days on detection probability. Specifically, non-predator lures were more effective at detecting native species (e.g. Day 1 Long-footed Potoroo detection native species lure = 0.202 [95% CI = 0.191, 0.212], Day 1 Long-footed Potoroo detection predator lure = 0.131 [95% CI = 0.114, 0.15]), with the probability of detection reducing over time for both lures (Figure 10). The conditional effects (on a logit scale) are presented for daily maximum temperature and days since deployment with lure type in Appendix 5.

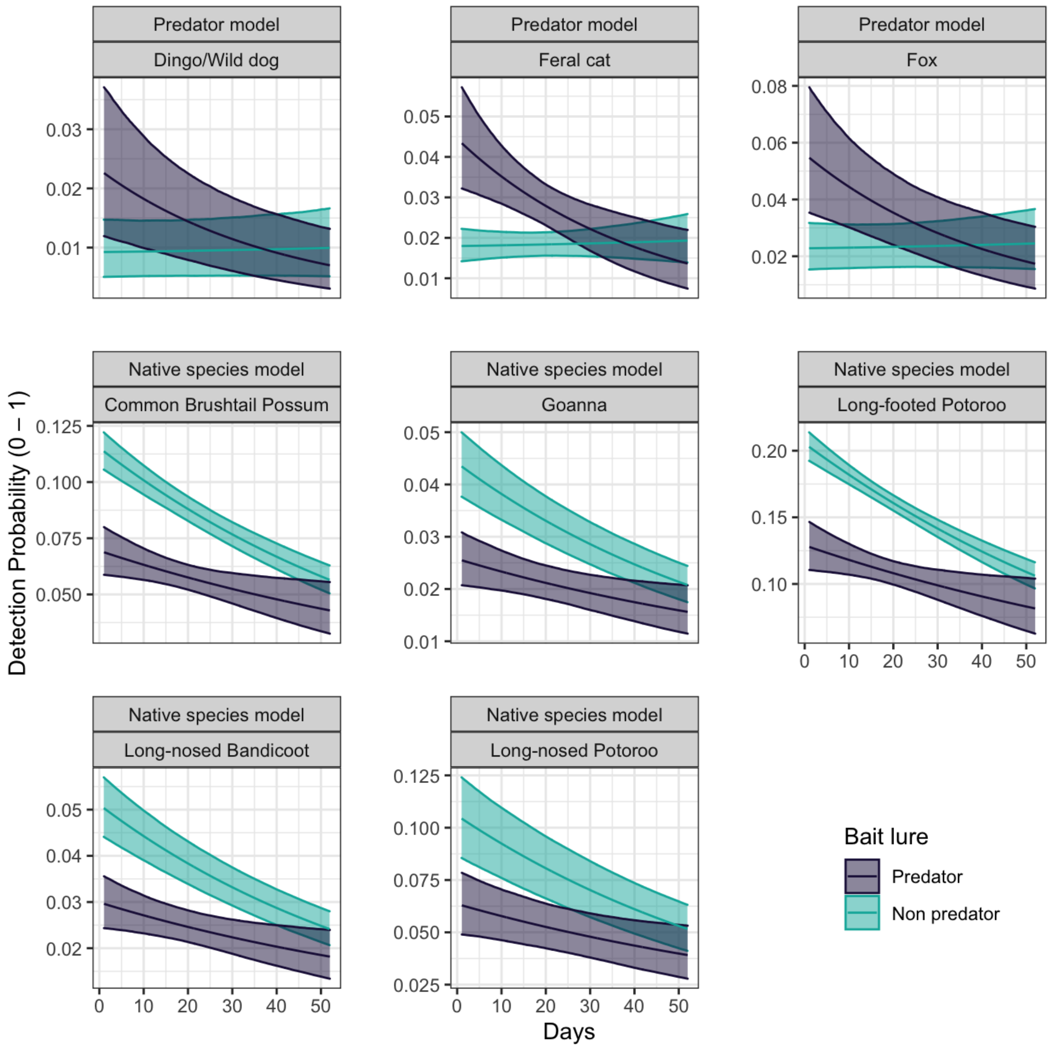

Figure 10. The estimated detection probabilities for the different type of lure used (predator versus non-predator) as well as the number of days since the lure was set up

3.3.2 Daily maximum temperature impacts detection rates

We found daily maximum temperature had a positive association with the probability of detection for Lace Monitors, Long-nosed Bandicoots and Long-nosed Potoroos (Figure 11). Conversely a strong negative relationship was observed for Long-footed Potoroos, and a weak negative relationship for Common Brush-tailed Possums. The conditional effects (on a logit-scale) are presented for all species in Appendix 5.

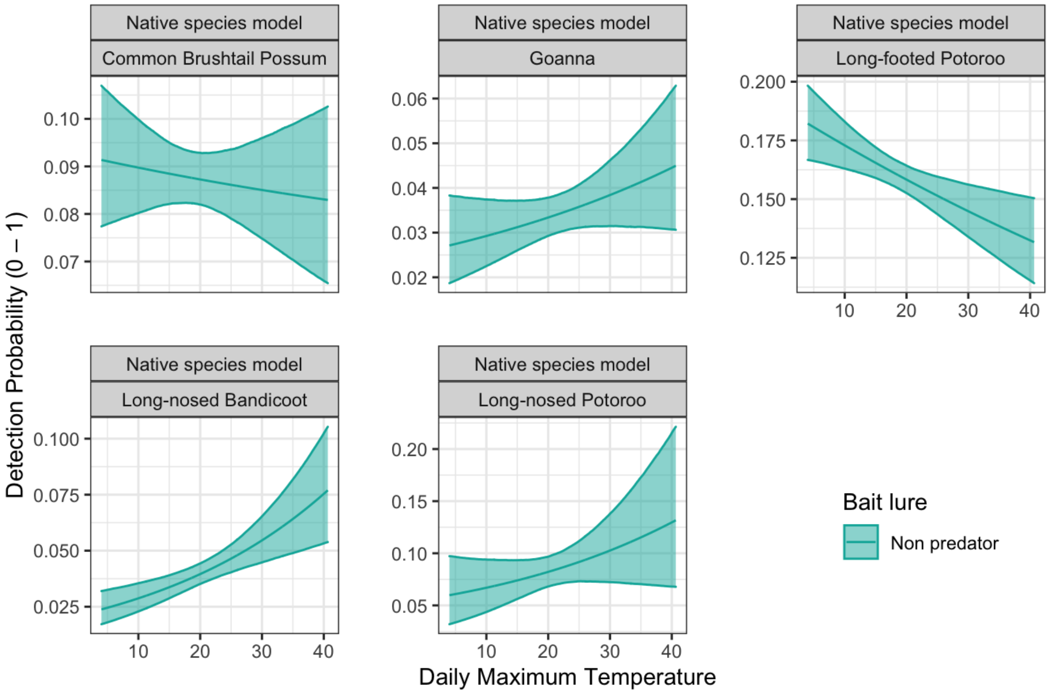

Figure 11. Estimated detection probabilities of native species according to daily maximum temperature. The middle line indicates the median estimate, and the shading represents the 95% CIs

3.3.3 Cumulative detection rates

The cumulative detection rate varied for native species, meaning adequate deployment durations would have varying lengths depending on the target species. We found that, across all species, a 75% probability that a detection will occur (if the site is truly occupied) within the maximum deployment duration used in this study (52 days). However, a high cumulative detection probability (> 90%) was reached quicker for Long-footed Potoroos (11 days), Common Brush-tailed Possums (23 days) and Long-nosed Potoroos (25 days). Cumulative detection rates are displayed in Appendix 6, noting that these estimates assume a fixed mean maximum temperature (18.8°C) and are specific to non-predator lures.

3.4 Model fit and diagnostics

There was strong evidence to suggest both Bayesian models reached convergence. In both models, all parameters achieved an effective sample size (ESS) above 200 (apart from the random effect for group in the predator model where ESS = 40). However, there was strong evidence of chain mixing and convergence of all parameters. Specifically, residual tests from Wright et al. (2019) – where residuals are calculated independently for the observation and detection submodels – showed expected patterns of residual distribution (see Supplementary material for relevant R code and plots).

The predictive power of the predator and native species models were assessed via posterior predictive checks. We found that models accurately predicted: (a) the number of stations at which each species were detected; and (b) the average number of times each species was detected at each station. This suggests that at least on a broad scale, models fit and predictive power is high. Additionally, the AUC for the GLMM of the occupancy submodel (used in model selection) was relatively high (83.9 % [95% CI = 82.8%, 85.1%]). We sought to implement a chi-squared test to assess the congruity of the true detection patterns against simulated detection patterns (MacKenzie and Bailey 2004). The initial results from this test suggested a large degree of disparity between the true observations and model simulations. However, based on inspections of individual chi-squared values, it is very likely that this disparity is driven by temporal correlation in detections (e.g. detections are often clustered in adjacent nights); this is a known limitation of the Mackenzie–Bailey chi-squared test (Wright et al. 2016). Details of posterior predictive checks and relevant R code are included in the Supplementary material.

4 Discussion

Overall, there was only weak evidence that mean site occupancy for all the assessed native species and one of the two introduced predator species differed from before to after the 2019–20 bushfires. However, the severity of fire in the surrounding landscape did influence site occupancy for 4 of the 5 native species and feral cats in 2020. The exception was Common Brush-tailed Possums, with a slight positive change in occupancy. Mean occupancy rates differed (non-overlapping 95% CIs) for Lace Monitors, which showed the largest estimated impact, followed by Long-nosed Potoroos and Long-nosed Bandicoots, with weaker evidence that severity affected Long-footed Potoroos. There was moderate evidence that feral cats were impacted by fire severity while there was no detectable difference for wild dogs or foxes. Other variables that influenced occupancy included: fox density for Long-footed Potoroo; mean temperature for all 5 native species, feral cats and foxes; and annual precipitation for Common Brush-tailed Possums, Long-nosed Potoroos and foxes.

The Southern Ark camera monitoring program was not designed to investigate the impact of fire on native species. However, despite varying spatial and temporal sampling effort, there was enough data to investigate the differences in occupancy of Long-footed Potoroos before and after the 2019–20 bushfires at sites where they were previously detected. For other species, such as the Long-nosed Potoroo, the post-fire sampling design limits the ability to estimate the role of predation and other environmental variables and to make comparisons more broadly across the Southern Ark area. Also, the data as collected only allows for estimates of declines in occupancy and not increases in the distribution of species. If a more comprehensive assessment of the role fire and predation might play is desirable for a broader range of species, ideally all 720 camera sites should be repeat measured (including sites where species of interest have not been detected) and across a range of fire severity and predator densities.

A similar study, undertaken by Lumsden et al. (in press) on the Great Dividing Range population of Long-footed Potoroos in 2004–09, found that potoroos were able to survive severe bushfire but were positively associated with unburnt or lightly scorched sites compared with more severely burnt sites. This supports our finding that Long-footed Potoroo occupancy tended to diminish as fire severity increased. Lumsden et al. (in press) also found that potoroos were strongly negatively associated with the occurrence of foxes, and this was a key factor influencing the species' distribution and habitat associations. The results from the current study indicating that a low density of foxes can significantly influence the occurrence of Long-footed Potoroos is supported by the results of the Lumsden et al. (in press).

However, apparent associations between fire severity and the occurrence of potoroos need to be interpreted with caution, for two reasons. Firstly, as the survey sites were selected based on known records of potoroo occurrence pre-fire, rather than each fire severity category, they may not be representative of fire severities occurring in the whole fire-affected region of the Southern Ark area. Secondly, it is possible that the occurrence of potoroo observations in unburnt habitats may reflect the propensity of different habitat types to burning (e.g. gully versus ridge) as much as any preference of potoroos for unburnt habitats *per se.*

Lumsden et al. (in press) found that resolving the Long-footed Potoroo's relationships with vegetation type was complicated by the fact that foxes were negatively associated with potoroo primary habitat, making it difficult to separate the causal effects of foxes and vegetation type on occurrence of Long-footed Potoroos. Previous studies on the drivers on occurrence of potoroos and bandicoots were undertaken in areas with no history of long-term, ongoing fox control. Earlier research indicated that the Long-footed Potoroo's niche is dependent on moist soils and aspects sheltered from fire, likely due to the species' relationship with hypogeal fungi as a primary food source (DSE 2009), with year-round fungal abundance and diversity likely a key requirement for the distribution of this species (Wauchope-Drumm 2020). By their nature (often in cooler, wet positions in the landscape, e.g. gullies, south-facing aspect), these sites may provide a natural refuge from the impact of fire and may be less favoured by foxes. Also, potoroos that survive fire may take advantage of an immediate post-fire flush of fungi that can occur in some situations (Claridge and Trappe 2004) and therefore be detected foraging in burnt areas, which may explain some of the records in the more severely burnt areas in this study.

In the absence of foxes, or at least where fox densities are low, it is possible that Long-footed Potoroos could become much more widespread and occupy a wider range of habitats (Lumsden et al. in press). This proposition, and the findings that Long-footed Potoroos can survive severe fire, are supported by results from camera surveys undertaken at Tulloch-Ard State Forest since 2011 (Robley et al. in press). Fox control commenced in June 2011 and camera monitoring began in November of that year, and was undertaken in 2012–13, 2016, 2018 and 2020. In the first year, no Long-footed Potoroos were detected at the 60 monitoring sites, increasing to them being detected at 60% (n = 36) of sites in 2020. The presence of foxes was low throughout the survey period and foxes were excluded from the modelling as they were detected only 12 times across all camera sites and years. Despite 88% of the survey area being impacted in 2019 by fire that removed understorey, there was no major change in the proportion of sites occupied by Long-footed Potoroos (Robley et al. in press).

Our results of declines in Lace Monitors, Long-nosed Potoroos and Long-nosed Bandicoots suggest that fire has had an important role in this shift, keeping in mind the limitations of the post-fire sampling design. This is supported by previous findings on the interaction between fire and the role of predators in shaping faunal communities. For example, at 122 sites in East Gippsland, Robley et al. (2013) investigated how native species at risk from fox predation are distributed in relation to the landscape patterns created by different fire histories and the presence or absence of fox control. They found that vegetation cover was a significant predictor for 7 of the 12 species included in their models, particularly Long-nosed Bandicoot and Long-nosed Potoroo, and that the presence of fox control was significant for Lace Monitor and Long-nosed Bandicoot.

In south-western Victoria, Smith (2013) found Long-nosed Potoroos present across a wide range of fire histories where there was intensive fox control. She concluded that it is likely that large, unburnt patches of vegetation play an important role in survival immediately following a fire event, providing refuge from predators until vegetation that provides cover from predators returns. In tall mixed forest in south-western Victoria from 2013 to 2015, Robley et al. (2016), using a replicated before–after design, investigated the combined impacts of fox predation and planned burning on native mammals. Southern Brown Bandicoot occupancy decreased from 38% of sites pre-fire to 9% post-fire. Fox impact apparently increased following the burns, with Southern Brown Bandicoots increasing in the fox diet by 33% immediately post-fire, even though bandicoot occupancy rates declined at the same time, and then declined in the fox diet by 50% 1 year later. Long-nosed Potoroos were not detected in the diet in the immediate post-burn period but increased by 56% from the pre-burn period to 1 year post-burn despite being undetected in camera surveys immediately after the burn.

Hradsky et al. (2019a) used a before–after, control–impact experiment to investigate the short-term effects of a prescribed fire on foxes and feral cats and their native mammalian prey in the Otway Ranges. Fire reduced understorey cover by more than 80% and resulted in a 5-fold increase in the occurrence of invasive predators. Concurrently, relative consumption of medium-sized native mammals by foxes doubled, and predation of Long-nosed Bandicoots and Short-beaked Echidnas (*Tachyglossus aculeatus*) by foxes increased.

Varanid lizards are key functional components in Australian predator guilds, and could benefit, via ecological release, when introduced predators are managed successfully. Yang et al. (2019) examined the difference in abundance of Lace Monitors in the Southern Ark area between a single non-baited site and one that had been baited for several years. They found that Lace Monitors had significantly higher abundances in an area baited for foxes relative to a non-baited area. This supports the experimental and circumstantial evidence for mesopredator release of Sand Goanna (*Varanus gouldii*) and about 5 other medium-to-large species of varanid lizard following fox and feral cat management in Western Australia (Sutherland et al. 2011).

Our result of a significant reduction in the occurrence of Lace Monitors post-fire suggests that this species is at risk from severe fire. While the modelling we have undertaken accounts for the effect of temperature on Lace Monitors, it is possible that this is underestimated if the relationship between temperature and activity is non-linear; for example, if there is a threshold temperature at which Lace Monitors significantly decrease activity and are therefore not as detectable.

Few studies in Australia contain data on varanid responses to fire. Varanid sample sizes have been generally small, presented as part of a broader reptile dataset, and rarely a key focus of the study, and are dominated by studies in semi-arid or arid systems (Masters 1996; Schlesinger et al.1997; Lentic et al. 2004;

Woinarski et al. 2004a; Mott et al. 2010; Pianka and Goodyear 2012; Bird, et al. 2013). As a result, the outcomes for the impact of fire on varanids are inconclusive. Woinarski et al. (2004a) reported on the response of terrestrial vertebrate fauna to fire over a 6-year period. Although they recorded a decline in Spotted Tree Monitor (*Varanus timoriensis*), they noted that their study had a limited sample size which hampered precise estimation of the change in abundance. There is a need to improve our understanding of the role fire and predation play in structuring large reptile communities; repeating post-fire surveys under the Southern Ark monitoring program will help to address this knowledge gap.

We found that detection rates of native species and predators differed according to the type of lure used at camera sites and the number of days those lures were applied. Native species were less likely to be detected using a predator lure than a mammal lure. Predators were less likely to be detected on a mammal lure than a predator lure initially, but over time this difference reduced. The attractiveness of both lure types diminished over time. Maximising detection rates at static points (camera traps, bait stations, leg-holds or confinement traps) can enhance the quality of data collected and the efficacy of predator management actions. There are some ethical issues to be considered in luring predators and native species to the same point: are we facilitating predation by attracting both predator and prey? In this study, by luring predators after native species and using a different lure type, the risk of enhancing or facilitating predation was minimised. There were no images recorded of any direct predator–prey interactions at camera sites over about 25,200 camera-trap nights. The duration of attractiveness of lures varied with species, but lures were substantially less effective after 25–30 days. Further consideration should be given to the rate at which detections accumulate and at what point this new data becomes marginal. There may be operational and logistical reasons why a shorter deployment would be useful; for example, to allow a limited pool of cameras to be deployed at a greater number of sites for a shorter period.

Apart from the relationship between fox density and Long-footed Potoroo occurrence and fire severity on Lace Monitor occurrence, the confidence intervals for the remaining relationships are generally wide. This is mostly a result of a large sampling error, which in turn is related to the sample size and variation in the (statistical) population. There is little that can be done about population variation. As a rule of thumb, quadrupling the sample size will halve the error rate leading to a much more precise estimate. L. Bluff (pers. comm.) undertook a power analysis as part of developing the design for Southern Ark and again following the initial 2016–17 survey. He found that the sampling effort (n = 681 – herbivore-only baited sites) was sufficient to detect a 20% change in Long-footed Potoroo occupancy but this sample size was too low for Long-nosed Bandicoot and Long-nosed Potoroo. However, that analysis wasn’t aimed at assessing changes that may be related to the role of predators.

It might be possible to investigate changes in the predicted distribution of Long-footed Potoroos over time in Southern Ark. There are records from a range of other surveys that have been used to model the distribution of Long-footed Potoroos in the Southern Ark area (e.g. Figure 12). Using pre-2005 (pre-baiting) records of presence, modelled fox density (no control) and historical remotely sensed environmental data, it might be possible to recreate historical distribution predictions. These modelled predicted extents of occupancy could be compared with a similarly modelled pre-2019 fire extent to investigate the possible change in occupied habitat related to differences in fox density over time resulting from fox control. This could also form the basis for a longer-term monitoring design to look at the rate of post-fire recovery, the influence of different levels of (field-validated) fox density and other potential drivers of occupancy?

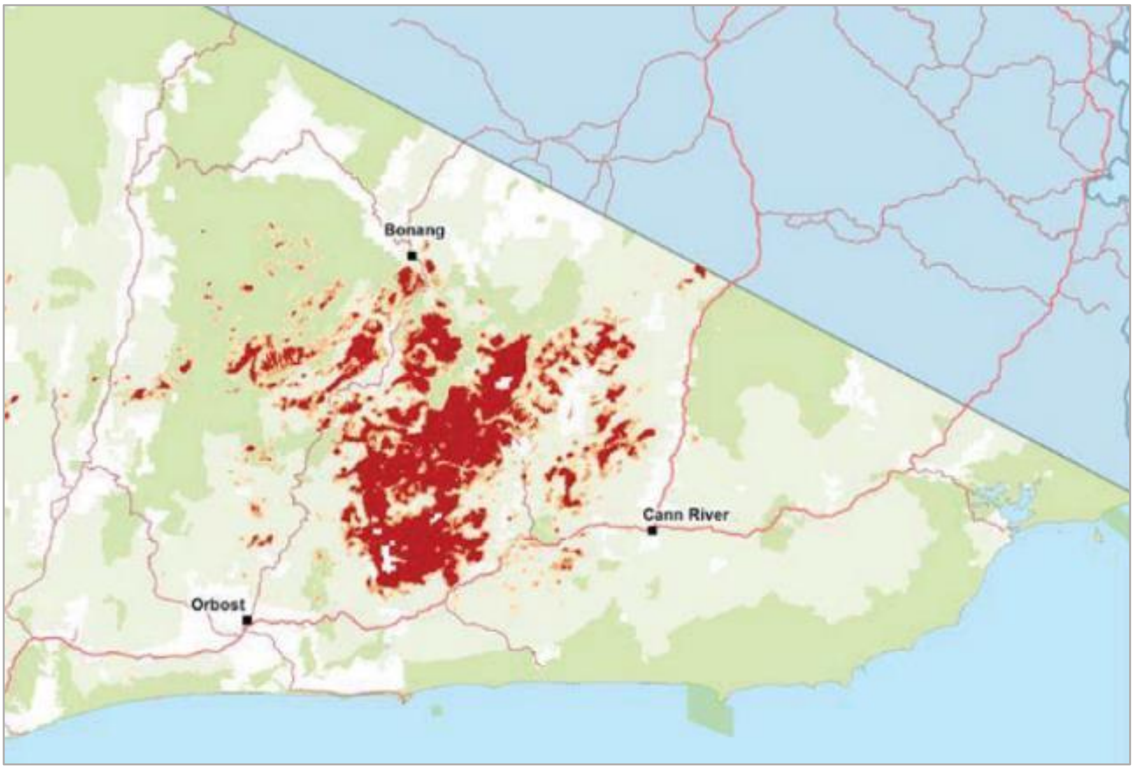

Figure 12. Predicted probability of occupancy for Long-footed Potoroos in East Gippsland based on all records to 2013 (Lumsden et al. 2013).

5 Conclusion

This study indicates that the severe bushfires that impacted large areas of East Gippsland in 2019–20 had varied impacts on a range of native species, with generally low impact on the immediate (6-months) post-fire occurrence of Long-footed Potoroos. However, there were some indications that more severely burnt sites were less likely to be occupied. Of the animals studied, Lace Monitors appear to be the most impacted by the fires, but Long-nosed Potoroos and Long-nosed Bandicoots were also detected at fewer sites after the fires. The role of predators was less clear. At low densities, foxes appeared to play a significant role in affecting the occurrence of Long-footed Potoroos but had no impact on the remaining species. There was no significant relationship between the presence of feral cats or wild dogs and the occurrence of any of the species studied.

Continued repeated sampling of a broader range of habitats with a range of fire severity impacts and predator densities will provide a valuable insight into the response of native species to the 2019–20 bushfires. Updating the initial power analysis with current data would establish the power of the monitoring to confidently detect changes in the status of species of interest. Although we did not detect any significant effect of fire severity or predation on several species (because of wide confidence intervals), there were trends in the data; for example, Long-footed Potoroo occupancy tended to increase with increases in feral cats. However, this suggested causality needs careful interpretation. It may be just as plausible that feral cats have responded positively to lowered fox densities and feral cats are selecting locations with Long-footed Potoroos.

Future work

There are several possible activities that could build on the current camera monitoring to improve our understanding of the response of native species in severely fire-affected landscapes and the role that introduced and native predators might play in their recovery:

- To reduce issues arising from the variable sampling effort, re-cast future camera surveys to sample across a range of fire-severity categories and predator densities.
- To assess the effectiveness of fox control, develop a long-term monitoring protocol for a selected range of species of conservation interest. This could use spatially explicit occupancy models to explore the changes in species distribution pre-baiting (constructing distributions from historical data) to post-baiting and incorporating the impact of the 2019–20 fires to predict likely future distribution.
- To improve model predictions and to assess the response of native species if feral cat control were introduced into Southern Ark in the future, robustly estimate fox and feral cat densities.
- Update the power analysis undertaken by Bluff (2016) using current data to determine what level of changes could be detected related to different levels of fox density and to explore options to improve the design.
- Include the 2021 camera monitoring data into the analysis undertaken in this report when it becomes available.

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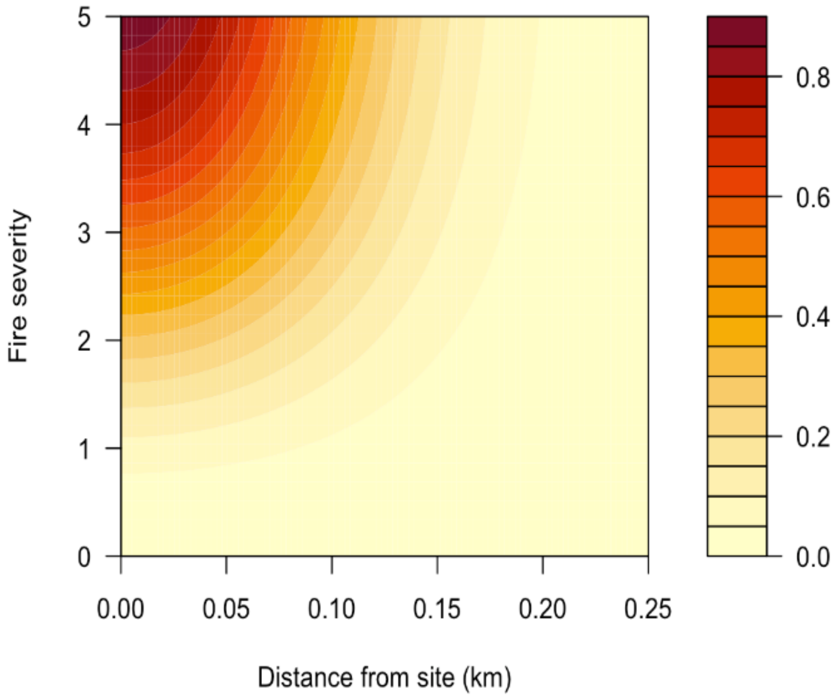
Appendices

Appendix 1. The relationship between fire severity category and distance from camera survey site

Weightings (red = high weighting, yellow = low weighting) of fire severity at a particular camera station in 2020 were based on the fire severity at the site and in the surrounding area (250 m radius). The figure shows the possible weighting values that can be ascribed to each raster cell within a 250 m radius of a camera. Every raster cell from the fire severity data (Department of Environment 2021) within a 250m radius will thus be weighted according to: (a) the distance of that raster cell from the camera and (b) the ascribed fire severity of that raster cell (0 to 5). At each site, once all raster cell weightings are calculated, weightings are summed and then scores are scaled (from 0 to 1).



Appendix 2. Total detections (no. of independent images per 24-hour period) for the target species during the study period.

Species	Season	No. Stations	N0. Detection	% sites
Common Brush-tailed Possum	2016	685	145	21.17
	2019	238	76	31.93
	2020	250	68	27.20
Eastern Ring-tailed Possum	2016	685	21	3.07
	2019	238	6	2.52
	2020	250	4	1.60
Lace Monitor	2016	685	168	24.53
	2019	238	60	25.21
	2020	250	2	0.80
Long-footed Potoroo	2016	685	208	30.36
	2019	238	86	36.13
	2020	250	136	54.40
Long-nosed Bandicoot	2016	685	154	22.48
	2019	238	21	8.82
	2020	250	21	8.40
Long-nosed Potoroo	2016	685	24	3.50
	2019	238	3	1.26
	2020	250	2	0.80
Potoroo sp.	2019	238	27	11.34
Southern Brown Bandicoot	2016	685	6	0.88
	2019	238	2	0.84
	2020	250	1	0.40
Feral cat	2016	685	200	29.20
	2019	238	36	15.13
	2020	250	47	18.80
Fox	2016	685	21	3.07
	2019	238	10	4.20
	2020	250	13	5.20
Wild dog	2016	685	27	3.94
	2019	238	14	5.88
	2020	250	4	1.60

Appendix 3. Conditional effects from model estimates showing the effects of various covariates on the probability of occupancy (logit-scale) for native species. Estimates with a 95% CI not overlapping zero are shown in bold.

Variable	Species	Mean	SD	CI 2.5%	CI 97.5%
Fire severity (scaled)	Common Brush-tailed Possum	–0.154	0.500	–1.132	0.801
	Lace Monitor	**–5.487**	**1.241**	**–8.135**	**–3.255**
	Long-footed Potoroo	**–1.120**	**0.554**	**–2.202**	**–0.065**
	Long-nosed Bandicoot	**–2.045**	**0.705**	**–3.473**	**–0.702**
	Long-nosed Potoroo	**–2.919**	**1.401**	**–6.012**	**–0.499**
Fox density	Common Brush-tailed Possum	0.506	0.297	–0.087	1.087
	Lace Monitor	0.663	0.376	–0.062	1.423
	Long-footed Potoroo	**–8.148**	**0.900**	**–9.925**	**–6.448**
	Long-nosed Bandicoot	-0.359	0.410	–1.194	0.414
	Long-nosed Potoroo	–0.069	0.737	–1.644	1.227
Feral cat	Common Brush-tailed Possum	0.689	1.179	–1.580	3.047
	Lace Monitor	0.098	1.737	–3.335	3.492
	Long-footed Potoroo	2.646	1.724	–0.589	6.126
	Long-nosed Bandicoot	1.602	1.712	–1.680	5.074
	Long-nosed Potoroo	0.112	1.942	–3.826	3.835
Wild dog	Common Brush-tailed Possum	–0.021	1.467	–2.951	2.936
	Lace Monitor	0.161	2.222	–4.246	4.682
	Long-footed Potoroo	–1.372	2.236	–5.932	2.869
	Long-nosed Bandicoot	0.840	2.172	–3.228	5.332
	Long-nosed Potoroo	–0.205	2.274	–-4.781	4.182
Mean temperature	**Common Brush-tailed Possum**	**0.516**	**0.134**	**0.253**	**0.789**
	Lace Monitor	**1.255**	**0.216**	**0.838**	**1.677**
	Long-footed Potoroo	**–0.862**	**0.160**	**–1.177**	**–0.552**
	Long-nosed Bandicoot	**0.439**	**0.171**	**0.116**	**0.773**

Variable	Species	Mean	SD	CI 2.5%	CI 97.5%
	Long-nosed Potoroo	**1.082**	**0.351**	**0.445**	**1.797**
Annual precipitation	**Common Brush-tailed Possum**	**–0.410**	**0.101**	**–0.607**	**–0.213**
	Lace Monitor	0.132	0.131	–0.119	0.393
	Long-footed Potoroo	–0.019	0.109	–0.233	0.192
	Long-nosed Bandicoot	0.197	0.123	–0.042	0.441
	Long-nosed Potoroo	**0.477**	**0.216**	**0.055**	**0.906**

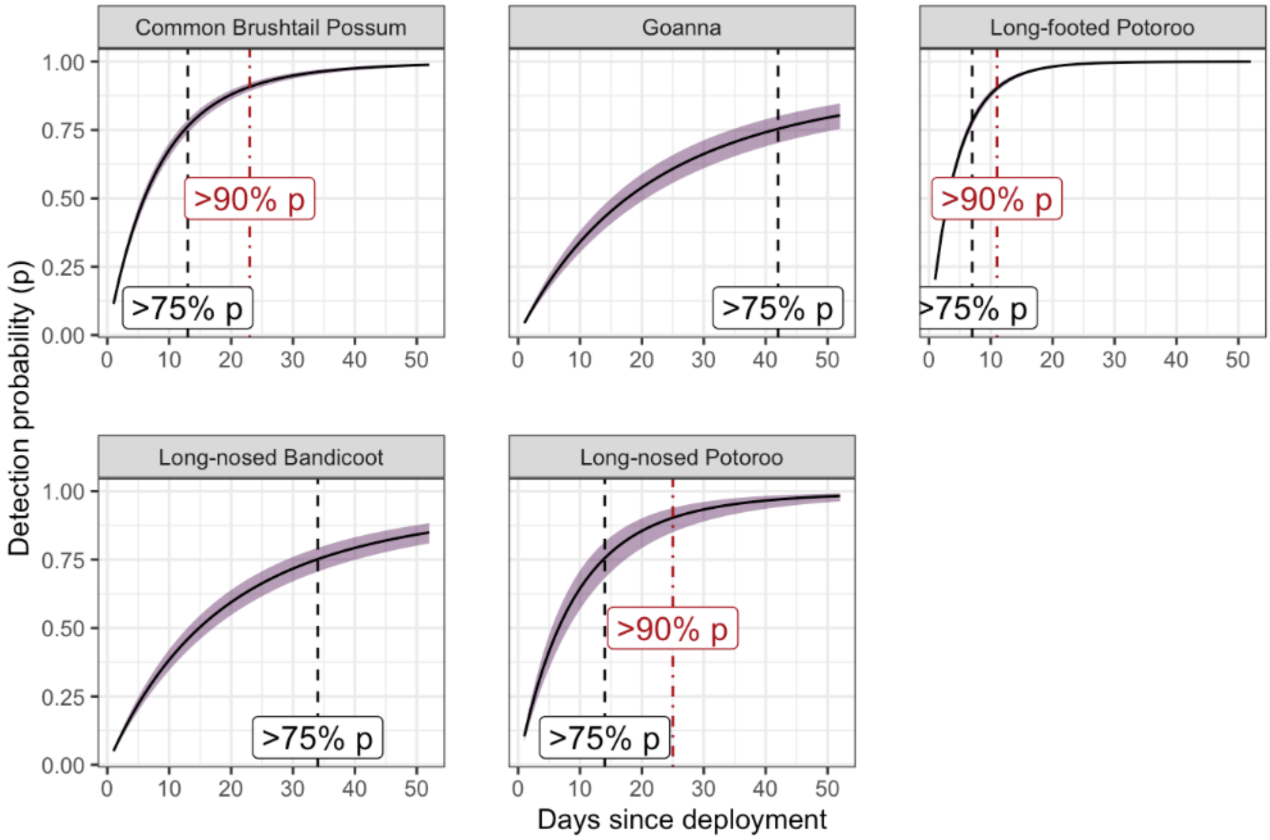
Appendix 4. Conditional effects from model estimates showing the effects of various covariates on the probability of occupancy (logit-scale) for predators. Estimates with a 95% CI not overlapping zero are shown in bold.

Variable	Species	Mean	SD	CI 2.5%	CI 97.5%
Fire severity (scaled)	**Feral cat**	**–1.136**	**0.632**	**–2.546**	**–0.026**
	Wild dog	–1.741	1.064	–4.029	0.098
	Fox	–0.153	0.826	–1.836	1.326
Mean temperature	**Feral cat**	**–0.309**	**0.119**	**-0.550**	**–0.082**
	Wild dog	–0.156	0.193	–0.517	0.233
	Fox	**–0.555**	**0.172**	**–0.892**	**–0.214**
Annual precipitation (scaled)	Feral cat	0.024	0.131	–0.232	0.272
	Wild dog	–0.376	0.225	–0.814	0.077
	Fox	**–0.829**	**0.215**	**–1.252**	**–0.417**
Season precipitation (scaled)	Feral cat	–0.063	0.124	–0.308	0.182
	Wild dog	0.093	0.220	–0.337	0.525
	Fox	**0.729**	**0.212**	**0.322**	**1.146**

Appendix 5. Conditional effects from model estimates showing the effects of various covariates on the probability of detection (logit scale) for native species. Estimates with a 95% CI not overlapping zero are shown in bold.

Variable	Category	mean	SD	2.5%	97.5%
Daily maximum temperature	Common Brush-tailed Possum	–0.003	0.006	–0.014	0.008
	Lace Monitor	0.014	0.010	–0.005	0.033
	Long-footed Potoroo	**–0.011**	**0.003**	**–0.017**	**–0.004**
	Long-nosed Bandicoot	**0.034**	**0.009**	**0.016**	**0.051**
	Long-nosed Potoroo	0.024	0.017	–0.009	0.058
Days since deployment	**Non-predator lure**	**–0.015**	**0.001**	**–0.018**	**–0.012**
	Predator lure	**–0.010**	**0.004**	**–0.018**	**–0.002**

Appendix 6. Cumulative detection probabilities over the course of deployment varies between species. Depending on target species, cameras may be deployed for varying time periods. The solid line represents the mean detection probability estimate with the shaded bands representing the 95% CIs.



Supplementary material

The relevant R code alongside additional plots, tables and descriptions of analyses conducted here are available in an RMarkdown at https://justincally.github.io/blog/posts/2021-09-21-southernarkfire/

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