Camera trapping to monitor the results of predator removal on Waitere Station

Margaret Nichols[1] and Al Glen[2]

[1]Centre for Wildlife Management and Conservation, Lincoln University, Canterbury
[2]Landcare Research, Private Bag 92170, Auckland 1142

Abstract
This study assessed the ability of camera traps as a non-invasive method for monitoring the presence of feral cats. Another objective was to determine the optimal statistical approach to estimate cat abundance from the camera trapping data. Beginning in April 2014, 80 camera traps (40 per site) were placed on two pastoral properties in Hawke's Bay. One property was subject to predator removal while the other was not. Cameras were spaced 500 m apart. The cameras were deployed for a pre-removal monitoring period of three weeks, followed by three weeks of intensive predator removal, and three weeks of post-removal monitoring. Using the resulting data, three different modelling methods were assessed for accuracy and precision: capture-recapture, occupancy modelling, and the newly developed spatial presence-absence (SPA) model. The SPA model is a Bayesian mark-recapture model that does not require all individuals to be identified. These models were compared to the established standard of the Caughley index-manipulation-index method. The SPA model gave robust estimates of change in cat abundance following predator removal. The occupancy model also showed significant decrease in cat presence. The optimal spacing between each camera trap within the grid was also determined by statistically removing every second camera, both in a 'hollow grid' model and every other camera in the grid. With these results we hope to provide an affordable and robust procedure for monitoring feral cats that may be implemented in the *Cape to City* programme.

Introduction
Efficient monitoring is vital to the success of invasive species management (Bengsen et al., 2012; Gompper et al., 2006; Long et al., 2007; Sweetapple and Nugent, 2011). Complete counts of a population are seldom possible, especially for hard to identify species (Ball et al., 2005; Edwards et al., 2000). Making inferences into the relative abundance of a species by sampling subsets of the population becomes vital for monitoring in an area (Pickerell et al., 2014). Rare and/or cryptic species require novel monitoring methods to be detected and circumvent this limitation (Glen et al., 2013; Kelly and Holub, 2008). Surveys can use a variety of index techniques including hair traps, tracking tunnels, scat surveys, chew cards, leg-hold traps and camera traps (Ball et al., 2005). One common issue for population density estimates is that the percentage of detected/captured individuals is an unknown portion of those present (Chandler and Royle, 2013).

Camera trapping is emerging as a useful, non-invasive method for monitoring elusive species (De Bondi et al., 2010; Rowcliffe and Carbone, 2008; Silveira et al., 2003; Smith and Coulson, 2012). However, methods to estimate animal abundance from camera traps often require identification of individual animals (e.g. using unique coat patterns; Karanth, 1995a), while other methods are costly (e.g. Rowcliffe et al., 2008) or estimate only an index of relative abundance (e.g. Bengsen et al., 2011). Recent advances in modelling have identified approaches that may allow affordable, accurate monitoring of cryptic species using camera traps, without the need for individual identification. We compared two recently developed methods – dynamic occupancy modelling (Bengsen et al., 2014) and spatial presence-absence modelling (Ramsey et al., in press) – against an established estimator: the index-manipulation-index method (Caughley, 1976). The aim was to identify an accurate, precise and cost-effective method for estimating cat abundance in the *Cape to City* area in Hawke's Bay.

Overview of available modelling approaches

Capture-recapture

Capture-mark-recapture has long been a staple method in monitoring a variety of carnivore species (Chapman and Balme, 2010). However, the intensive labour involved in physically capturing, marking and releasing animals has led to the expanse of non-invasive methods such as camera trapping (Karanth, 1995b; Karanth and Nichols, 1998). This method of capture-recapture performs well to monitor elusive species that may become trap shy (such as most felids) (Chapman and Balme, 2010; Karanth, 1995b; Karanth and Nichols, 1998). The first studies to use capture-recapture methods with camera traps monitored tigers (*Panthera tigris*) (Karanth, 1995b; Karanth and Nichols, 1998). While large felids such as tigers (Karanth, 1995b), Jaguars (*Panthera onca*) (Soisalo and Cavalcanti, 2006), and Snow leopards (*Uncia uncia*) (Jackson et al., 2006) are easily identifiable by their unique coat patterns, the same may not apply to populations that lack clear and unique markings (Chandler and Royle, 2013).

Occupancy modelling

Occupancy can be defined as the proportion of sites throughout a landscape that are inhabited by a target species (Field et al., 2005). Occupancy modelling is a practical approach for wide scale monitoring due to the moderately low-cost of collecting detection/non-detection data (Jones, 2011). While the presence of a particular species may be relatively uncomplicated to determine, it is usually impossible to accurately confirm a species' absence (MacKenzie, 2005). An observed absence may simply be the effect of the monitoring method failing to detect the species although it is present (MacKenzie, 2005). Accordingly the occupancy of many rare, cryptic species at low densities may be frequently underestimated (Gu and Swihart, 2004); which leads to misleading assumptions about a system (MacKenzie, 2005).

The spatial presence-absence (SPA) model

The present study aims to test an extension of the spatial capture-recapture model (Chandler and Royle, 2013) to monitor feral cat abundance in rural Hawke's Bay, NZ. The spatial presence-absence (SPA) model allows for a variety of non-invasive devices; such as bait stations or camera traps, to sample individual encounters that are non-independent (Ramsey et al., in press). It estimates the spatial detection parameters (g0 and sigma) of the target species.

Home range size is taken into account while deploying sampling devices so that multiple devices are assumed to be encountered by a single individual (Ramsey et al., in press).

Optimal grid spacing

Appropriate spatial placement of camera traps is especially important for population estimates (Meek et al., 2014). There are now basic guidelines regarding camera placement (Meek et al., 2014) as in systematic (along pre-determined transects), random allocation, or deliberately biased placement. The spacing between camera trap sites determines the independence of observations between locations (Meek et al., 2014). Approximate home range should also be taken into account when determining trap placement (Bengsen et al., 2012) (whether looking for independence or non-independence among sites as in Ramsey et al., in press). For example, feral cats in New Zealand pastoral locations are thought to have home ranges of approximately 1–2 km^2 (Langham and Porter, 1991). Based on this information, placing cameras 500 m apart would allow for non-independence among samples (Ramsey et al., in press). However, the possibility of reducing the numbers of cameras for future studies aiming to achieve the same monitoring goals has great potential to reduce costs and therefore needs to be examined.

Materials and methods

Study sites

Toronui and Waitere stations are pastoral farms in Hawke's Bay, North Island, New Zealand (~39º S, 176º E) with small patches of native bush throughout. With no recent history of predator control, Toronui Station was used as the non-treatment site, and Waitere station as the treatment area. The study took place from early April through to early June 2014. In total, 80 Reconyx PC 900 (RECONYX Inc, Holmen, Wisconsin) cameras were deployed (40 each site) in a grid with 500-m spacing. However, there was a lenience of 100 m at each site in case of hazardous terrain/close proximity to livestock/roads. Cameras were mounted on wooden stakes with the base of each camera sitting 5 cm from the ground. All cameras were set to take a series of three photos per trigger. A lure of ferret odour and rabbit meat (Garvey et al., submitted) was placed in a vial 1.5 m in front of the camera, and secured with a tent peg. Cameras were deployed for a total of nine weeks. Intensive predator removal was carried out in weeks 4–6 of camera deployment. Specialist trappers removed cats using a combination of cage, leg-hold and kill traps. Live traps were checked daily soon after sunrise, and captured animals humanely destroyed. The predator removal was part of a routine management program by the Hawke's Bay Regional Council. The 3-week periods will be referred to as 'pre-removal', 'removal' and 'post-removal'.

Sampling design and methods

Feral cats have relatively large home ranges (e.g. Bengsen et al., 2012). With camera traps positioned 500 m apart, individual cats are likely to be detected at more than one camera location. This allows the use of recently developed spatial capture-recapture (SCR) models, which assume individual cats would encounter multiple cameras (Chandler and Royle, 2013; Ramsey et al., in press).

Analysis

We analysed the data from the camera trapping using a recently developed spatially explicit model of presence/absence data (Ramsey et al., in press). This model relaxes the assumption of independence among sampling locations and instead relies on individuals to be detected at more than one location. A spatially-explicit model of the detection probabilities is fitted to the detection data, enabling estimates of abundance as well as the spatial detection parameters g0 and sigma.

The model was fitted using JAGS 3.3.0 (Plummer 2003) called from R 3.1.1 (R Development Core Team 2014) using code from Ramsey et al. (in press). Due to a shortage of published information on the capture probabilities of cats, a vague prior was placed on g0. Based on a review of published information on cat home ranges and movements (Glen and Byrom, 2014), an informative prior was used for sigma (Figure 1). The upper limit of population size N (used for data augmentation; Ramsey et al. in press) was 200 for each area.

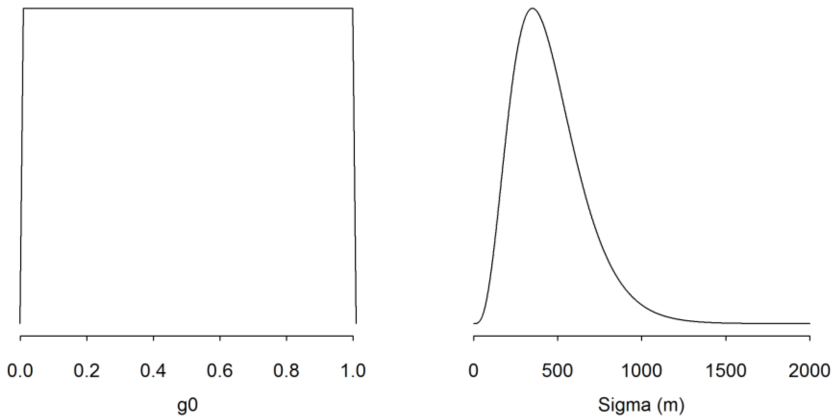


Figure 1: Priors used for g0 (a vague or uninformative prior) and sigma (an informative prior).

To assess whether the number of cameras could be reduced and still allow the model to converge, the SPA model was employed again using half the number of cameras (20). This was done in two ways: (1) data from every second camera were discarded, simulating a grid of 20 cameras spaced 1 km apart; (2) data from cameras in the interior of the grid were discarded, leaving only the data from cameras on the perimeter, simulating a 'hollow grid' (*sensu* Efford et al. 2005) of 20 cameras. The aim was to determine if the model could still estimate a significant reduction in cat abundance (n) post predator control using fewer cameras.

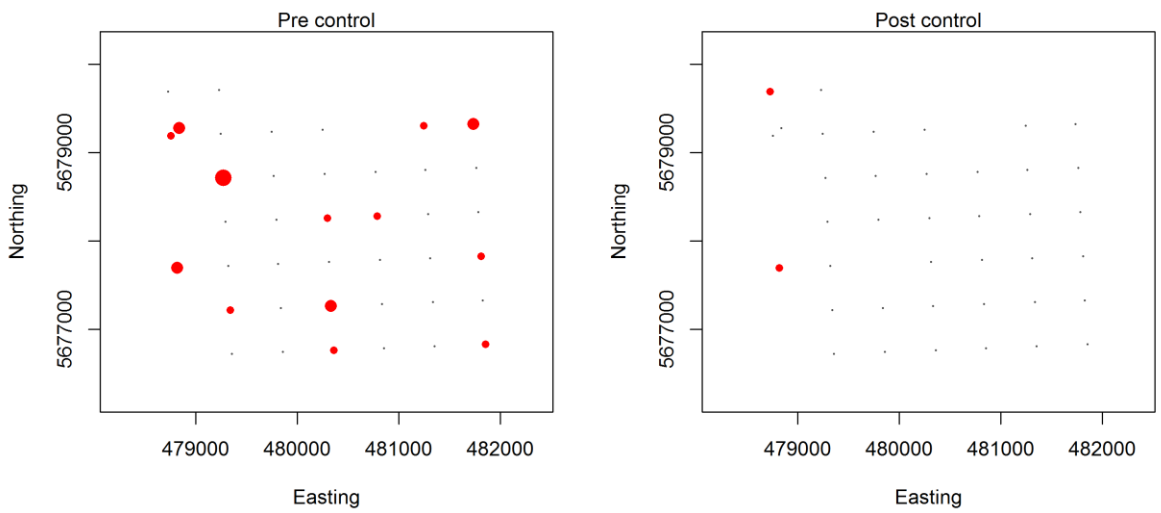
We also used a dynamic occupancy model (Bengsen et al., 2014) to detect change in cat presence over time (pre- and post-removal). This is an extension of occupancy modelling (MacKenzie et al., 2002). A generalised linear mixed model (GLMM) with a Poisson-logit distribution was used to estimate variability in cat presence before and after the removal period. The model was fitted in R 3.1.1 (R Development Core Team 2014) using code from Bengsen et al. (2014).

Results

Of the 80 cameras deployed, 77 remained operative throughout the study period. During the post-removal period two cameras failed on Toronui Station and one on Waitere. The specialist trappers removed 17 cats from Waitere Station during the removal period.

On Waitere, cats were detected on 19 occasions at 13 of 40 locations during the pre-removal period. In the post-removal period 2 cats were detected at 2 of 39 locations (Figure 3a). On Toronui, cats were detected on 39 occasions at 20 of 40 locations in the pre-removal, and on 39 occasions at 21 of 38 locations in the post-removal period (Figure 3b).

a)



b)

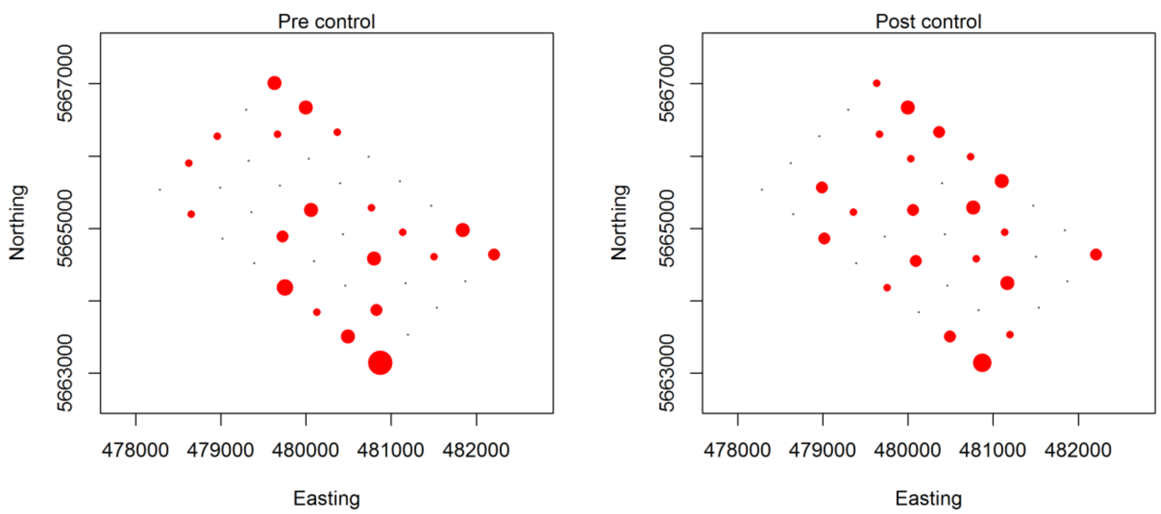


Figure 3: Cat detections on Waitere (a) and Toronui (b), pre- and post-removal. Black dots indicate no cat detections, whist red circles indicate positive detections with larger circles indicating more detections per camera location.

Waitere Station

Removal of 17 cats led to a reduction of 89% in the number of cat detections, and an 85% reduction in the number of cameras detecting cats. If 17 cats represented 85–89% of the population, the index-manipulation-index method (Caughley, 1976) estimates there were 19–20 cats on Waitere in the pre-removal period. Two or three cats are therefore estimated to have survived.

The occupancy model showed a significant decrease in cat presence after predator removal (P = 0.04). Detection probability ranged from 0.07 to 1.85% for the pre-removal period and 0.01 to 0.72% post-removal (Figure 2).

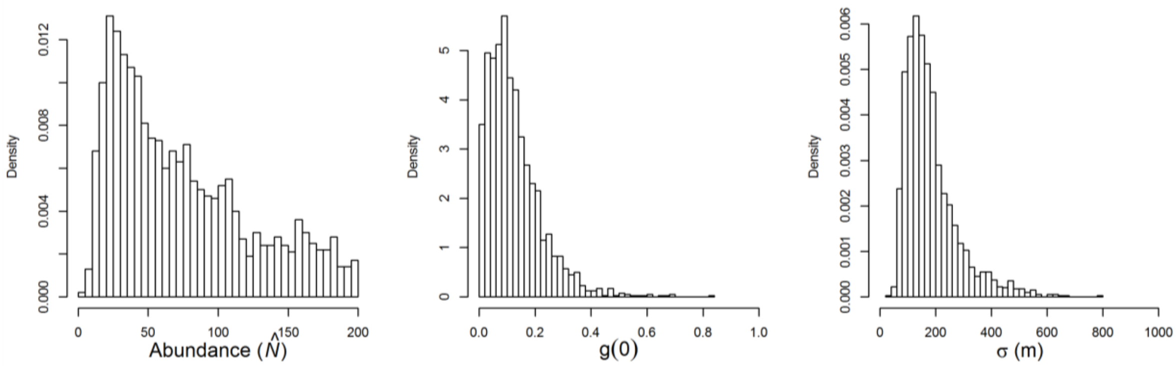
Figure 2: Probability of capture of feral cats per camera (± 95% confidence interval) before and after predator removal on Waitere Station. Data have been transformed onto the logit scale, hence the overlapping error bars.

The SPA model (Ramsey et al., in press) estimated a population of 24 cats on Waitere during the pre-removal period, and 3 cats in the post-removal period (Table 1; Figure 3).

Table 1: Estimates of mode for N and means for *g*0 and sigma, and 95% confidence intervals

Parameter	Pre	Post
N	24 (13 – 185)	3 (2 – 184)
g0	0.012 (0.010 – 0.360)	0.074 (0.001 – 0.696)
Sigma	182.5 (71.7 – 453.1)	264.9 (63.9 – 627.2)

(a)



(b)

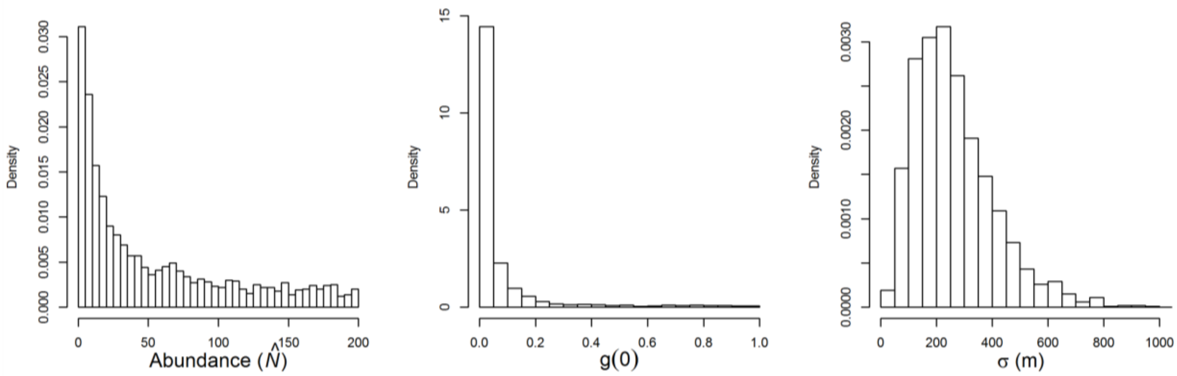


Figure 3: Estimated abundance (N) and spatial detection parameters (g0 and σ) of feral cats on Waitere Station in the pre-removal (a) and post-removal (b) periods.

Toronui Station

As no predator removal occurred on Toronui, the index-manipulation-index method could not be applied. The SPA model estimated 30 cats were present in the pre-removal period. During the post-removal period the SPA model failed to converge (Table 2; Figure 4).

Table 2: Estimates of mode for N and means for *g*0 and sigma, and 95% confidence intervals

Parameter	Pre	Post
N	30 (19 – 186)	64 (29 – 193)*
g0	0.224 (0.086 – 0.449)	0.148 (0.031 – 0.323)
Sigma	179.5 (83.9 – 311.7)	96.5 (366.1)

*Estimate of N in the post-removal period may be unreliable as the model did not converge.

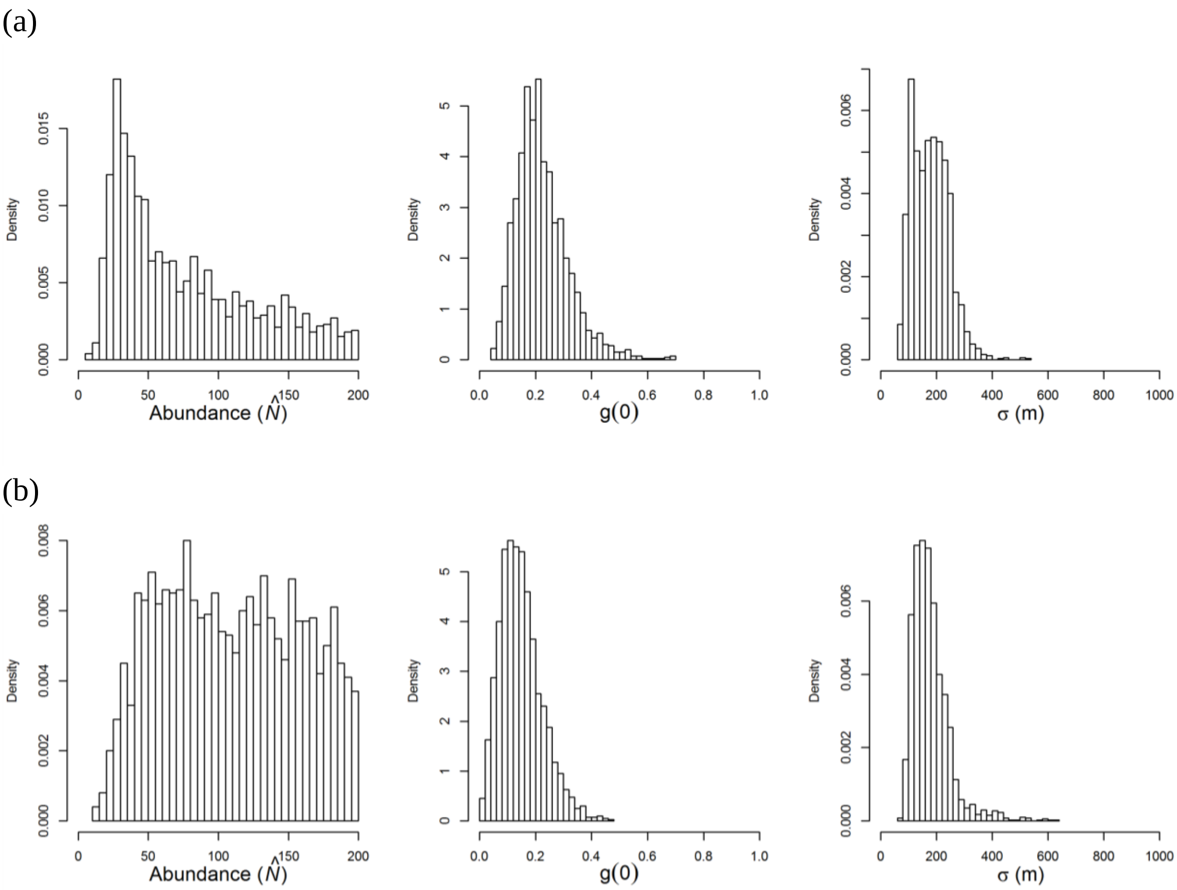


Figure 4: Estimated abundance (N) and spatial detection parameters (g0 and σ) of feral cats on Toronui Station in the pre-removal (a) and post-removal (b) periods.

Optimal grid spacing

When a 50% reduction in cameras was used to represent a grid of 20 cameras spaced 1 km apart, occupancy modelling still detected a significant reduction in cats on Waitere station (P= 0.04) (Figure 5).

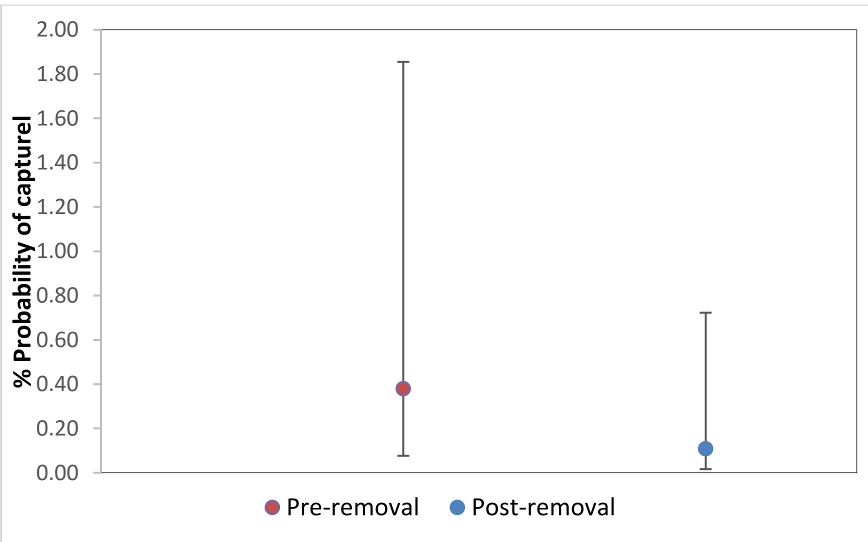


Figure 5: Probability of capture of feral cats per camera (± 95% confidence interval) before and after predator removal on Waitere Station, using data from 20 cameras (every second camera in grid).

Hollow grid model

The hollow grid model was taken from a previous study by Efford et al. (2005), whereby a perimeter of trap sites are used in a square grid, leaving an empty area within. Again there was a 50% reduction in cameras, however only the perimeter cameras on the Waitere grid were sampled. This hollow grid still allowed the occupancy model to detect a significant reduction in cats on Waitere post-removal (P= 0.01).

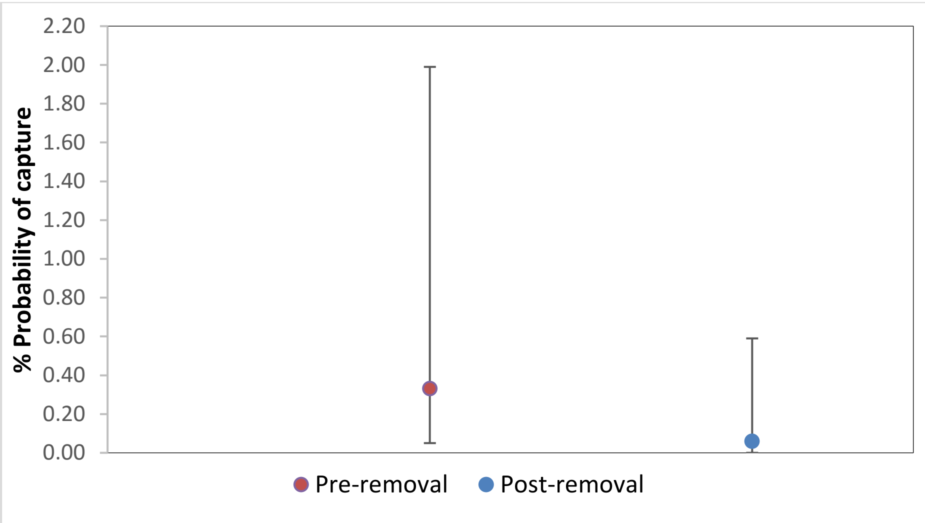


Figure 6: Probability of capture of feral cats per camera (± 95% confidence interval) before and after predator removal on Waitere Station, using data from 20 cameras arranged in a hollow grid.

Discussion

Camera trapping is useful as a non-invasive method for detecting a range of different species (De Bondi et al., 2010). However, progressive monitoring methods do not necessarily resolve the non-detection issue (MacKenzie, 2005). Additionally, cryptic species and individuals with unclear markings may be under-represented in recent camera trapping research (Rowcliffe et al., 2008). When imperfect detection occurs, parameter estimators are biased; often leading to exaggerated results (MacKenzie, 2005). We must also note that the study season (April-June) is peak immigration for feral cats. If the remaining undetected cats were actually new individuals emigrating from a different area, then the model actually underestimated the strength of the control operation.

There are a number of unobservable causes for variation among detection rates; such as behavioural preferences and overall abundance (Dorazio and Royle, 2005). Using the index-manipulation-index method (Caughley, 1976), we derived a robust estimate of the cat population on Waitere Station. This method was used as a benchmark to compare with two alternative modelling methods (occupancy and the SPA model).

The SPA model and the index-manipulation-index method gave similar estimates of cat abundance in the pre-removal and post-removal periods. We can assume from these results that the approximate 500-m spacing of camera traps is still within the optimal range for cats to encounter multiple sites. We also ran the SPA model using half the number of cameras for each site (20), first removing every second camera in the grid, then using a hollow grid (Efford et al., 2005). While dynamic occupancy modelling (Bengsen et al., 2014) using data from 20 cameras showed a significant reduction in cat presence, the SPA model was unable to converge with less data. Thus we can determine that any fewer than 40 cameras with 500-m spacing will be too few to model cat abundance estimates. The SPA model also failed to converge for Toronui in the post-removal period, suggesting that even 40 cameras may be too few in some instances. Although occupancy and abundance are not directly comparable, it is encouraging that both measures fell by approximately an order of magnitude after predator removal. Although calculating presence does not provide the detailed estimates that the SPA model is capable of, this seems to be a sufficient method of determining whether a removal operation has had a significant effect on a population (Bengsen et al., 2014).

Future research

Further analyses will attempt to estimate cat abundance using traditional capture-mark-recapture methods, identifying individual cats by their coat patterns. These results will be compared with the results presented here. The SPA model is also capable of using information on the identity of individual animals (Ramsey et al., in press). SPA modelling will be repeated using coat patterns to identify individual cats where possible. This may improve the precision of the population estimates.

Conclusions

This trial has shown that camera traps are an effective tool to measure changes in relative and/or absolute abundance of feral cats in response to management. Camera trap data may be analysed in various ways to estimate cat abundance. While the SPA model can estimate animal abundance at a localised scale of a few square kilometres, it requires intensive sampling and large amounts of data. Occupancy modelling, on the other hand, can provide reliable evidence of a population reduction in response to management, and can do so using fewer cameras. Further trials should test the effectiveness of camera trapping and occupancy modelling over the larger spatial scale of the *Cape to City* area.

Acknowledgements

This study was funded by Hawke's Bay Regional Council and by Core Funding from the Ministry of Business, Innovation and Employment. We thank A. Gormley, D. Ramsey and J. Ross for assistance with the analyses.

References

Ball, S.J.; Ramsey, D.; Nugent, G.; Warburton, B.; Efford, M. 2005. A method for estimating wildlife detection probabilities in relation to home-range use: insights from a field study on the common brushtail possum (*Trichosurus vulpecula*). *Wildlife Research 32*: 217-227.

Bengsen, A.; Butler, J.; Masters, P. 2011. Estimating and indexing feral cat population abundances using camera traps. *Wildlife Research 38*: 732-739.

Bengsen, A.; Robinson, R.; Chaffey, C.; Gavenlock, J.; Hornsby, V.; Hurst, R.; Fosdick, M. 2014. Camera trap surveys to evaluate pest animal control operations. *Ecological Management & Restoration 15*: 97-100.

Bengsen, A.J.; Butler, J.A.; Masters, P. 2012. Applying home-range and landscape-use data to design effective feral-cat control programs. *Wildlife Research 39*: 258-265.

Caughley, G. 1976. *Analysis of vertebrate populations*. Wiley, London.

Chandler, R.B.; Royle, J.A. 2013. Spatially explicit models for inference about density in unmarked or partially marked populations. *The Annals of Applied Statistics 7*: 936-954.

Chapman, S.; Balme, G. 2010. An Estimate of Leopard Population Density in a Private Reserve in KwaZulu-Natal, South Africa, using Camera—Traps and Capture-Recapture Models. *South African Journal of Wildlife Research 40*: 114-120.

De Bondi, N.; White, J.G.; Stevens, M.; Cooke, R. 2010. A comparison of the effectiveness of camera trapping and live trapping for sampling terrestrial small-mammal communities. *Wildlife Research 37*: 456-465.

Dorazio, R.M.; Royle, J.A. 2005. Estimating Size and Composition of Biological Communities by Modeling the Occurrence of Species. *Journal of the American Statistical Association 100*: 389-398.

Edwards, G.P.; de Preu, N.D.; Shakeshaft, B.J.; Crealy, I.V. 2000. An evaluation of two methods of assessing feral cat and dingo abundance in central Australia. *Wildlife Research 27*: 143-149.

Efford, M.G.; Warburton, B.; Coleman, M.C.; Barker, R.J. 2005. A Field Test of Two Methods for Density Estimation. *Wildlife Society Bulletin 33*: 731-738.

Field, S.A.; Tyre, A.J.; Possingham, H.P. 2005. Optimizing allocation of monitoring effort under economic and onservational constraints. *Journal of Wildlife Management 69*: 473-482.

Garvey, P.M.; Pech, R.P.; Glen, A.S. submitted. Dominant predator odours trigger caution, but not avoidance in a mesopredator: Stoat responses to ferret, cat and African wild dog kairomones.

Glen, A.S.; Byrom, A.E. 2014. *Implications of Landholder Buy-in for the Success of Regional-scale Predator Control: Part 1: Review of Predator Movements*. Landcare Research, Lincoln.

Glen, A.S.; Cockburn, S.; Nichols, M.; Ekanayake, J.; Warburton, B. 2013. Optimising camera traps for monitoring small mammals. *PLoS ONE 8*: e67940.

Gompper, M.E.; Kays, R.W.; Ray, J.C.; Lapoint, S.D.; Bogan, D.A.; Cryan, J.R. 2006. A comparison of noninvasive techniques to survey carnivore communities in northeastern North America. *Wildlife Society Bulletin 34*: 1142-1151.

Gu, W.; Swihart, R.K. 2004. Absent or undetected? Effects of non-detection of species occurrence on wildlife–habitat models. *Biological Conservation 116*: 195-203.

Jackson, R.M.; Roe, J.D.; Wangchuk, R.; Hunter, D.O. 2006. Estimating Snow Leopard Population Abundance Using Photography and Capture-Recapture Techniques. *Wildlife Society Bulletin 34*: 772-781.

Jones, J.P.G. 2011. Monitoring species abundance and distribution at the landscape scale. *Journal of Applied Ecology 48*: 9-13.

Karanth, K.U. 1995a. Estimating tiger *Panthera tigris* populations from camera-trap data using capture-recapture models. *Biological Conservation 71*: 333-338.

Karanth, K.U. 1995b. Estimating tiger Panthera tigris populations from camera-trap data using capture—recapture models. *Biological Conservation 71*: 333-338.

Karanth, K.U.; Nichols, J.D. 1998. Estimation of Tiger Densities in India Using Photographic Captures and Recaptures. *Ecology 79*: 2852-2862.

Kelly, M.J.; Holub, E.L. 2008. Camera Trapping of Carnivores: Trap Success among Camera Types and across Species, and Habitat Selection by Species, on Salt Pond Mountain, Giles County, Virginia. *Northeastern Naturalist 15*: 249-262.

Langham, N.P.E.; Porter, R.E.R. 1991. Feral cats (*Felis catus* L.) on New Zealand farmland. I. Home range. *Wildlife Research 18*: 741-760.

Long, R.A.; Donovan, T.M.; Mackay, P.; Zielinski, W.J.; Buzas, J.S. 2007. Comparing scat detection dogs, cameras, and hair snares for surveying carnivores. *Journal of Wildlife Management 71*: 2018-2025.

MacKenzie, D.I. 2005. What are the issues with presence-absence data for wildlife managers? *Journal of Wildlife Management 69*: 849-860.

MacKenzie, D.I.; Nichols, J.D.; Lachman, G.B.; Droege, S.; Royle, J.A.; Langtimm, C.A. 2002. Estimating site occupancy rates when detection probabilities are less than one. *Ecology 83*: 2248-2255.

Meek, P.; Ballard, G.; Claridge, A.; Kays, R.; Moseby, K.; O'Brien, T.; O'Connell, A.; Sanderson, J.; Swann, D.; Tobler, M. 2014. Recommended guiding principles for reporting on camera trapping research. *Biodiversity and Conservation 23*: 2321-2343.

Pickerell, G.A.; O'Donnell, C.F.; Wilson, D.J.; Seddon, P.J. 2014. How can we detect introduced mammalian predators in non-forest habitats? A comparison of techniques. *New Zealand Journal of Ecology 38*: 86-102.

Ramsey, D.S.L.; Caley, P.A.; Robley, A. in press. Estimating population density from presence-absence data using a spatially-explicit model. *Journal of Wildlife Management*.

Rowcliffe, J.M.; Carbone, C. 2008. Surveys using camera traps: are we looking to a brighter future? *Animal Conservation 11*: 185-186.

Rowcliffe, J.M.; Field, J.; Turvey, S.T.; Carbone, C. 2008. Estimating Animal Density Using Camera Traps without the Need for Individual Recognition. *Journal of Applied Ecology 45*: 1228-1236.

Silveira, L.; Jácomo, A.T.A.; Diniz-Filho, J.A.F. 2003. Camera trap, line transect census and track surveys: a comparative evaluation. *Biological Conservation 114*: 351-355.

Smith, J.K.; Coulson, G. 2012. A comparison of vertical and horizontal camera trap orientations for detection of potoroos and bandicoots. *Australian Mammalogy 34*: 196-201.

Soisalo, M.K.; Cavalcanti, S.M.C. 2006. Estimating the density of a jaguar population in the Brazilian Pantanal using camera-traps and capture–recapture sampling in combination with GPS radio-telemetry. *Biological Conservation 129*: 487-496.

Sweetapple, P.; Nugent, G. 2011. Chew-track-cards: a multiple-species small mammal detection device. *New Zealand Journal of Ecology 35*: 153.