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RESEARCH

Predator control on farmland for biodiversity conservation: a case study from Hawke's Bay, New Zealand

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Abstract: Invasive predator control to protect native fauna usually takes place in native habitat. We investigated the effects of predator control across 6000 ha of multi-tenure, pastoral landscape in Hawke's Bay, North Island, New Zealand. Since 2011, low-cost predator control has been conducted using a network of kill traps for mustelids (*Mustela* spp.), and live trapping for feral cats (*Felis catus*). Although not deliberately targeted, other invasive mammals (particularly hedgehogs *Erinaceus europaeus*) were also trapped. We monitored predators and native prey in the predator-removal area and an adjacent non-treatment area. Predator populations were monitored using large tracking tunnels, which also detected native lizards. Invertebrates were monitored using artificial shelters (weta houses). Occupancy modelling showed that site use by cats and hedgehogs was significantly lower in the predator-removal area than in the non-treatment area. Site use by mustelids also appeared to be lower in the treatment area, although sample sizes were too small to allow firm conclusions. Site use by invasive rats (*Rattus* spp.) was higher in the treatment area, while that of house mice (*Mus musculus*) showed no difference between treatments. There was evidence of positive responses of some native biodiversity, with site use by native lizards increasing significantly in the treatment area, but not in the non-treatment area. Counts of native cockroaches were higher in the treatment area, but other invertebrates were detected in similar numbers in both areas. Our results show that low-cost predator control in a pastoral landscape can reduce invasive predator populations, with apparent benefits for some, but not all, native fauna.

Keywords: feral cat, invasive predators, invertebrates, landscape-scale, lizards, mustelids, rodents

Introduction

Invasive predators are controlled to protect native fauna in many parts of New Zealand (e.g. Innes et al. 1999; Reardon et al. 2012; Russell et al. 2015). However, predator control is usually in conservation reserves, wildlife sanctuaries or remnants of native habitat. Few published studies have investigated the effects of controlling predators for conservation purposes in multi-tenure, pastoral landscapes.

Although landscape-scale predator control may be desirable, financial and logistical challenges often prevent it. The tools and techniques used to control predators at localised scales (e.g. exclusion fencing; Innes et al. 2012; Hayward et al. 2014) may be prohibitively expensive at the landscape scale (Norbury et al. 2014). Managing wildlife across different land tenures can also be challenging, both logistically and socially. Access to private property may not always be feasible, and landholders may vary in their attitudes towards proposed management activities (Epanchin-Niell et al. 2009; Glen et al. 2017). Practical and affordable methods are needed to reduce the impacts of invasive predators across multi-tenure, pastoral landscapes.

We controlled invasive predators over 6000 ha of farmland with fragments of native bush adjacent to an 800-ha conservation reserve where intensive predator control had been in place since 1996. The primary targets of the trapping were feral cats (*Felis catus*) and mustelids (stoats *Mustela erminea,* ferrets *M. furo* and weasels *M. nivalis*); however, large numbers of other invasive mammals, particularly hedgehogs (*Erinaceus europaeus*), were also trapped. By removing invasive predators from a pastoral landscape with fragments of native forest, we aimed to facilitate recovery of native fauna. We predicted that predator control would lead to increased abundance and distribution of native prey species. For example, many of New Zealand's lizard and invertebrate taxa have declined due to the impacts of mammalian predators (Hitchmough et al. 2010;

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Stringer & Hitchmough 2012). Here we describe trends in predator populations and native biodiversity following this landscape-scale intervention.

Methods

Study area

Our study took place on four adjacent pastoral properties in Hawke's Bay, North Island, New Zealand: Opouahi, Rangiora, Toronui and Rimu Stations (39⁰10′ S; 176⁰46′ E). These sheep and cattle stations are mainly covered by introduced pasture grass with fragments of native beech forest (*Fuscospora solandri*). Beech forest fragments range in size from about 10 to 100 ha. Adjoining the study area to the north is Boundary Stream Mainland Island, an area of mixed broadleaf and podocarp forest managed by the Department of Conservation (DOC). Elevation in the study area ranges from about 300 to 1000 m a.s.l. Invasive predators have been controlled in Boundary Stream since 1996 as part of DOC's Mainland Island programme (Saunders & Norton 2001; Abbott et al. 2013). There was no recent history of predator control on the adjacent pastoral properties. Predator control was implemented on Opouahi and Rangiora Stations while Toronui and Rimu Stations were non-treatment areas (Fig. 1). The treatment and non-treatment areas were similar in terms of habitat, although the treatment area was directly adjacent to Boundary Stream, where invasive predators are also suppressed.

Predator control

Predator control was conducted by Hawke's Bay Regional Council (HBRC). In November 2011, 680 kill traps were deployed across an area of 6000 ha. These included 550 DOC-250 traps (DOC, Wellington) for mustelids, and 130 Timms traps (KBL Rotational Moulders, Palmerston North) for cats. Traps were spaced 100 m apart in bush fragments or 200 m apart on cleared farmland, based on the assumption that predators were more likely to be found in bush fragments (e.g. Alterio et al. 1998; King & Murphy 2005; Harper 2007; Garvey 2016). Traps were baited with various combinations of fresh rabbit meat, a rabbit-based paste (Erayz®, Connovation Ltd, Auckland) or a synthetic, rat-scented lure (Goodnature Ltd, Wellington). To minimise labour costs, traps were set in locations that were easily accessible by an all-terrain vehicle (ATV). The DOC-250 traps were modified to include a built-in handle for quick re-setting. The position of the handle also served as a visual signal to indicate whether the trap had been triggered, eliminating the need to inspect each trap closely. These modifications were refined during the course of the project, and will be described in detail in a separate publication. Traps were initially checked every 3 weeks at an annual cost of \$5.53 ha^{-1}; however, from November 2014, they were checked four times a year (January, April, June and November), which cost \$2.30 ha^{-1} (HBRC, unpubl. data).

The DOC-250 traps were left in place for the duration of the study. The Timms traps were left in place for the first year, after which cat control was conducted in two annual pulses (May and August each year). The pulsed cat control was carried out using a combination of live traps (cage (Havahart Traps, Lititz, Pennsylvania), leg-hold (Victor #1$^{1/2}$ soft-catch, Oneida Victor, Cleveland, Ohio)), kill traps (Timms and Possum Master traps (Possum Master Industries, Tauranga)), and opportunistic shooting. Live traps were checked daily and captured predators were euthanased. Cat control targeted areas of high rabbit (*Oryctolagus cuniculus*) activity as rabbit abundance is a strong predictor of cat abundance (Norbury & McGlinchy 1996; Norbury et al. 2002; Cruz et al. 2013).

Monitoring

In October 2011, we established 15 monitoring lines in the treatment area and 14 lines in the non-treatment area to assess trends in populations of invasive predators and native prey. However, due to access restrictions, the number of monitoring lines in the non-treatment area was reduced to 12 from spring 2014 onwards. Each line consisted of five large tracking tunnels (see below) spaced 100 m apart, spanning the interface between a native bush fragment and the adjacent pasture. The first point was inside the bush fragment, 200 m from the edge, the third point was on the edge of the fragment, and the fifth point was in cleared pasture, 200 m outside the fragment. Where possible, monitoring lines were at least 1 km apart to improve spatial independence; however, steep topography made this impracticable in some cases. The shortest distance between any two monitoring lines was 500 m.

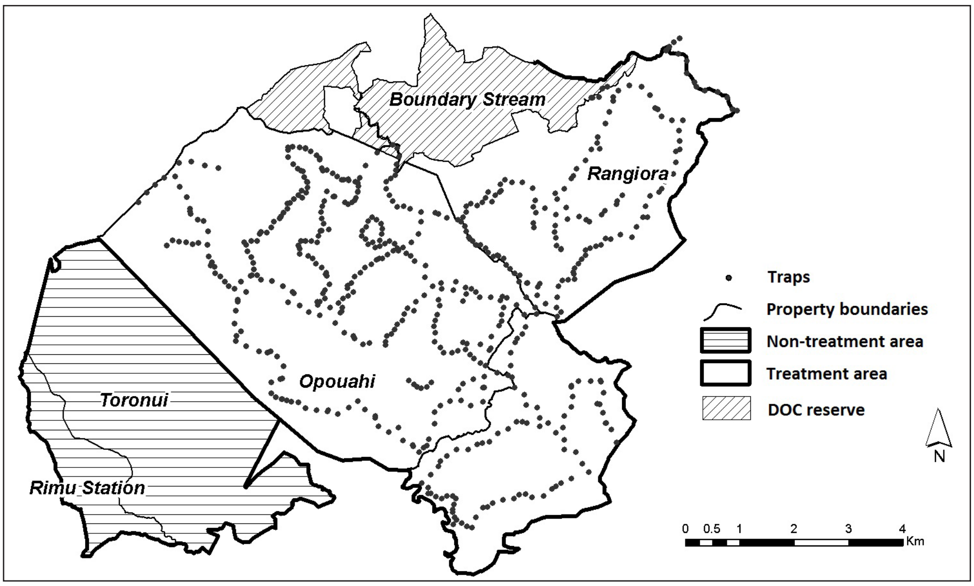

Figure 1. Map of the study area showing the treatment and non-treatment areas relative to Boundary Stream Reserve in Hawke's Bay, North Island, New Zealand. The locations of kill traps are indicated by dots.

We used large tracking tunnels (20 × 20 × 100 cm) with a removable floor, as described by Pickerell et al. (2014). Tracking ink (Black Track, Pest Management Services, Wellington) was applied to the floor in the middle of each tunnel, and sheets of tracking paper (18 × 30 cm) were fastened to the tunnel floor at each end with bulldog clips and drawing pins. Each tunnel was baited with a cube of fresh rabbit meat in the middle of the tracking ink. Tracking papers were retrieved after 3 days and labelled with tunnel number and date; tunnels were left in place year-round. Footprints left on the tracking papers were identified using field guides (Gillies & Williams 2002; Agnew 2009; NPCA 2014).

The first and third point on each monitoring line also had an artificial shelter (wētā house) for monitoring invertebrates. Wētā houses were 7.5 × 62 cm, with six galleries, a clear Perspex cover and a wooden door (Fig. 2). These were attached to tree trunks at approximately chest height and left in place year-round. By opening the wooden door, we were able to count and identify invertebrates through the Perspex cover.

Monitoring lines were checked twice per year (spring and summer) from 2011–2014. In 2015 and 2016 we sampled only once each year (in summer).

Data analysis

We analysed the tracking tunnel data using an occupancy modelling approach (MacKenzie et al. 2006) in which each monitoring line was treated as a site. Although we placed monitoring lines as far apart as practicable, we cannot rule out the possibility that individual predators were detected on more than one line. Occupancy models usually assume spatial independence between sites; when this assumption is relaxed the models estimate the proportion of the area used by the target species during the sampling period, which we refer to as 'site use' (MacKenzie & Royle 2005). Within a monitoring line, each tracking tunnel was treated as a separate survey so that each monitoring line yielded a detection history with five 'occasions' per season. For example, if a species was detected in the first and last tunnel in a line, this yielded a detection history of '10001'. A detection history need not comprise data on detection / non-detection at different times; it can also be made up of data collected at different points within each site (MacKenzie et al. 2006).

We used a multi-season dynamic occupancy model (MacKenzie et al. 2003) to estimate site use for cats, hedgehogs, mustelids, rats, mice and skinks (Scincidae) in each area and sampling season. This model estimates the proportion of sites used by each species, as well as the probability that a species will disappear from a site where it previously occurred ('extinction'), or appear at a site where it had been absent ('colonisation'). Probabilities of colonisation, extinction and initial site use were allowed to vary between treatment and non-treatment. Our model allowed detection probability to vary between tracking tunnels within a monitoring line. However, this model failed to converge for cats and mustelids; therefore, we estimated site use by these predators assuming constant detection probability for all tracking tunnels within a line. Analyses were conducted using the 'unmarked' package (Version 0.11-0) in R (Version 3.4.4; Fiske & Chandler 2011). We tested for differences between treatments by bootstrapping the 95% confidence intervals to estimate a P-value for the null hypothesis that site use was the same between the treatment and non-treatment area.

We tested for differences in the numbers of detections of animals inside bush fragments, at the edge of fragments, or in pasture. Pooling data from all sites and sampling occasions, we used chi-squared contingency tests with the null hypothesis that detections in each habitat would be proportional to the number of tracking tunnels in each habitat (i.e. 40% bush; 20% edge; 40% pasture).

For invertebrates, we calculated the mean number per monitoring line of each taxon counted in the wētā houses in each sampling season. Values for each season were compared between the treatment and non-treatment areas using paired t-tests after using Levene's test to confirm homogeneity of variances. These tests were performed using Microsoft Excel 2010.

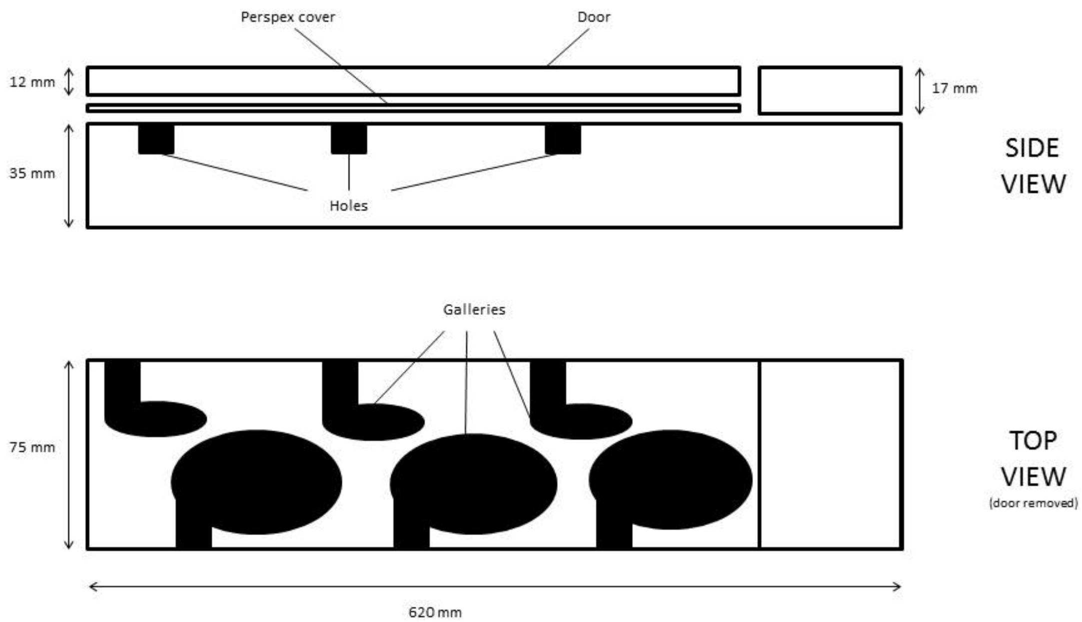

Figure 2. Schematic diagram of a wētā house used to monitor invertebrates. Invertebrates enter through the holes in the side and shelter in the hollow galleries. When the door is open, invertebrates can be identified and counted through the Perspex cover.

Results

The kill traps captured cats, mustelids, hedgehogs, ship rats (*Rattus rattus*) and rabbits. The pulsed cat control removed 134 cats and 21 ferrets (Table 1).

The tracking tunnels detected a range of invasive mammals, including cats (n = 53 detections), mustelids (*Mustela* spp.; n = 15), hedgehogs (n = 218), rats (*Rattus* spp.; n = 148), mice (*Mus musculus*; n = 202) and possums (*Trichosurus vulpecula*; n = 47). Tracking tunnels also detected skinks (n = 54). We were unable to identify individual skink species; however, species likely to be present in the area include the common skink *Oligosoma polychroma*, spotted skink *O. lineoocellatum* and small-scaled skink *O. microlepis* (Bell 2012; Abbott et al. 2013; DOC 2018).

Site use estimates for cats (Fig. 3a) and hedgehogs (Fig. 3b) were similar in both areas during the first sampling season, before predator removal began. After predator removal, site use estimates for these species were consistently lower in the treatment area than in the non-treatment area. Bootstrapping of the 95% confidence intervals for the equilibrium probability of occupancy (i.e. the long-run probabilities based on the function of probabilities of site colonisation and extinction) showed that these differences were statistically significant (cats: treatment (t) = 0.4, non-treatment (n.t.) = 0.99, P = 0.02; hedgehogs: t = 0.49, n.t. = 0.99, P < 0.001). Mustelids were detected in low numbers, leading to a high level of uncertainty in site use estimates (Fig. 3c). Thus, although the mean estimate was lower in the treatment area (0.09) than in the non-treatment area (0.28), the difference was not significant.

Site use by rats was initially higher in the treatment area, and remained so for the duration of the study (t = 0.67, n.t. = 0.39, P = 0.006; Fig. 3d). Mice showed no difference in site use between the treatment and non-treatment area (0.58 for both treatments, P = 0.48; Fig. 3e). Skinks (Fig. 3f) were not detected in either area before predator removal began. However, site use by skinks increased rapidly in the treatment area, while remaining near zero in the non-treatment area (t = 0.43, n.t. = 0.01, P < 0.001).

Cats were detected more often at the edge of bush fragments than expected based on sampling effort (χ^2 = 6.83, d.f. = 2, P = 0.03; Fig. 4a). Mice were detected more often than expected in edge or bush habitats (χ^2 = 21.6, d.f. = 2, P < 0.0001; Fig. 4b), while rats (χ^2 = 98.5, d.f. = 2, P < 0.0001) and possums (χ^2 = 20.87, d.f. = 2, P < 0.0001) were detected more often in bush habitat (Fig. 4c, d). Skinks were detected more often than expected at the edge of fragments or in pasture (χ^2 = 16.4, d.f. = 2, P = 0.0003; Fig. 4e). Hedgehog detections showed no significant difference between habitats (χ^2 = 4.19, d.f. = 2, P = 0.12; Fig. 4f). There was a trend for mustelids to be detected more often in edge or bush habitats than in pasture (Fig. 4g), but low sample size precluded statistical testing.

Taxa observed in wētā houses included tree wētā (*Hemideina* spp.), cave wētā (Rhaphidodophoridae), cockroaches (Blattodea), spiders (Araneae) and slaters (Isopoda). No non-native invertebrates were recorded. During the pre-treatment period, no invertebrates had occupied the wētā houses. During subsequent seasons, counts of cockroaches were higher in the treatment area (P = 0.001). No differences were observed between treatments for any other invertebrate taxon (Table 2).

Discussion

Our results show that extensive, low-cost trapping in a pastoral landscape was associated with lower site use by invasive predators (feral cats and hedgehogs), with apparent benefits for some native fauna (skinks and cockroaches). Although cat control was not continuous year-round, we believe that a large proportion of the cat population was removed. The total of 245 cats captured (Table 1) equates to one cat per km^2 per year. This density is roughly twice the estimated population density of 0.49 cats per km^2 for Hawke's Bay farmland (Langham & Porter 1991).

Table 1. Numbers of animals removed by kill trapping and pulsed cat control on pastoral properties, November 2011 to November 2015.

Species	Number removed Kill trapping	Pulsed cat control
Cat (*Felis catus*)	111	134
Ferret (*Mustela furo*)	51	21
Stoat (*Mustela erminea*)	90	
Weasel (*Mustela nivalis*)	2	
Hedgehog (*Erinaceus europaeus*)	748	
Rabbit (*Oryctolagus cuniculus*)	431	
Ship rat (*Rattus rattus*)	463	

Table 2. Mean numbers of invertebrates recorded per monitoring line in wētā houses in the treatment and non-treatment area during eight sampling sessions, February 2012 to November 2015. All *t*-values and P-values are for two-tailed, paired t-tests with 7 degrees of freedom. Each monitoring line had two wētā houses; one 200 m inside a bush fragment and one near the bush-pasture margin.

Taxon	Mean count (±SD) per monitoring line Treatment	Non-treatment	*t*	P
Cockroaches (Blattodea)	1.3 ± 0.7	0.3 ± 0.3	-3.84	0.001
Spiders (Araneae)	1.5 ± 0.4	1.8 ± 0.6	1.32	0.21
Cave wētā (Rhaphidodophoridae)	1.5 ± 0.7	1.0 ± 0.6	-1.6	0.14
Tree wētā (*Hemideina* spp.)	1.5 ± 0.5	1.9 ± 0.9	1.1	0.3
Slaters (Isopoda)	0.1 ± 0.2	0 ± 0	n/a*	n/a*

*Slaters were recorded in the treatment area in only one sampling season; therefore, numbers were too low for statistical analysis.

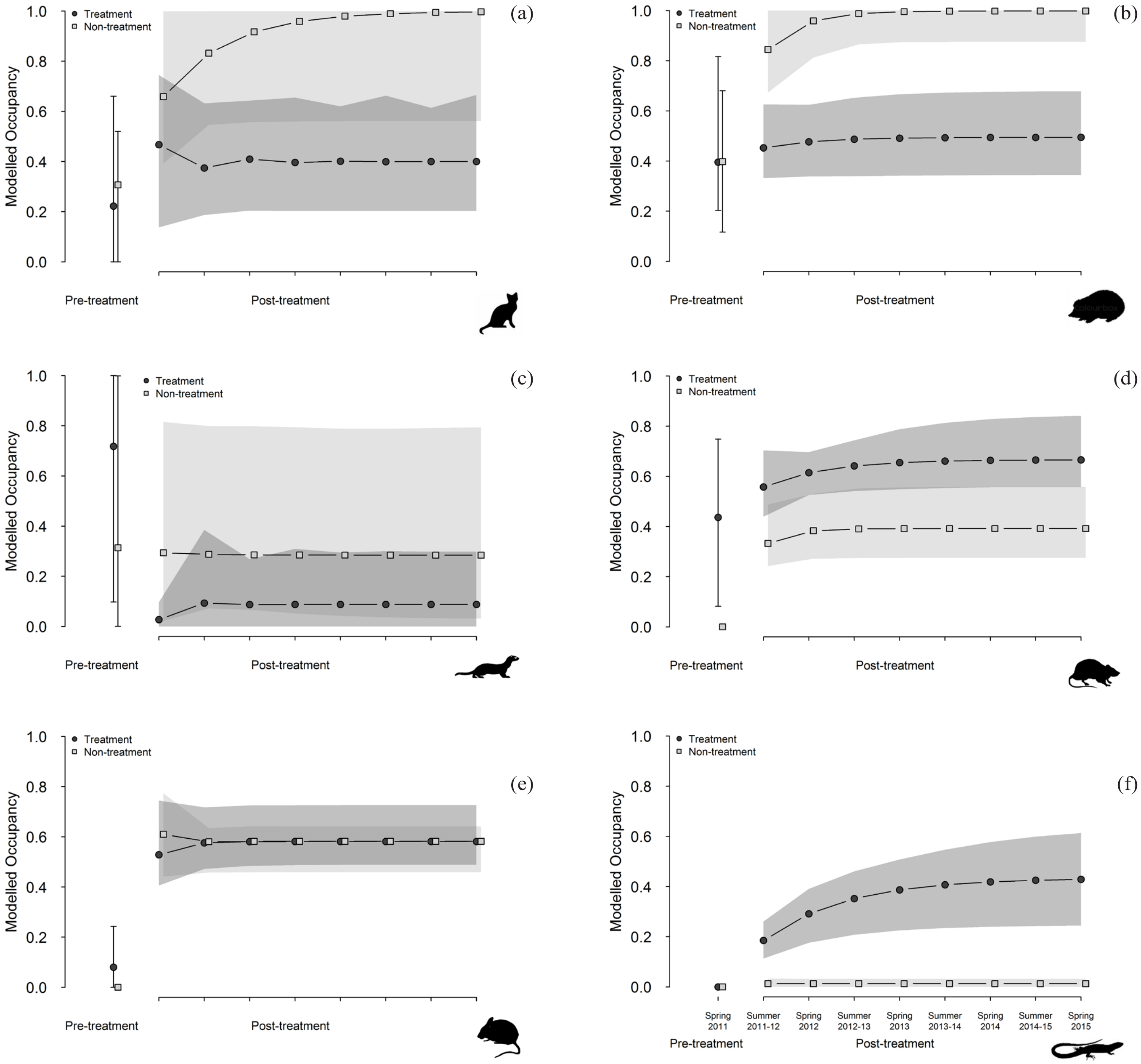

Figure 3. Site use during each sampling season (with 95% confidence intervals indicated by grey shading) of (a) cats, (b) hedgehogs, (c) mustelids, (d) rats, (e) mice and (f) skinks in the treatment (predator removal) and non-treatment areas. Predator removal began in the treatment area after the first sampling season.

Developing low-cost methods to remove predators from large areas is an essential step towards the vision of a Predator-Free New Zealand (Russell et al. 2015). Our predator control did not reduce predators to zero-density, but did achieve measurable reductions in predator populations at a relatively low cost compared to intensive predator control. This cost-effectiveness was due to the innovative approach used. The spatial coverage of our trapping effort was made possible by placing traps in accessible locations where they could be checked rapidly by staff on an ATV. This design maximised the number of traps that could be checked in a day, thereby increasing the area that could be trapped with the available budget. Our network of kill traps also used mechanical signals that allowed the trapper to see whether a trap had been triggered without dismounting the ATV, saving time and reducing labour costs. Recent developments in wireless sensor networks (Jones et al. 2015) may further reduce costs of trapping by alerting managers when a trap is triggered.

Our study is among the first to use large tracking tunnels for detecting invasive predators such as cats, mustelids and hedgehogs (see also Pickerell et al. 2014). However, tracking tunnels detected few animals during the first sampling season. This effect may have been due to neophobia as the tunnels had been in place for only a few days. Detection rates were much higher after 3 months, suggesting that this was sufficient time for animals to become habituated to the tracking tunnels. It is likely that predator site use was underestimated in the first sampling session because of neophobia; the apparent increase in predator site use in the non-treatment area may be an artefact of this initial underestimation. We believe predator

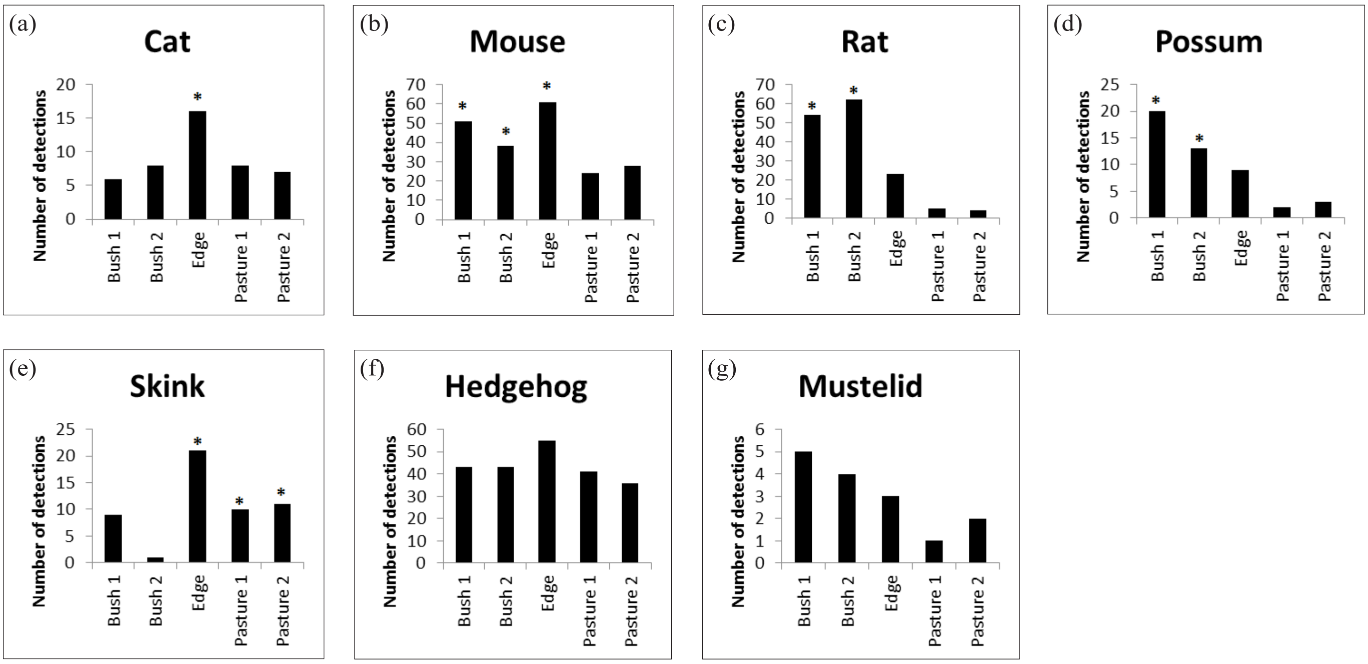

Figure 4. Numbers of detections of (a) cats, (b) mice, (c) rats, (d) possums, (e) skinks, (f) hedgehogs and (g) mustelids by large tracking tunnels in bush, edge and pasture habitats from October 2011 to February 2016. The first tracking tunnel in each monitoring line was inside a bush fragment, 200 m from the edge (bush 1). Tracking tunnels were 100 m apart, with the third tunnel being at the bush-pasture margin (edge), and the last being in pasture, 200 m beyond the edge of the fragment (pasture 2). Asterisks indicate habitats where species were detected significantly more frequently than expected by chance.

site use at both sites during the pre-treatment period was likely much higher than our estimates suggest, and probably declined in the treatment area while remaining relatively stable in the non-treatment area. Future trials should compare the efficacy of large tracking tunnels with other tools for detecting predators, e.g. camera traps and wildlife detector dogs (Glen et al. 2014, 2016; Glen & Veltman 2018). Studies using large tracking tunnels should include a longer period of repeated sampling in the pre-treatment period to test for the possible effect of neophobia and generate more reliable estimates of pre-treatment site use or abundance. A longer deployment period has been shown to increase the probability of detecting predators in tracking tunnels, although this may require the bait to be replaced periodically (Pickerell et al. 2014).

Ideally our study would have included spatial replication (Underwood 1994); however, this is often unaffordable for large-scale adaptive management programmes such as ours. One solution would be to apply a treatment reversal (e.g. Innes et al. 1999) in which the treatment and non-treatment areas are switched. However, stopping predator control in our current treatment area would be contrary to the aims of this conservation intervention. Another alternative may be to apply a 'treatment extension' in which predator removal is applied to both areas. If similar results and outcomes were observed in the former non-treatment area, this would increase confidence that the observed changes were due to predator removal.

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