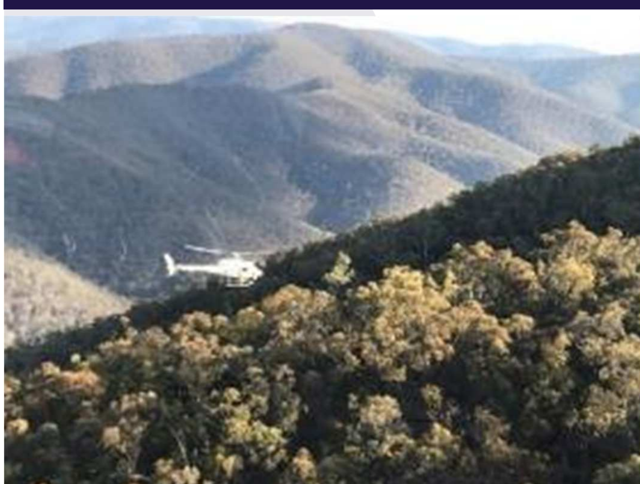
Estimating population changes in wild dogs, feral cats and foxes in relation to an aerial baiting operation in eastern Victoria

A. Robley, D.S.L. Ramsey and L. Woodford

July 2018





Arthur Rylah Institute for Environmental Research
Technical Report Series No. 292

Arthur Rylah Institute for environmental research

VICTORIA State Government | Environment, Land, Water and Planning

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Citation: Robley, A., Ramsey, D.S.L. and Woodford, L. (2018). Estimating population changes in wild dogs, feral cats and foxes in relation to an aerial baiting operation in eastern Victoria. Arthur Rylah Institute for Environmental Research Technical Report Series No. 292. Department of Environment, Land, Water and Planning, Heidelberg, Victoria.

Front cover photo: Deploying wild dog baiting (Vaughn Kingston). Wild dog, Red Fox, Feral Cat, and setting up remote camera (DELWP).

Printed by Victorian Government, Melbourne

Edited by Organic Editing

ISSN 1835-3827 (print)
ISSN 1835-3835 (pdf))
ISBN 978-1-76077-126-3 (Print)
ISBN 978-1-76077-127-0 (pdf/online/MS word)

Accessibility

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Estimating population changes in wild dogs, feral cats and foxes in relation to an aerial baiting operation in eastern Victoria

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Acknowledgements

We would like to thank the Agriculture, Energy and Resources Division of the Department of Economic Development, Jobs, Transport and Resources (DEDJTR) for funding this project. We would like to acknowledge the support of Andrew Woolnough and Will de Milliano from DEDJTR, of Drew Sutton from the Department of Environment, Land, Water and Planning for assistance with mapping and GIS information, and of Vaughn Kingston for guidance and input into project planning. We would also like to acknowledge the assistance of staff from Wildlife Unlimited (Hagan Brightman, David Caldwell, Greg Fyfe, Rena Gaborov and Claire McGall) in placing and retrieving cameras before and after the aerial baiting operation. Earlier drafts of this report were improved by contributions from Lindy Lumsden, and editorial input was provided by Organic Editing.

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Summary

Context:

In early 2014, the Australian Government provided conditional approval under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC) for Victoria to conduct aerial baiting for the control of wild dogs (collectively, *Canis lupus dingo*, *Canis lupus familiaris* and their hybrids). This approval allows for baiting to occur until 31 December 2019 in nominated zones in the eastern highlands of Victoria.

Changes in the numbers of dingoes and wild dogs resulting from aerial baiting have been assessed in other States. However, those assessments were undertaken using indices of abundance, the control actions occurred in different environments (e.g. arid or semi-arid Western Australia), and they used different baiting rates (40 baits per linear transect in New South Wales versus 10 used in Victoria). No information on changes in wild dog density or abundance following aerial baiting has until now been available for Victorian conditions.

There is some evidence in the literature that the reduction or removal of apex predators can have unintended consequences for lower-order predators. We thus also assessed the population changes in feral cats (*Felis catus*) and red foxes (*Vulpes vulpes*).

Aims:

We aimed to assess changes in wild dog abundance and density following a single aerial baiting operation in eastern Victoria. We also aimed to assess changes in the abundance and density of feral cats and changes in occupancy of foxes.

Methods:

This project was carried out in state forest approximately 15 km south-west of Swifts Creek in East Gippsland, Victoria. The aerial baiting operation was conducted by the Department of Economic Development, Jobs, Transport and Resources (DEDJTR) in October 2017 along the eastern fringe of the study site and within 3 km of farmland. Baits were deployed at a rate of 10 baits per linear kilometre, using 250 g meat baits injected with 6 mg/kg of 1080 (sodium mono-fluoroacetate).

To collect the information required for determining the density and abundance of wild dogs and feral cats (based on the ability to recognise individual animals), and the occupancy rates of foxes, we deployed 107 digital cameras. These were spaced at 1.6 km intervals along roads and tracks throughout the study area for 26 days before and 38 days after the aerial baiting operation, covering a sample area of 1844 km^2.

We identified individual wild dogs and feral cats from distinctive markings and physical appearance, and used this information to construct detection histories. Estimation of changes in density and abundance were made using spatial mark–recapture models in the *secr* package in R (v. 3.4.4). Estimates of changes in occupancy rates for foxes were made using multiseason occupancy models in *Presence* (v. 2.12.9).

Results:

Wild dog abundance and density decreased by 27% in the period following the aerial baiting operation, from 48 to 36 wild dogs, and from 0.026 to 0.019 dogs/km^2, respectively. The estimated mean home range size for the remaining wild dogs increased from 137 km^2 to 193 km^2 in the period following baiting. The daily detection probability was 0.051 [95% CI 0.042–0.062]. Before and after the baiting wild dogs centred their activity in the western and southern parts of the study area and were generally absent from the baited area.

Feral cat abundance and density increased by 21% in the period immediately following aerial baiting from 82 to 103 feral cats and from 0.075 to 0.107 cats/km^2, respectively. The estimated mean home range size for feral cats increased slightly, from 1.16 km^2 to 1.40 km^2, in the period following baiting. The daily detection probability in the pre-baiting period was 0.144 (95% CI 0.114–0.180) and following baiting it was 0.097 (95% CI 0.081–0.115). The feral cats' activity was spread uniformly across the study area before and after the baiting.

The fox occupancy rate decreased by 23% across the study site following the aerial baiting, from 0.50 (95% CI 0.38–0.62) to 0.41 (95% CI 0.30–0.51).

Conclusions and implications:

Wild dog density and abundance and fox occupancy declined following the aerial baiting operation. A key difference between our study and previous studies is that we assessed density directly using the detection rates of known individual wild dogs, as well as considering the detection of unidentifiable individuals, and incorporated this information into a modelling approach that determined wild dog density and abundance.

As far as we are aware, this work is the first time in Australia that this approach has been applied to estimating and assessing changes in wild dog density and abundance resulting from a management action. This study builds on previous research which assessed approaches to determining wild dog density and abundance at Gudgenby, New South Wales.

We also showed that feral cat density, abundance and home range size increased following the aerial baiting operation. These increases may have been due to an apparent increase in their daily activity as there was no significant increase in new individual between periods.

There is clearly a correlation between the decline in both wild dogs and foxes, the apparent increase in feral cats, and the implementation of the aerial baiting operation. Our findings are in-line with other studies and mesopredator theory which suggests smaller predators like feral cats should benefit from the removal or reduction in lager predators such as foxes and wild dogs. However, without replication and/or comparison with untreated locations it is not possible to discount other possible causes or confounding effects, or to ascribe causation.

Directly estimating density and abundance of wild dogs is a significant advancement in the capacity of land managers to robustly assess the outcome of management interventions, and it should lead to more informed decisions being made about the effectiveness and efficiency of current and future operations.

1 Introduction

Dingoes (*Canis lupus dingo*) and feral or wild-living domestic dogs (*Canis lupus familiaris*) and their hybrids, collectively termed ‘wild dogs’ in Victoria, reduce farm productivity and prey on native animal species. The economic impact of wild dogs across Australia is estimated to be between $66.3 million and $48.5 million per annum (McLeod 2004; Gong et al. 2009, respectively). In Victoria, the State Government invests approximately $4.4 million per annum in managing the impact of wild dogs on agricultural enterprises.

Wild dog control in Victoria takes an integrated approach, which includes baiting (both ground-based and aerial), trapping and shooting (all within a 3 km buffer within public land adjoining private land), exclusion fencing, guardian animals, community-based control activities and the encouragement of good animal-husbandry practices to minimise impacts.

In early 2014, based on an application that included information on Spotted-tailed Quoll (*Dasyurus maculatus*) prevalence in baiting areas, the Australian Government provided conditional approval under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC) to conduct aerial baiting. This approval allows for aerial baiting to occur until 31 December 2019 in nominated zones across Gippsland and north-east regions, which collectively cover the eastern highlands of Victoria.

Both wild dogs and foxes are susceptible to poison bating, and while the effectiveness of aerial baiting has been assessed in other States (Tomlinson 1954; Newsome et al. 1972; Thomson 1986; Fleming et al. 1996), to date this has either occurred in different environments (e.g. arid or semi-arid Western Australia) or used different baiting rates [40 baits per linear transect in New South Wales (NSW) versus 10 in Victoria]. Also, previous assessments used indirect measures of assessing changes in wild dog populations, e.g. the number of radio-collared individuals killed (Thomson et al. 1992) or the number of visitations to watering holes (Fleming et al. 2001). No information on changes in wild dog density or abundance following aerial baiting has until now been available for Victorian conditions. Feral cats can also be reduced by poison baiting (Comer et al. 2017), however; the feral cats are unlikely to take a 250g dried meat bait preferring small, most baits (Algar and Burrows 2004).

There is considerable interest in the interactions between wild dogs, feral cats (*Felis catus*) and red foxes (*Vulpes vulpes*) in Australia (Brook et al. 2012; Allen et al. 2014). Reduction in the abundance of both wild dogs and foxes could be beneficial to feral cats, potentially releasing feral cats from direct (predatory) and indirect (competitive) effects.

We estimated the relative population sizes of introduced predators both before and after an aerial baiting operation in October 2017 in eastern Victoria, using digital cameras spaced throughout several wild dog home ranges. To estimate population changes in wild dogs and feral cats, we used spatial mark–resight methods (Sollmann et al. 2012). This approach [developed for the Department of Economic Development, Jobs, Transport and Resources (DEDJTR) by the Arthur Rylah Institute of Environmental Research (ARI)] builds on previous work by ARI estimating introduced predator density at Gudgenby, NSW (Forsyth et al. 2018 submitted). To estimate population changes in foxes, we assessed any differences in seasonal occupancy rates across the study area (MacKenzie et al. 2006).

To improve future investigations into the changes in wild dog density arising from aerial baiting several factors would need to be considered. Ground-based baiting, shooting, fencing and trapping are all tools employed to various degrees within the 3-km buffer zone of our study area and would need to be adequately controlled for. Also, replicating the trial, comparing outcomes with introduced predator population changes at untreated sites, and randomly allocating the treatment would all improve the robustness and generality or the outcomes.

This report does, however, provide the first direct estimate of wild dog density, further validates the methods developed by ARI for assessing wild dog and feral cat abundance and density, provides insights into the possible impact aerial baiting (as part of an integrated control approach) has on wild dogs, feral cats and foxes, and suggests future approaches to improving our current state of knowledge of the impact of control actions on introduced predators in Victoria.

2 Methods

2.1 Study site

This research project was carried out in state forest, approximately 15 km south-west of Swifts Creek in East Gippsland, Victoria (147°63' 02" S, 37° 21' 28" E) (Figure 1) and covered approximately 1844 km^2.

The area is primarily foothill forest, characterised by Grassy Dry / Heathy Dry Forest and Tall Mixed Forest. The site is flanked to the east by farmland used for the rearing of livestock. The height above sea level within the study area ranges from ~300 m in the valleys to up to ~1000 m on the peaks. The average annual rainfall is 781 mm (Bureau of Meteorology Online Climate Statistics www.bom.com.au).

The site was selected because it adjoins private land with a history of stock attacks by wild dogs, is subject to an annual wild dog control program delivered through aerial baiting, had had no ground-based baiting in the previous 12 months, had good track access, and wild dogs were known to be active in the study area.

Aerial baiting was conducted by the DEDJTR along the eastern fringe of the study site in an area within 3 km of farmland (Figure 1). The baiting occurred along flown transects, mostly on ridge tops and spurs. The transects were flown in October 2017, and 250 g meat baits containing 6 mg/kg of the poison 1080 (sodium mono-fluoroacetate) were distributed from the aircraft at a rate of 10 baits per kilometre.

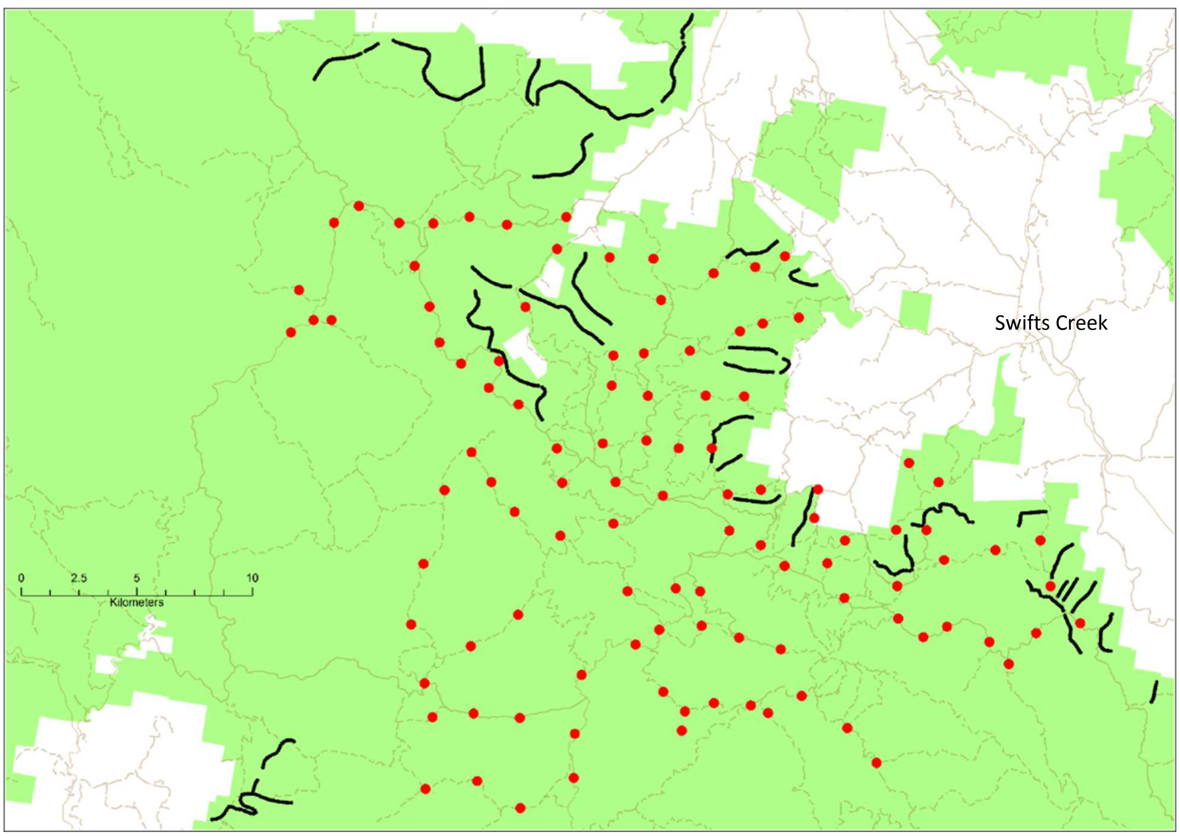

Figure 1. Location of the study site near Swifts Creek. The red dots represent the location of the camera traps, and the black lines indicate the transects flown for the aerial baiting program in October 2017.

2.2 Monitoring population changes in wild dogs, feral cats and red foxes

To collect the information required for determining density and abundance of wild dogs and feral cats, and fox occupancy rate, we deployed 107 digital cameras along roads and tracks throughout the study area (Figure 1), covering a much larger area than the 3-km buffer zone. This was to allow for the detection of wild dogs across multiple home ranges.

During late September 2017, 107 heat-in-motion cameras [(Reconyx HC600/HC900 Hyperfire H.O. Covert IR (RECONYX, Inc., Holmen, Wisconsin, USA)] were deployed and left in the field for an average of 26 days. Cameras were spaced at approximately 1.6 km intervals along forest roads and tracks, and placed 1–3 m from the road or track edge, with the sensor located approximately 25 cm above the ground. This was to help hide the cameras from passing vehicular traffic, while allowing the camera to detect and photograph mid-sized mammals such as wild canids and feral cats that frequently uses tracks and roads and are more detectable. Cameras were locked to trees where possible, or attached to stakes, and camouflaged with nylon camouflage netting and natural vegetation (Figure 2).

Cameras were orientated to face south, where possible, to reduce direct exposure to the sun (which can cause the cameras to take false images). The cameras were set to take five images in succession, with no delay between triggers. All image files were saved onto SD memory cards.

Figure 2. Camera trap set low and near the edge of a track to capture images of wild canids and feral cats

In the second week of November 2017, and one month following the aerial baiting operation, 107 Reconyx camera traps were deployed in the same locations, using the same methodology as for the pre-baiting monitoring. Cameras were retrieved during mid-December, resulting in an average of 38 camera trap nights.

2.2.1 Wild dog, feral cat and red fox identification

Photographs showing images of wild dogs, red foxes and feral cats were placed into separate, named folders. When images were of poor quality or too dark to see clearly, software (www.fotor.com) was used to brighten and clarify the images. To estimate the minimum number of wild dogs in the area, individual dogs were identified by first grouping images by general coat colour (yellow, sable, black, brindle). Within these groups, individual dogs were identified by studying multiple images and comparing markings (e.g. presence, location and size of ‘socks’, colouration around muzzles, chest markings), signs of age (e.g. grey muzzle, greying coat, size), and physical condition, including injuries. Figure 3 shows an example of wild dogs with distinguishable markings.

Figure 3. Examples of wild dogs caught on camera, showing how individuals could be distinguished by their markings

Wild dogs were then assigned a unique identifier based on their overall coat colour (e.g. BT1 would be black and tan dog 1), and the date and location they were seen were then recorded in a spreadsheet. Any dogs that appeared the same but which were detected at extreme ends of the study site were singled out and rechecked to ensure that they were the same individuals. Dogs that could not be assigned a unique identifier were collectively recorded as 'unknown'.

Wild dogs that were recorded during the post-baiting monitoring were cross-referenced with dogs with the same unique identifier from the pre-baiting monitoring to ensure that they were the same animal. New dogs detected were assigned a new unique identifier. A second person was shown images of the dogs throughout the dog recognition process to verify they were the same individuals.

Many feral cats, especially 'tabby' cats, have readily identifiable coat patterns, particularly on the flanks, but these may vary from one side to the other of the same cat. The flank coat pattern typically features distinctive dots, lines or swirls. Figure 4 shows an example of wild dogs with distinguishable marking.

Figure 4. Examples of feral cat caught on camera, showing how individuals could be distinguished by their markings

To facilitate the identification of individuals, each flank pattern, for both sides of the cat when possible, was drawn freehand and given a unique identifier. This enabled each known flank pattern to be compared more easily with the flank patterns on the images of feral cats obtained from the various cameras. This improved the accuracy and efficiency of the process for recognising individuals. Each individual feral cat was assigned a unique identifier and its data was recorded on a spreadsheet. Most black-coloured feral cats were recorded as unknown individuals and not assigned unique identifiers, because it was very difficult to tell them apart. Feral cats that were recorded during the post-baiting monitoring were cross-referenced with cats with the same unique identifier from the pre-baiting monitoring to ensure that they were the same animals. New cats detected post-baiting were assigned new unique identifiers.

2.2.2 Estimating wild dog and feral cat densities and abundance

The photographic catalogue of known wild dogs and feral cats was used to construct detection histories for these individuals. Wild dogs and feral cats without distinctive natural markings were essentially unidentifiable ('unmarked'). Hence, the detection data consisted of the detection histories of marked individuals, and also the number of detections of unmarked individuals by each camera for each species.

Spatial capture–recapture (SCR) methods (Borchers and Efford 2008; Royle and Young 2008) are ideally suited for estimating population density in wide-ranging and cryptic predator species because, unlike conventional (non-spatial) capture–recapture methods, SCR models explicitly incorporate the sampled area into the estimation process, so that the estimation of population density is straightforward. SCR methods also overcome other technical problems that cause bias in conventional capture–recapture methods, such as heterogeneity in detection due to differential exposure of individuals to detection devices, i.e., camera traps in this study (Borchers and Fewster 2016). Recent advances in SCR methods have developed spatially explicit alternatives for density estimation in populations that are only partially marked. These types of models are known as spatial mark–resight (SMR) models (Chandler and Royle 2013; Royle et al. 2013).

To estimate the densities of both wild dogs and feral cats at the site, we used SMR models that utilised both marked and unmarked individuals for analyses. SMR models assume that the marked individuals are a random sample from the population, and that marking occurs throughout the defined state-space. For our data, we considered that the assumption that wild dogs and feral cats with distinctive marks were a random sample from the wider population was reasonable. We also considered it likely that 'marked' wild dogs or feral cats were no more likely to be detected than wild dogs or feral cats without such markings. In addition, it was also assumed that all marked individuals were correctly identified, and that no marked individuals were lost or emigrated from the area during the pre- or post-baiting surveys.

SMR algorithm

The data consisted of the array of J cameras having locations at X = (x_{j1}, x_{j2}), (j = 1, 2, … J) and set for K occasions (k = 1, 2, … K), where an occasion is a 24-hr period. The observations at each camera denoted *hjk* and take binary values (0 not detected, 1 detected), indicating detection of at least one individual on camera j at occasion k. Hence h_1 = (01001) indicates detections on occasions 2 and 5 for camera number 1. The resulting data are a J × K matrix of detections h.

The conceptual model underlying the detection process is a spatially explicit, individual-based model of detections in cameras located in two-dimensional space. Consider a population of N individuals that are potentially at risk of being detected, with each individual (i = 1, 2, …, N) defined by a centre of activity s_i = (s_x, s_y), its nominal home range centre. The locations of home range centres are unknown, but are considered fixed for the duration of sampling. Individuals move about their home range centres according to some probability distribution (e.g. bivariate normal), and in the process, can potentially be exposed to detection. We also assumed that home range centres could be located anywhere within the area of interest A with equal probability. This was achieved by assuming that the home range centres were distributed according to a homogeneous spatial Poisson process with constant intensity (density) over the area of interest A.

$$\mathbf{s}_i \sim Poisson(\mu) \tag{1}$$

where μ is the mean intensity (density) of the spatial process. We also considered alternative models based on an inhomogeneous Poisson process, in which the distribution of the s_i could vary spatially, dependent on spatially varying covariates.

$$\mathbf{s}_i \sim Poisson\big(\mu(s,\beta)\big) \tag{2}$$

where the intensity process μ now depends on spatial covariates β. Thus, inference is concerned primarily with estimating the locations of the unknown home range centres s_i and hence, the abundance $\widehat{N}$ (and density μ) of individuals within the region A (Royle et al. 2009).

Encounter process

Individuals can only encounter cameras that occur within their home range. If we consider the situation with only one animal and one camera, the probability of detecting the individual declines as a function of the distance d between the camera and the home range centre. Assuming movements around the home range centre occur with bivariate normal probability, the probability of detection is given by the half-normal function:

$$p_{ij} = g_0 e^{-d_{ij}^2/2\sigma^2} \tag{3}$$
$$d_{ij} = \left\|\mathbf{x}_j - \mathbf{s}_i\right\|$$

where g_0 is the per-occasion (24-hr period) probability of detection when the home range centre and camera location coincide (i.e. d = 0), and σ is the spatial scale over which the detection probability declines with increasing distance between the home range centre and the camera (Efford 2004; Ramsey et al. 2005).

The data for the SMR algorithm consists of two parts. The first part consists of the detection histories *hij* for each marked individual i (i = 1...m), detected by camera j on occasion k. Since there could be more individuals with unique natural markings in the population than those detected, the size of the 'marked' population is essentially unknown. The second part consists of the detections of unmarked individuals. These data consist of matrix of detections for each camera on each occasion and are modelled separately by assuming they arise from a separate (possibly inhomogeneous) Poisson process, conditional on the site-specific detection function (equation 3) (Efford and Hunter 2018).

2.2.3 Data analysis

SMR models were fitted to the wild dog and feral cat detection data with maximum likelihood, using the *secr* package (v. 3.1.3; Efford and Hunter 2018) in R (R Development Core Team 2015). We combined the data for each period (i.e. pre- and post-baiting periods) as a multi-session capture history matrix, with the unmarked detections considered as additional sightings for analysis within *secr*. We initially fitted four models to the combined pre-baiting and post-baiting camera data to assess whether the detection parameters (g_0, σ) (Equation 3) varied between the survey periods (Models 1–4; Table 1). The best-fitting of these models was then adjusted for overdispersion of the counts of unmarked sightings by estimating a variance inflation factor ($\hat{c}$; Efford and Hunter 2018). The variance–covariance matrix of all further models was then adjusted using the estimated value of $\hat{c}$. For the wild dog data, we then fitted two inhomogeneous density models in which the dog density varied linearly with easting (x) and northing (y) coordinates or in response to the locations of the baited transects (Distance To Bait, *DTB*) (Models 5 and 6; Table 1). The relative fit of all models was compared using the Akaike Information Criterion (AIC) adjusted for small sample size (AICc) (Burnham and Anderson 1998).

Table 1. List of models fitted to the spatial mark–resight data from the wild dog and feral cat detections in the pre- and post-baiting period surveys

The parameters are D = density, P = survey period specific, g_0 = per-occasion (24-hr) detection probability, σ = home range scale, $x + y$ = linear trend, DTB = distance to baits. M5 and M6 were not fitted to the feral cat data.

Model	Model formula
M1	$D \sim P, g_0 \sim 1, \sigma \sim 1$
M2	$D \sim P, g_0 \sim 1, \sigma \sim P$
M3	$D \sim P, g_0 \sim P, \sigma \sim 1$
M4	$D \sim P, g_0 \sim P, \sigma \sim P$
M5	$D \sim x + y$
M6	$D \sim DTB$

For these analyses, we constructed a habitat mask which included buffering around the outermost camera locations by 10 km for wild dogs and by 5 km for feral cats. A buffer was included to capture individuals whose home range centres adjoined the main study area. These values were chosen based on trial model runs to estimate the species home range scale parameter σ, with the buffer width calculated as 4σ. Buffer widths greater than this value were shown to have negligible impact on density estimates (assuming a half-normal detection function). Using this buffer, a discrete habitat mask for wild dogs was created, consisting of 2577 cells with a resolution of 846 m (i.e. a cell area of 71.5 ha) and having a total area of 1844 km^2. The same process was undertaken for feral cats, resulting in a habitat mask consisting of 2281 cells with a resolution of 689 m (i.e. a cell area of 47 ha) and having a total area of 1085 km^2.

2.2.4 Fox occupancy

The phrase 'occupancy' is used here to mean the proportion of a sampling unit (study site) that contain the target species at a given point in time. As sampling is repeated before and after baiting, and the same sites are surveyed each time, this approach can be used to estimate the occupancy rate. The change in occupancy rate between periods is then modelled as a function of site colonisation and extinction rate, analogous with the birth and death rates in an open-population mark–recapture study.

Two key assumptions in occupancy modelling are that: (i) sites are closed during the survey period, and (ii) the detection process is independent at each site (MacKenzie et al. 2005). The main aim of our study was to estimate the density of wild dogs, and this required camera stations to be placed at 1.6 km intervals. This spacing was likely to result in correlated detections between camera sites, i.e. foxes detected on one camera were likely to have been detected on an adjacent camera, as foxes are known to be capable of travelling >1.6 km along tracks (Hradsky et al. 2017).

To account for this correlation in detection probability, we used an occupancy modelling approach that explicitly allowed for correlated detections (Hines et al. 2010). Data for foxes was summarised by camera site such that the observed presence–absence on each day *j* at site *i* in period *t* was indicated by $Y_{i,j,t}$ = 1 and 0, respectively, for each *j* day corresponding to the day of monitoring in each period. It was considered likely that the probability of detection differed between periods, due to either changes in abundance or activity. The true, but unknown, occurrence at a site in a given period ($z_{i,t}$) was modelled as a random variate from a Bernoulli distribution with the parameter $\psi_{site[i],t}$, the probability of occurrence (i.e. occupancy) in period *t*:

$$z_{i,t} \sim \mathrm{Bern}(\psi_{site[i],t}).$$

The repeat surveys allowed us to construct a detection history for each camera site, and thus estimate a separate detection probability for each period, conditional on the site being occupied. The observed presence–absence data were modelled as:

$$Y_{i,j,t} \sim \mathrm{Bern}\ (z_{i,t} \times p\theta_t, p\theta'_t),$$

where $p\theta_t$ = Pr(species present on camera | sample unit occupied and species not present at previous camera site); $p\theta'_t$ = Pr(species present on camera | sample unit occupied and species present on previous camera site) and $z_{i,t}$ is the true occurrence estimated above.

We pooled estimates of true occurrence to derive estimates of the total number of occupied sites in each period, enabling a direct assessment of the effect of aerial baiting on fox occupancy.

We also used the estimates of the true occurrence to derive estimates of average site colonisation and extinction between periods. Colonisation (γ_t) is analogous to the reproduction rate of a population, and is the probability that a site that was not occupied at time t_1 becomes occupied at t_2 (MacKenzie et al. 2005). Extinction (ε_t) is analogous to mortality and is the probability that sites that were occupied at t_1 were not occupied at t_2.

Colonisation and extinction are important as they are the processes that drive occupancy, i.e. occupied sites will become unoccupied (extinct) with probability ε, and unoccupied sites will be colonised with probability γ. They can either be estimated directly (MacKenzie et al. 2005), or derived from the estimates of true occupancy.

We initially fitted models to the combined pre- and post-baiting camera data to assess whether the occupancy, colonisation, extinction and detection parameters (Ψ, γ, ε, P) were constant, or whether detection varied between the survey periods (Models 1–2; Table 2). We then fitted models to assess the effect, if any, of correlated detections (θ, θ') (Models 4 and 5; Table 2). The relative fit of all models was compared using Akaike's Information Criterion (ΔAIC) (Burnham and Anderson 1998).

Analysis was undertaken using the statistical software program *Presence* (version 2.12.9, Hines 2006).

Table 2. List of models fitted to the occupancy data from the fox detections in the pre- and post-baiting period surveys

Model	Model formula
M1	Ψ, $\gamma \sim 1$, $\varepsilon \sim 1$, $p \sim 1$
M2	Ψ, $\gamma \sim 1$, $\varepsilon \sim 1$, $p \sim P$
M3	Ψ, θ,θ', $\gamma \sim 1$, $\varepsilon \sim 1$, $p \sim 1$
M4	Ψ, θ,θ', $\gamma \sim 1$, $\varepsilon \sim 1$, $p \sim P$

Ψ = Pr (sample unit occupied); γ = colonisation rate; ε = extinction rate; p = Pr (detection at a site | sample unit occupied and species present on segment); θ = Pr (species present on site | sample unit occupied and species not present on previous segment); θ' = Pr (species present on site | sample unit occupied and species present on previous segment). ~1 indicates constant.

3 Results

3.1.1 Wild dogs

For wild dogs, both adults and pups were detected on cameras. As pups were likely to be closely associated with one or more adults and were also likely to have had quite different home ranges and detection probabilities to those of adult dogs, we removed them from our analyses. Hence, this analysis uses detections of adult and sub-adult dogs only. One detection of a likely domestic dog was also removed from the analysis. Of the wild dog detections, 26 individuals could be uniquely identified based on natural markings from the pre-baiting monitoring (133 total detections at an average of 5.1 detections per day), and 24 individuals could be uniquely identified from the post-baiting monitoring (152 total detections at an average of 4.0 detections per day) (Figure 5). In addition, there were a total of 18 and 15 detections of unmarked individuals in the pre- and post-baiting monitoring, respectively. Figure 6 indicates the location and frequency of unmarked wild dog detections.

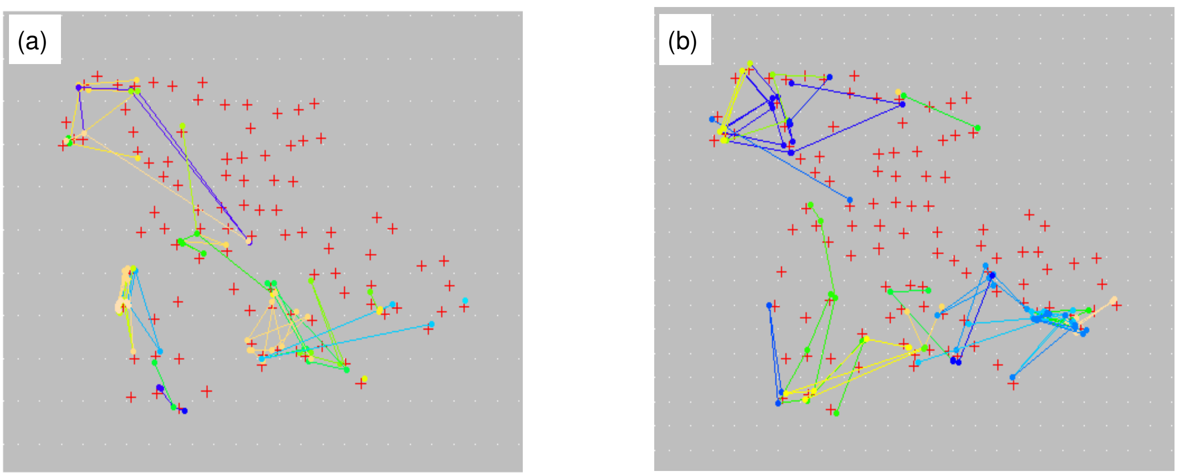

Figure 5. Detections of individually recognisable wild dogs in the (a) pre- and (b) post-bait camera trap monitoring. Detections of individual dogs are connected by lines with colours representing individuals.

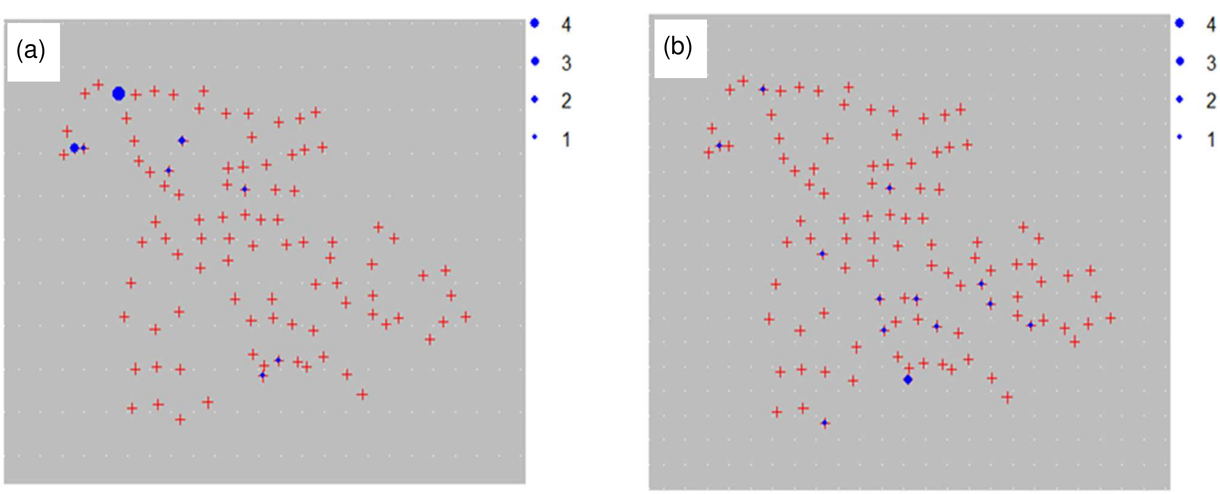

Figure 6. Locations of detections of wild dogs that were unidentified (blue circles) during the (a) pre- and (b) post-baiting periods. The size of the circle indicates the number of detections at a camera trap.

Results from the model selection of the detection parameters (g_0, σ) revealed that there was relatively stronger support for the model where the per-occasion detection probability g_0 was constant between survey periods, with the spatial scale parameter σ varying between survey periods (model M2; Table 3). We therefore used model M2 as the basis for model selection of the alternative models for density and for estimation of the overdispersion parameter.

Table 3. Results from the model selection of the detection parameters (g_0, σ). Models varied as to whether the detection parameters were constant (~1) or varied by survey period (~*P*).

Model	Model formula	AICc	∆AICc	Weight
M2	$D \sim P$, $g_0 \sim 1$, $\sigma \sim P$	4175.5	0.0	0.51
M4	$D \sim P$, $g_0 \sim P$, $\sigma \sim P$	4178.2	1.4	0.26
M1	$D \sim P$, $g_0 \sim 1$, $\sigma \sim 1$	4178.0	2.0	0.18
M3	$D \sim P$, $g_0 \sim P$, $\sigma \sim 1$	4180.0	4.5	0.05

(AICc refers to the Akaike information criterion with a correction for small sample size.)

The estimates of the overdispersion parameter $\hat{c}$ revealed the presence of relatively strong overdispersion in the unidentified counts of wild dogs ($\hat{c}$ of 12 and 30 for the pre- and post-baiting periods, respectively). We therefore used estimates of $\hat{c}$ to adjust the variance estimates for all the alternative models of wild dog density.

Models fitted to explain variation in wild dog density revealed that the model where density was constant across space, but varied between survey periods, received overwhelming support compared with the alternative models where density varied across space (i.e. inhomogeneous density models) (Table 4).

Table 4. Results from the model selection of alternative density models for the wild dog data. Density varied as a function of either survey period (*P*), a linear function of space ($x + y$) or distance to bait line (*DTB*). Detection parameters were modelled using the best-fit model from Table 2. All models were adjusted for overdispersion using an estimate of $\hat{c}$.

Model	Model formula	AICc	∆AICc	Weight
M0	$D \sim P$	3846.9	0.0	1.0
M5	$D \sim x + y$	3871.7	24.8	0.0
M6	$D \sim DTB$	3973.6	126.8	0.0

(AICc refers to the Akaike information criterion with a correction for small sample size.)

Estimated abundance ($\widehat{N}$) of wild dogs within the habitat mask during the pre-baiting period was 48 dogs, with the corresponding abundance during the post-baiting period being 36 (Table 5). These abundances were converted to density estimates of 0.026 and 0.019 dogs/km^2, respectively (Table 5). The change in density between pre- and post-baiting periods equated to a 27% reduction, which was significantly greater than zero [log difference of –0.299, 95% CI; –0.562, –0.037]. The estimated spatial scale parameter (σ) varied from 2.7 km during the pre-baiting period to 3.2 km during the post-baiting period, which corresponded to home range sizes of 137 and 193 km^2, respectively. The daily probability of detection when a camera coincided with the centre of a dog home range was 0.051 (Table 5).

Table 5. Parameter estimates of wild dog population size $\hat{N}$ and density, and parameters of the detection function (g_0, σ) from the spatial mark–resight model applied to detections in camera traps from pre- and post-baiting-period surveys

Period	Parameter	Estimate	SE	2.5%	97.5%
Pre-baiting	$\hat{N}$	48.3	4.27	40.6	57.4
	Density (dogs/km^2)	0.026	0.0023	0.022	0.031
	g_0	0.051	0.0049	0.042	0.062
	σ (km)	2.72	0.151	2.44	3.03
Post-baiting	$\hat{N}$	35.7	3.59	28.4	43.5
	Density (dogs/km^2)	0.019	0.0019	0.016	0.024
	g_0	0.051	0.0049	0.042	0.062
	σ (km)	3.21	0.204	2.83	3.63

Estimates of the locations of dog home range centres (Figure 5) indicated that in the pre-baiting period wild dogs centred their activity primarily in the western and southern parts of the study site and were largely absent from the baited area (Figure 7) This did not change in the post-baiting period.

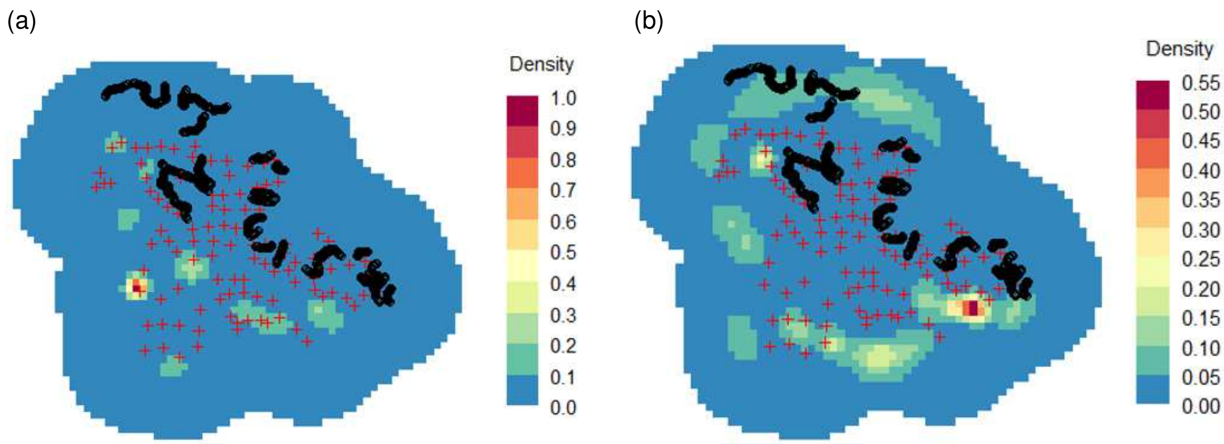

Figure 7. The probability densities of the locations of home range centres of wild dogs during the (a) pre- and (b) post-baiting periods. Note, there is a difference in the density scale between 'a' and 'b'. The black lines are the locations of the bait transects, and the red crosses are the locations of cameras.

3.1.2 Feral cats

Of the 203 feral cat detections, 53 individuals could be uniquely identified based on natural markings from the 26 days of pre-baiting monitoring at an average detection rate of 7.8/day; 65 individuals could be uniquely identified from the 329 detections over the 38 days of post-baiting monitoring at an average detection rate of 8.6/day (Figure 8). In addition, there were a total of 23 and 99 detections of unmarked feral cats in the pre- and post-bait monitoring, respectively. Figure 9 indicates the location and frequency of unmarked feral cat detections pre- and post-baiting.

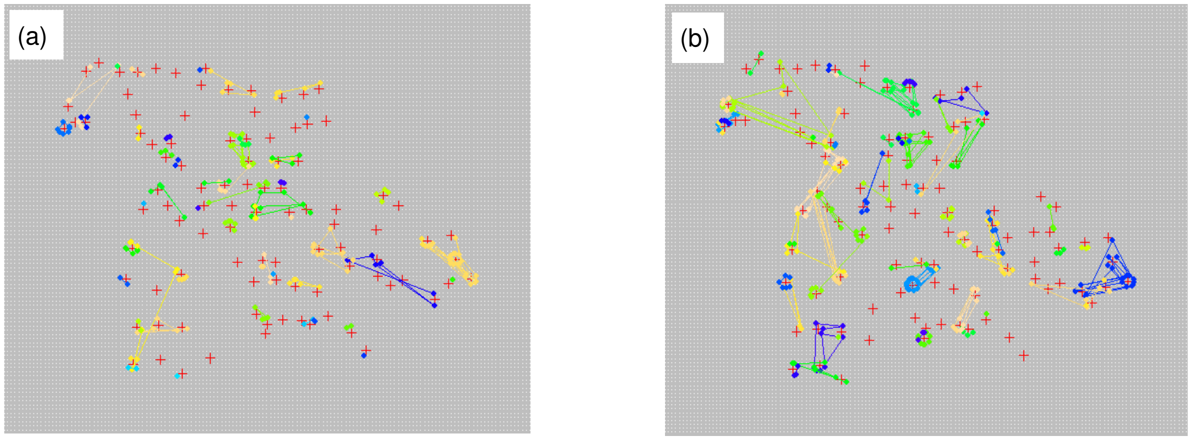

Figure 8. Detections of individually recognisable feral cats in the (a) pre- and (b) post-bait camera trap monitoring. Detections of individual cats are connected by lines with colours representing individuals.

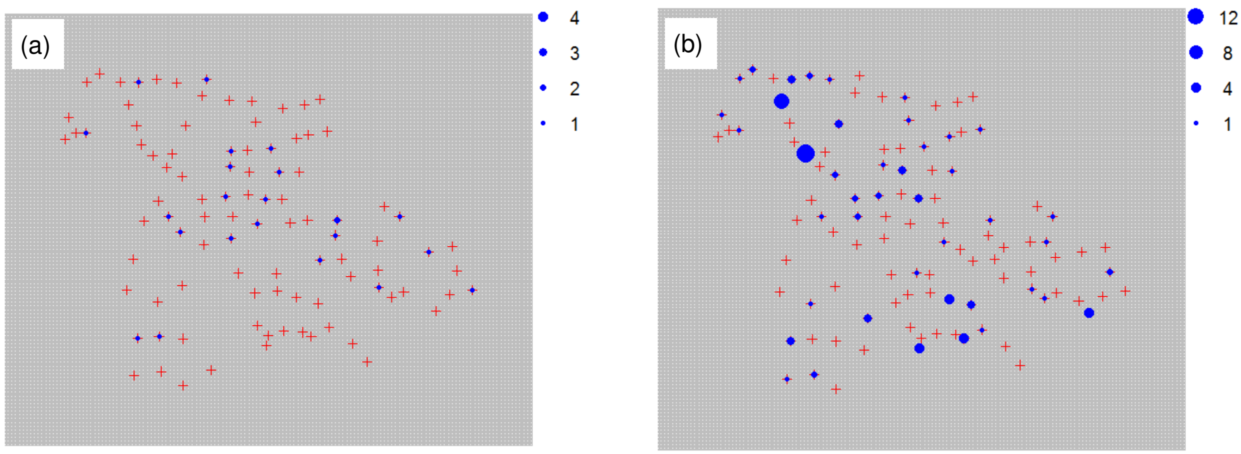

Figure 9. Locations of detections of feral cats that were unidentified (blue circles) during the (a) pre- and (b) post-baiting periods. The size of the circle indicates the number of detections at a camera trap.

Results from the model selection of the detection parameters (g_0, σ) revealed that there was stronger support for the model where the per-occasion detection probability (g_0) and the scale parameter (σ) varied between survey periods (model M4; Table 6). We therefore used model M4 as the basis for the estimation of the overdispersion parameter.

Table 6. Results from the model selection of the detection parameters (g_0, σ). Models varied as to whether the detection parameters (were constant ($\sim\mathbf{1}$) or varied by survey period ($\sim\boldsymbol{P}$).

Model	Model formula	AICc	∆AICc	Weight
M4	$D \sim P$, $g_0 \sim P$, $\sigma \sim P$	8106.2	0.00	0.77
M2	$D \sim P$, $g_0 \sim 1$, $\sigma \sim P$	8110.4	4.23	0.09
M1	$D \sim P$, $g_0 \sim 1$, $\sigma \sim 1$	8111.1	4.84	0.07
M3	$D \sim P$, $g_0 \sim P$, $\sigma \sim 1$	8111.3	5.05	0.06

(AICc refers to the Akaike information criterion with a correction for small sample size.)

The estimated abundance ($\hat{N}$) of feral cats within the habitat mask during the pre-baiting period was 82 (95% CI 72–93) feral cats, with the corresponding abundance during the post-baiting period being 103 (95% CI 88–121) (Table 7). These abundances equated to density estimates of 0.075 and 0.095 cats/km^2, respectively (Table 7). The change in density between pre- and post-baiting periods equated to a 21% increase, which was significantly greater than zero (log difference of 0.02, 95% CL; 0.01, 0.03). The estimated spatial scale parameter (σ) varied from 1.32 km during the pre-baiting period to 1.53 km during the post-baiting period, which corresponded to home range sizes of 1.16 and 1.40 km^2, respectively. The daily probability of detection when a camera coincided with the centre of a feral cat home range was 0.144 pre-baiting and 0.097 post-baiting (Table 7).

Table 7. Parameter estimates of feral cat population size $\hat{N}$ and density, and the detection function (g_0, σ) from the spatial mark–resight model applied to detections in camera traps from pre- and post-baiting surveys

Period	Parameter	Estimate	SE	2.5%	97.5%
Pre-baiting	$\hat{N}$	82	5.2	72.4	92.7
	Density (feral cats/km^2)	0.075	0.005	0.066	0.085
	g_0	0.144	0.017	0.114	0.180
	σ (km)	1.32	0.57	1.21	1.44
Post-baiting	$\hat{N}$	103	8.7	88.1	121.1
	Density (feral cats/km^2)	0.095	0.004	0.081	0.111
	g_0	0.097	0.007	0.081	0.115
	σ (km)	1.53	0.05	1.43	1.64

Estimates of the locations of feral cat home range centres (Figure 10) indicated that feral cats were uniformly spread across the study area both before and after the baiting, also in the post-baiting period home range size decreased and density increased.

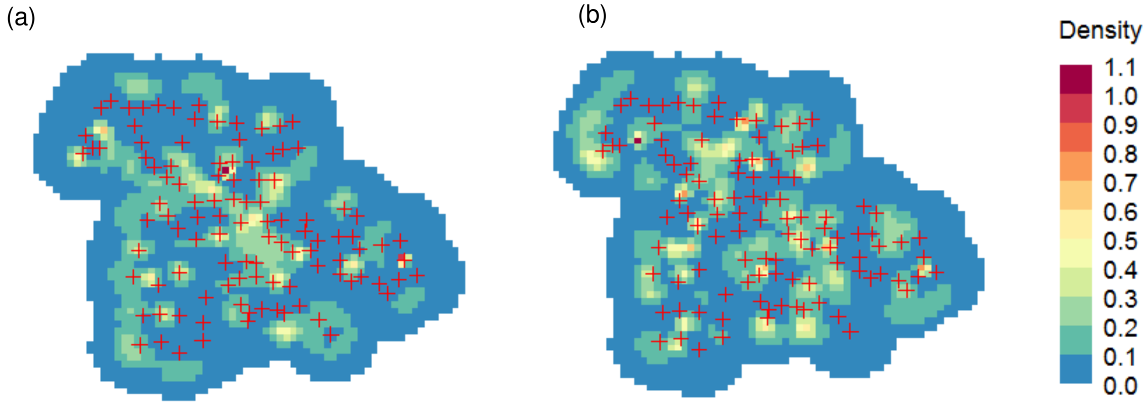

Figure 10. The probability densities of the locations of home range centres of feral cats during the (a) pre- and (b) post-baiting periods. The red crosses are the locations of cameras. There was no expectation that cat would take a bait so DTB was not modelled and bait lines not shown in this figure.

3.1.3 Red foxes

There was a 23% reduction in the fox occupancy rate across the study area from the pre-baiting to the post-baiting period. The initial occupancy was 0.50 (95% CI 0.38–0.62), and following the aerial baiting operation it was 0.41 (95%CI 0.30–0.51). The site colonisation rate was 0.30 (95% CI .017–0.46), i.e. 30% of sites in the post-baiting period recorded fox detections that had no detections during the pre-baiting period. The extinction rate was higher at 0.48 (95% CI 0.32–0.64), i.e. 48% of sites with previous detections failed to detect foxes post-baiting.

The best model for describing changes in fox occupancy allowed for correlation between detection rates, with detection varying between period (Table 8). The chance that a fox had been detected on an adjacent camera was 0.76 (SE 0.04) or 76%, compared with the chance it had not been detected, which was only 0.09 (SE 0.011) or 9%. The distance between camera traps was likely to have been too small for complete independence in detection rates. Generally, overall detection rates were high in both the pre-baiting survey (0.51; SE 0.055) and after the aerial baiting (0.42; SE 0.052).

Table 8. Parameter estimates for changes in fox occupancy following aerial baiting. Models varied as to whether the detection parameters were constant (~1) or varied between survey period (~*P*), and allowed for detections at a site to be correlated with detections at an adjacent site (θ) or not (θ').

Model	Model formula	AIC	ΔAIC	AIC weight
M1	Ψ, θ, θ', γ ~ 1, ε ~ 1, p ~ P	2257.21	0.00	0.506
M2	Ψ, θ, θ', γ ~ 1, ε ~ 1, p ~ 1	2257.26	0.05	0.494
M3	Ψ, γ ~ 1, ε ~1, p ~ P	2427.7	170.5	0.000
M4	Ψ, γ ~ 1, ε ~ 1, p ~ 1	2465.1	207.9	0.000

(AIC refers to the Akaike information criterion.)

4 Discussion

As far as we are aware, the current work is the first occurrence in Australia where the analytical approach to directly assessing density has been applied to estimate and assess changes in wild dog density and abundance resulting from management actions. This study builds on previous research (Forsyth et al. 2018 submitted), which assessed approaches to determining wild dog density at Gudgenby, New South Wales.

We found that wild dog density and abundance were significantly lower (27%) one month following the aerial baiting operation. This finding is lower than that of other assessments of changes in wild dog populations following aerial baiting operations. For example, Thomson (1986) showed that between 40 and 95% of radio-collared dingoes were dead at 4 weeks post-aerial baiting in north western Australia, and Fleming et al. (1996) showed that indices of wild dog abundance were lowered by between 69 and 85% at sites in north-eastern NSW.

A key difference between our study and those above is that we assessed density directly using the detection rates of known individual wild dogs, as well as the detection rates of unidentifiable individuals, and incorporated this information into a modelling approach that determines density, abundance and home range size. It also specifically accounts for the area sampled, which is a key consideration in determining density (Efford 2004). All previous assessments of density have been accomplished by indirect indices or relative density, e.g. visitation rates to watering holes (Fleming et al. 2001), or observations of packs via several radio-tracked individuals (Thomson et al. 1992) and where the sample area has been loosely defined. A key difference between the analytical approach used in our direct density estimates and those used previously is that our approach explicitly incorporates the area sampled, a key component of density (Efford 2004). The differences in analytical approach makes direct comparison of density or changes arising from management actions to previous studies problematic. This is reflected in the recent literature which discusses the issues associated with earlier wild dog/dingo monitoring approaches (Allen et al. 2013, Hayward and Marlow 2014, Nimmo et al. 2015).

Wild dog control actions in Victoria (aerial and ground-based baiting and trapping) are undertaken in the 3 km-wide buffer zone on public land interfacing with private land where livestock attacks and losses have been recorded. Previous studies have shown that wild dogs in eastern Victoria can have home ranges covering much greater areas than this, e.g. 45–124 km^2 (Robley et al. 2010), and our study estimated average home range size before and after aerial baiting to be 137 and 193 km^2, respectively. Differences in methodological approaches may explain the disparity between our results and those of Robley et al. 2010. The earlier study used locations collected from GPS collars fitted to nine wild dogs, with >2000 locations (recorded every 30 minutes for four months between March and June 2007). While the approach used in the *secr* model is to fit the standard deviation of the detected locations about the estimated home range centres Efford and Hunter (2018). Differences in home range may also be associated with the differences in wild dog control history at the two sites. Previous control at our site may have resulted in comparatively lower densities and hence larger home range sizes, while the site Robley et al. (2010) studied had little or no history of coordinated control wild dog density may have been higher and hence home ranges smaller.

It has been shown that foxes have home ranges that cover 2–8 km^2 in Australian forested habitat (Phillips and Catling 1991; Saunders et al. 1995; Diment 2010; Robley et al. 2016). Therefore, it is likely that both wild dogs and foxes outside the buffer zone may be exposed to lethal control actions, including consuming aerially deployed baits. Aerial baiting uses 250 g meat baits injected with 6 mg/kg of 1080 (sodium mono-fluoroacetate) poison; this amount is sufficient to kill both wild dogs and foxes (McIlroy 1981).

We showed that the fox occupancy rate across the study area was lower after the aerial baiting period than before. No studies looking at the impact of aerial baiting on wild dogs have also reported changes in fox populations. However, aerial baiting for the control of foxes is undertaken in other States in Australia using 3 mg/kg of 1080 and has been shown to be effective at reducing fox abundance. For example, Thomson et al. (2000) demonstrated a >95% reduction in fox population in Western Australian rangelands and Moseby et al. (2011) demonstrated a >85% reduction in fox detection rates on sites where aerial baiting had been conducted annually over a 5-year period. It is unclear why in our study we reduced foxes by only 23%, it is possible that foxes were already at low numbers because of the previous wild dog control actions, and hence a 23% reduction in a small population may still represent a significant population level impact.

There is considerable interest in the interactions between wild dogs, foxes and feral cats in Australia (Brook et al. 2012; Allen et al. 2014) with indications that feral cats may respond positively to the removal of larger competitive predators (Brook et al. 2012). Typically baits used for the control of wild dogs and foxes are 250 and 35 g and made from a range of bait matrices including dried boneless meat, chicken and commercially developed products. These baits are generally considered unlikely to be palatable and attractive to feral cats. Algar and Burrows (2004) showed that the preferred bait medium for feral cats was a small 15 g chipolata sized bait. Reduction in the abundance of both wild dogs and foxes could be beneficial to feral cats, potentially releasing feral cats from the direct (predatory) and indirect (competitive) effects of both higher order predators/competitors (Brook et al. 2012). For example, Molsher et al. (2017) showed that, where foxes were removed, feral cats increased their consumption of invertebrates and fresh carrion, decreased their home range size, and foraged in more open habitats. We showed that feral cat density and abundance increased significantly and with a slight increase in home range size following the aerial baiting operation. While the time between before and after baiting was likely to be too short to result in a numerical response either through immigration or births, a possible response by feral cats may be an increase in activity, i.e., feral cats may have responded to the reduction in the presence of wild dogs and foxes by increasing their daily level of activity and doing so across a wider area, increasing their recapture rates at multiple devices

There was a clear correlation between the decline in both wild dogs and foxes, the increase in feral cats and the implementation of the aerial baiting operation. However, as this study was undertaken at a single site with no corresponding non-aerial baiting sites being monitored to act as a control, it is not possible to discount other possible causes or confounding effects or ascribe causation.

The notion of determining causation about treatment effects in a scientifically robust manner relies on the implementation of three fundamental principles; 1) randomisation, 2) replication; and 3) control (Morgan and Winship 2007). Without these fundamental principles in place causation cannot be inferred. Randomly allocating units to be treated avoids underlying bias that may result from the influence of confounding factors. The effect of randomising treatments eliminates or reduce systematic bias due to non-random differences between treated and untreated units with respect to variables that are also related to the outcome. Replication is necessary to estimate average treatment effects, so that inferences are not unduly influenced by any sampling unit. Control groups are necessary to estimate treatment effects by allowing inferences about the sampling units if they were in an untreated (counterfactual) state (Morgan and Winship 2007).

4.1 Conclusion

Our findings that wild dog density and abundance and fox occupancy declined following an aerial baiting operation, and feral cat density and abundance increased are consistent with those of previous studies but were generally of a lesser magnitude.

The outcomes of this current project are limited to the site where the study was undertaken, and they cannot be used to ascribe causality to the observed changes in wild dog, feral cat or fox populations. To more robustly inform current management practices and to provide generalisable outcomes, it will be necessary to repeat this study at another site(s). There are several sites in eastern Victoria with no history of aerial baiting. Such sites could be randomly selected and surveyed, and the results obtained could then be compared with the results for sites at which aerial baiting is undertaken.

Our results fit with what would be predicted from mesopredator theory and the outcomes of published research, i.e., we would have expected feral cat densities to be high and to perhaps increase following a further reduction in the larger wild dogs and foxes. To date much of the published research on mesopredator release and the flow-on effects to native prey have been undertaken in arid and semi-arid environments. The interaction between wild dog control and changes in fox and feral cat abundance needs further investigation in mesic environments like eastern Victoria.

We have demonstrated the utility of the analytical approach for directly assessing wild dog and feral cat density and its application for assessing the outcome of management actions. Introduced predators generally occupy large areas and are cryptic and elusive by nature making monitoring them difficult. The ability to directly assess changes in density is a significant advancement in the capacity of managers to robustly assess the outcome of management interventions, allowing increased understanding of the cost: benefits of various management strategies, and should lead to more informed decisions being made about the effectiveness and efficiency of current and future operations.

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