Landholder participation in regional-scale control of invasive predators: an adaptable landscape model

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ORIGINAL PAPER

Landholder participation in regional-scale control of invasive predators: an adaptable landscape model

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Abstract Control of invasive predators is necessary for the conservation of many endemic species. Invasive predator management tends to focus on priority sites, which often comprise only a small fraction of the impacted area. Landscape-scale ecological recovery requires threatening processes to be managed not only in these priority areas, but also in the matrix between them. However, wide-scale control of invasive species can be logistically, economically and socially challenging. We developed a spatially explicit model to estimate the effects of varying levels of landholder participation in landscape-scale programs to control invasive predators. We demonstrate the use of this model with a case study from the North Island of New Zealand in which the results of predator control are projected over a 6 year period. Under various scenarios for landholder participation, we estimated how the participation rate, and size and location of non-participating properties, would influence effectiveness of predator trapping. We also modelled how trap deployment could be adjusted to limit reinvasion from non-participating properties. Under all modelled scenarios, predator populations remained below pre-control levels throughout the 6 years. Non-participation by owners of small properties (≤25 ha) had a negligible effect on the efficacy of predator control. If owners of large properties (>800 ha) failed to participate, reinvasion by predators from these properties reduced the efficacy of control; however, this could be largely offset by placing additional traps on the nearest participating properties. Predator control will thus be effective even if some landholders choose not to participate. Our model can be readily adapted to other invasive species and landscapes worldwide.

Keywords Agro-ecosystem · Community support · Feral cat · Ferret · Social-ecological models · Stoat

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Introduction

Control of invasive predators is one of the most important tools for conserving native fauna in island ecosystems, including Australia and New Zealand

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where endemic species are highly vulnerable to decline and extinction through predation (Dickman 1996; Salo et al. 2007; Innes et al. 2010; Simberloff 2010). Control of invasive predators is often restricted to priority sites such as wildlife sanctuaries, designated conservation areas or relict patches of habitat (e.g. Kinnear et al. 1988; Norbury et al. 2013), which may represent only a small proportion of the impacted landscape. If impacted faunal assemblages and ecological processes are to be restored at a landscape scale, invasive predators must also be controlled in the matrix between these areas (Glen et al. 2013). However, controlling invasive predators across entire landscapes is challenging due not only to the resources required, but also variation in land tenure and the need for a cohesive approach among many landholders who may have diverse views on conservation and invasive species management.

Increasingly, the role of the general public in invasive species management has been recognised worldwide, particularly in areas with a high diversity of land uses (Epanchin-Niell et al. 2009). Landholders' decisions to engage in control efforts may be influenced by the perceived value of reducing invasive species populations, the perceived difficulty and cost of engaging in control actions, their personal ability to take effective action, and the perceived likelihood that widespread control efforts will be successful (Aslan et al. 2009; Corbett 2002; McLeod et al. 2015; Prinbeck et al. 2011). Landowners may be influenced especially by their neighbours' actions, as normative influences are important predictors of pro-environmental behavior (Corbett 2002). The actions of neighbours may influence behaviour by indicating the potential for social rewards for engaging, or penalties for not engaging, in invasive predator control (Cialdini et al. 1990; Jackson 2005).

New approaches are needed that link environmental decision making, societal participation in environmental management, and biophysical requirements for intervention. Recent developments in social-ecological modelling (e.g. Rebaudo and Dangles 2013) provide an approach for improving environmental management, both in case studies with specific characteristics, and for establishing generic principles and 'rules of thumb' that can be applied to guide policy at regional and national scales.

Successful environmental management depends on societal cooperation and commitment as well as ecological responses to intervention (Lade et al. 2013). For instance, pest management needs to account for the dynamics (population changes and spread) of invasive species, the cost of control per unit area, the number of participating landholders, the size and spatial context of their properties, and the efficacy of control. When there are few participants rapid reinvasion of pests from surrounding unmanaged properties may reduce the efficacy of control at a landscape scale (Gentle et al. 2007; Cumming et al. 2013). Control efficacy is predicted to increase as more landholders participate until it reaches a maximum, which is set by the tools and techniques used. However, this scenario has not been tested at a regional scale in New Zealand production ecosystems or elsewhere.

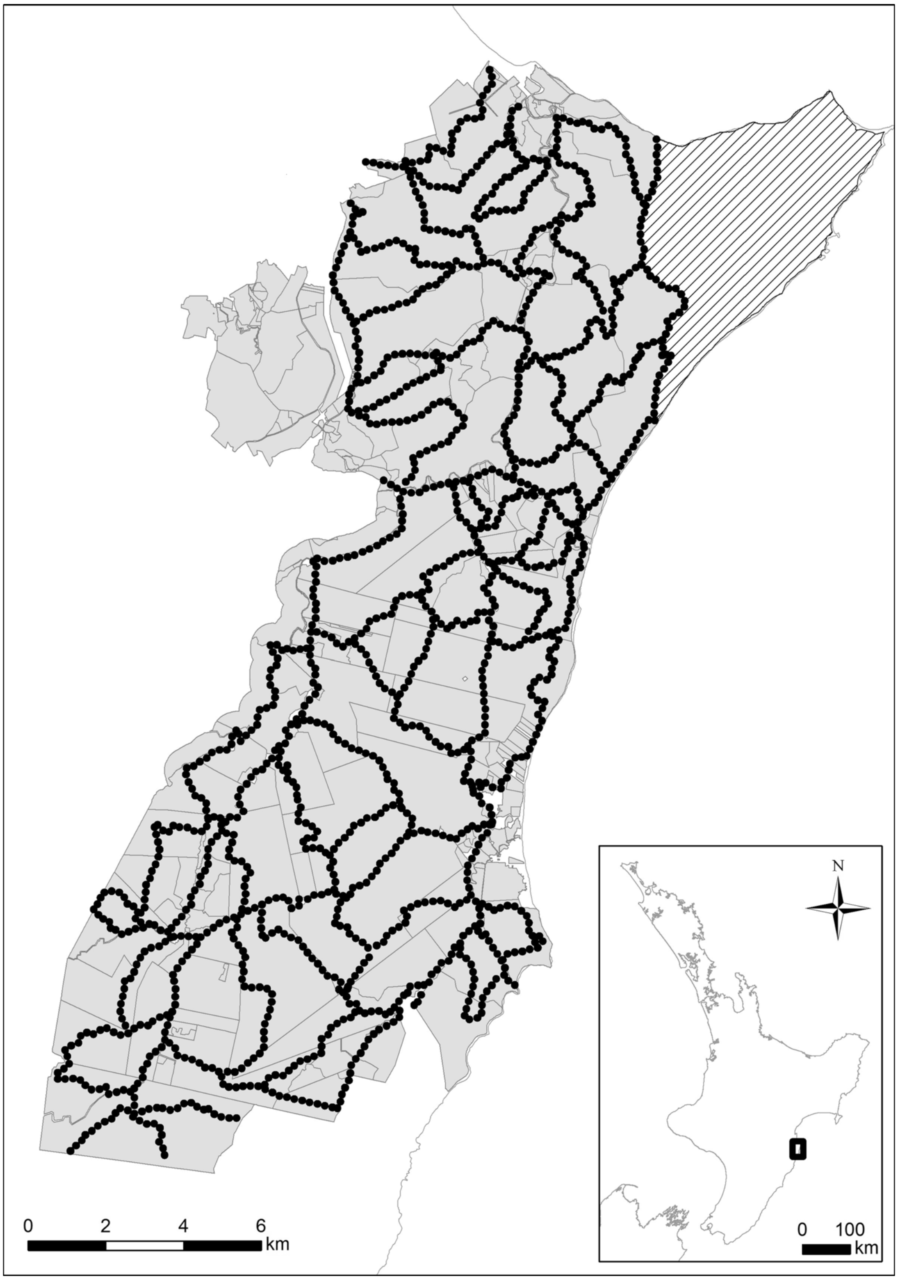
In the North Island of New Zealand, Hawke's Bay Regional Council plans to control invasive predators (feral cats *Felis catus*, stoats *Mustela erminea* and ferrets *M. furo*) across 26,000 ha of rural and peri-urban land under the proposed *Cape to City* program (http://capetocity.co.nz/). Most of the land in the *Cape to City* area is privately owned, and participation of landholders will be voluntary. There is a risk that non-participation by some landholders might leave uncontrolled pockets of predators, which might reinvade surrounding areas and undermine the effectiveness of the program. Here we develop and apply a spatially explicit simulation model to predict how different levels of landholder participation influence the effectiveness of predator control across the whole *Cape to City* area. We aim to estimate (1) the impact of varying levels of landholder participation; (2) how size and location of non-participating properties influence the effectiveness of predator control, and; (3) how trap deployment might be adjusted to limit reinvasion from non-participating properties. Our model is broadly applicable to scenarios in which invasive species are managed across multi-tenured landscapes.

Methods

Study area

Cape to City covers 26,000 ha of rural and residential land in Hawke's Bay, North Island New Zealand (39°47′S; 176°57′E; Fig. 1). The north-eastern boundary is adjacent to Cape Sanctuary, a privately owned

Fig. 1 Map of the *Cape to City* area (*shaded*) and the Cape Sanctuary reserve (*hatched*) in Hawke's Bay, New Zealand showing property boundaries (*dashed lines*) and the planned locations of traps (*dotted lines*) for on-going predator control



wildlife reserve protected by a predator-proof fence. Invasive predators are controlled to low densities in Cape Sanctuary, which has allowed population recovery and/or reintroduction of numerous threatened native species (Ward-Smith 2011; Innes et al. 2015). Ten kilometres west of Cape Sanctuary is the city of Havelock North (hence '*Cape to City*'). The aim of *Cape to City* is to allow native biodiversity, including rare and threatened species, to spread from Cape Sanctuary and other relict populations and coexist with people in residential, rural and recreational areas.

The *Cape to City* area comprises 163 properties ranging in size from 1.5 to 2033 ha (mean = 189 ha). These include sheep and cattle farms, orchards, vineyards and residential 'lifestyle' properties. There are also fragments of native vegetation, exotic timber plantations and some small conservation reserves that collectively cover ~8 % of the *Cape to City* area.

Invasive predator control

Control of invasive predators is planned to begin in 2016, and will be maintained for at least 5 years. Predators will be controlled initially by intensive use of a combination of trap types. After this initial knock-down phase, on-going control will involve a network of 1460 kill-traps (modified DOC 250 traps, Department of Conservation, Wellington, NZ) set ~200 m apart alongside roads and farm tracks on predefined routes throughout the *Cape to City* area except for a small portion near the western edge (Fig. 1). Easy access is required to make trapping affordable over this extensive area. Traps will have lures attractive to all three predator species and will be left in place year-round; however, they will be set for active trapping sessions over seven consecutive nights every 1.5 months.

Model design

A spatial model written in R version 3.1.3 (R Development Core Team 2015) was used to estimate predator population size over time under each of four landholder participation scenarios (see below). Each invasive predator species (feral cat, ferret, stoat) was modelled separately. The R code is provided in Appendix S1.

The first step was to specify which properties participated in the trapping program, which constrained the trap layout to include only traps that fell within those properties. The model was then initiated by creating a regular grid of available home range centres across *Cape to City*. The resolution of the grid of home range centres (500 m) was set so that each grid cell could be occupied by a single animal and the local density of predators did not exceed the maximum allowed (4 per km^2). Post-knock-down populations of predators (details below) were then randomly distributed across this grid. An occupied home-range-centre grid point remained occupied for the entire simulation unless the predator was captured by a trap, in which case it was removed from the population and that location became available to in situ dispersers or invaders. The carrying capacity (K) of the predator population over the whole of the *Cape to City* landscape was based on typical population densities for feral cats, ferrets and stoats in New Zealand (Clapperton and Byrom 2005; Gillies and Fitzgerald 2005; King and Murphy 2005). Carrying capacity was used to set a limit to population growth and immigration, whereas maximum local density was used to determine the spatial location of new recruits. Initial populations (pre-knock-down) were based on population densities estimated specifically for the study area (C. Leckie, unpubl. data), and the percent kill achieved during the knock-down phase was determined from data obtained from a pilot study at nearby Waitere Station (A. Glen, unpubl. data). Three levels of knock-down were simulated for each of the three predator species: low (cats: 54 % kill, ferrets/stoats: 61 %), medium (cats: 90 %, ferrets/stoats: 86 %) and high (cats: 98 %, ferrets/stoats: 95 %).

We quantified the probability that each individual predator would be removed by the network of traps. The probability of capture of an individual ($P(capture)_{ijt}$) with a home-range centre at location i by trap j during night t was:

$$P(capture)_{ijt} = g_0 \exp\left(\frac{-d_{ij}^2}{2\sigma^2}\right)$$

where d_{ij} was the distance between home-range centre i and trap j, g_0 was the probability of capture of an individual by a trap placed at the animal's home-range centre and σ was the spatial decay parameter for a half-normal home-range kernel (Efford 2004). Values for g_0 and σ were randomly drawn from Program Evaluation and Review Technique (PERT) distributions (Herrerias et al. 2003) with parameters described in Table 1. These parameters were determined through a review of the literature on home ranges, movements and capture probabilities for each species in New Zealand (Glen and Byrom 2014). Each animal retained the same g_0 and σ values across all trapping sessions, i.e., we assumed these are traits that characterise the behaviour of an animal from birth to death. Further, by drawing the g_0 and σ parameters from a distribution with sufficient variance, we ensured that selected values provide a representative sample of variation across individuals, sexes, and population densities. The probability that each individual would be captured by any one of the j traps in the *Cape to City* area over the seven nights of a trapping session was calculated as:

$$P(capture)_i = 1 - \prod_{j=1}^{j} \left(1 - P(capture)_{ijt}\right)^7$$

Each individual was then either captured (and thus removed from the population) or left untrapped based

Table 1 Parameters used in a simulation model of the effects of landholder participation on predator control within the *Cape to City* program, Hawkes Bay, North Island, New Zealand. All symbols are defined in the text and distances are in metres

	Feral cats			Ferrets			Stoats		
	Min	Likely	Max	Min	Likely	Max	Min	Likely	Max
Initial population size		500			200			200	
Carrying capacity (K)		900			300			250	
g_0	0.01	0.04	0.08	0.05	0.079	0.1	0.017	0.04	0.77
Sigma (σ)	259	351	436	430	466	500	492	600	891
Lambda (λ)		31			18			31	
Productivity (F)	0.98	1	1.09	1	1.1	1.65	0.1	0.23	2.3
Dispersal (m)		30,000			22,000			65,000	
Dispersal (μ, ε)		2000, 1.5			1500, 2.2			2000, 2.5	

on a random draw (1 or 0) from a binomial distribution with $p_i = P(capture)_i$. In the model we did not close the specific trap that caught an individual, assuming that trap saturation was not a problem (due to a high ratio of traps to predators in the *Cape to City* area and the protocol for re-setting all traps at least once every 6 or 7 weeks), and we did not adjust trap availability in the calculation of $P(capture)_i$ for subsequent individuals.

The simulation continued the trapping described above for the remaining population with eight trapping sessions per year over 6 years. Before the start of each trapping session (except for the first session), immigration from outside the western boundary of *Cape to City* was allowed to occur (the eastern boundary is ocean). We assumed this immigration was composed of mostly adults and occurred year-round, as opposed to juvenile dispersal which occurred only after the breeding season (see below). The number of invaders was determined as a random draw from a Poisson distribution with parameter lambda (λ; Table 1). These values were based on estimated rates of reinvasion from other predator control programs in New Zealand and Australia (Murphy and Dowding 1994; Alterio 1996; Short and Turner 2005; Anderson unpublished data), adjusted for the length of the invasion front in the *Cape to City* area (41.5 km). Following the addition of in situ recruits (details below), the randomly drawn number of invaders was adjusted so that residents plus invaders did not exceed K, i.e. immigration was assumed to be density-dependent. Adjusted invaders were then randomly placed on home range centres unoccupied by residents, i.e. all available home range centres were equally likely to receive an invader irrespective of their distance from the western invasion front. This is a realistic assumption for the *Cape to City* area given that it is only ~10 km wide, a distance well within the range of dispersal distances for all three predator species. Settled invaders were then added to the resident population and trapped following the steps described in the previous paragraph.

Each year before the start of the seventh trapping session, the modelled population was allowed to reproduce. The productivity per adult (F), i.e. the number of juveniles (after juvenile mortality) recruited to the breeding population per resident adult, was randomly drawn from a PERT distribution with parameters (Table 1) that were based on species body weight (a strong predictor of intrinsic rate of increase; Sinclair 1996) as well as information from previous population studies of these species (Derenne 1976; King 1983; van Aarde 1984; Thompson 1987; Korpimäki et al. 1991; Barlow and Norbury 2001; Barlow and Barron 2005; Short and Turner 2005). The drawn number of recruits per adult was adjusted so that the total number of adults plus recruits did not exceed K. This was done by first removing one recruit from adults having ≥ 2 recruits, or in the case no adult had multiple recruits, by randomly removing recruits from any of the adults. The adjusted number of juveniles then dispersed from the maternal home range centre to an unoccupied home range centre which was bounded by the maximum dispersal distance for the species (m; Table 1). From the pool of H remaining unoccupied home range centres, the destination of a juvenile was determined by a random draw from a multinomial distribution. The multinomial probability for a dispersing juvenile from maternal location i to available location h was calculated as:

$$P(disp)_{ih} = \frac{\Phi(d_{ih}|ln(\mu), ln(\varepsilon))}{\sum_{h=1}^{H} \Phi(d_{ih}|ln(\mu), ln(\varepsilon))}$$

where Φ was a log-normal probability density function, d_{ih} was the distance between the maternal (i) and available (h) location, and μ and ε were the mean and standard deviations of dispersal distances (Table 1). The juvenile was then placed in the selected home range centre, which became unavailable for the remaining juveniles. After dispersing the in situ recruits and adding them to the resident population, invaders were allowed to enter the study area following the steps described in the previous paragraph. In the model animals were only vulnerable to trapping once they had settled into a home range centre, and not during the dispersal or invasion stages. We also assumed that natural mortality was relatively low due to minimal competition at low population density (Byrom 2002). In this sense, we assumed that the main process influencing population dynamics was trapping; in comparison, any intra- or inter-specific dynamics were assumed to be small, and they were modelled implicitly through the stochastic processes of population growth and dispersal.

Population size was recorded at the end of each trapping session. The simulation was repeated 10,000 times, with new regular grids of home range centres and parameter values drawn for each animal at each iteration. The uncertainty in predictions of median population size was assessed by looking at the 2.5 and 97.5 percentiles of the distribution.

Simulated scenarios

1. *Status quo* Although most landholders in *Cape to City* have agreed to participate in the predator control program, there are two properties of ~900 ha adjacent to Cape Sanctuary whose participation has not yet been confirmed. We ran the model removing traps only from these two large properties. For comparison, we also ran the model with 100 % participation by landholders.
2. *Limited participation by lifestyle landholders* If landholders are influenced by neighbours' actions, then small clusters of adjacent properties may decline to participate, rather than scattered individual ones. In *Cape to City*, these clusters are expected to occur mostly in areas dominated by lifestyle properties (i.e. properties of ≤25 ha; Sanson et al. 2004). We identified five clusters of lifestyle properties, each composed of an average of 12 properties (range 5–19) and with an average cluster size of 110 ha (range 33–208). The model was run for three levels of lifestyle landholder participation by randomly excluding 1, 2, or 3 of the clusters.
3. *Failure of landholders with large properties to participate* There are seven landholders with large properties (>800 ha). If they choose not to participate, this might reduce trapping effectiveness more than if landholders with small properties opt out. The model was run for four levels of large landholder participation by randomly excluding 1, 2, 3, or 4 of these large properties.
4. *Traps allocated to the properties of non-participants relocated to neighbouring properties* If a large landholder in the middle of the study area fails to participate in the trapping program, this could provide a refuge from which surviving predators reinvade adjacent areas. This could be mitigated with a buffer of traps around the non-participating property. To simulate this, we removed traps from randomly selected large properties, and placed an equivalent number of traps on adjacent participating properties. These were intended to intercept predators whose home ranges extend beyond non-participating properties, and juvenile predators dispersing from such properties. The additional traps were accommodated by placing traps more closely along the routes already identified. The model was run for four levels of large landholder participation by randomly excluding 1, 2, 3, or 4 of these large properties (as in scenario 3).

Results

Abundance of all three predator species was predicted to decline rapidly after the initial knock-down (Table 1; Appendix S2). However, the predicted rate of population recovery varied between scenarios, and according to participation rate. Results for the medium initial knockdown are shown in Fig. 2. Varying the

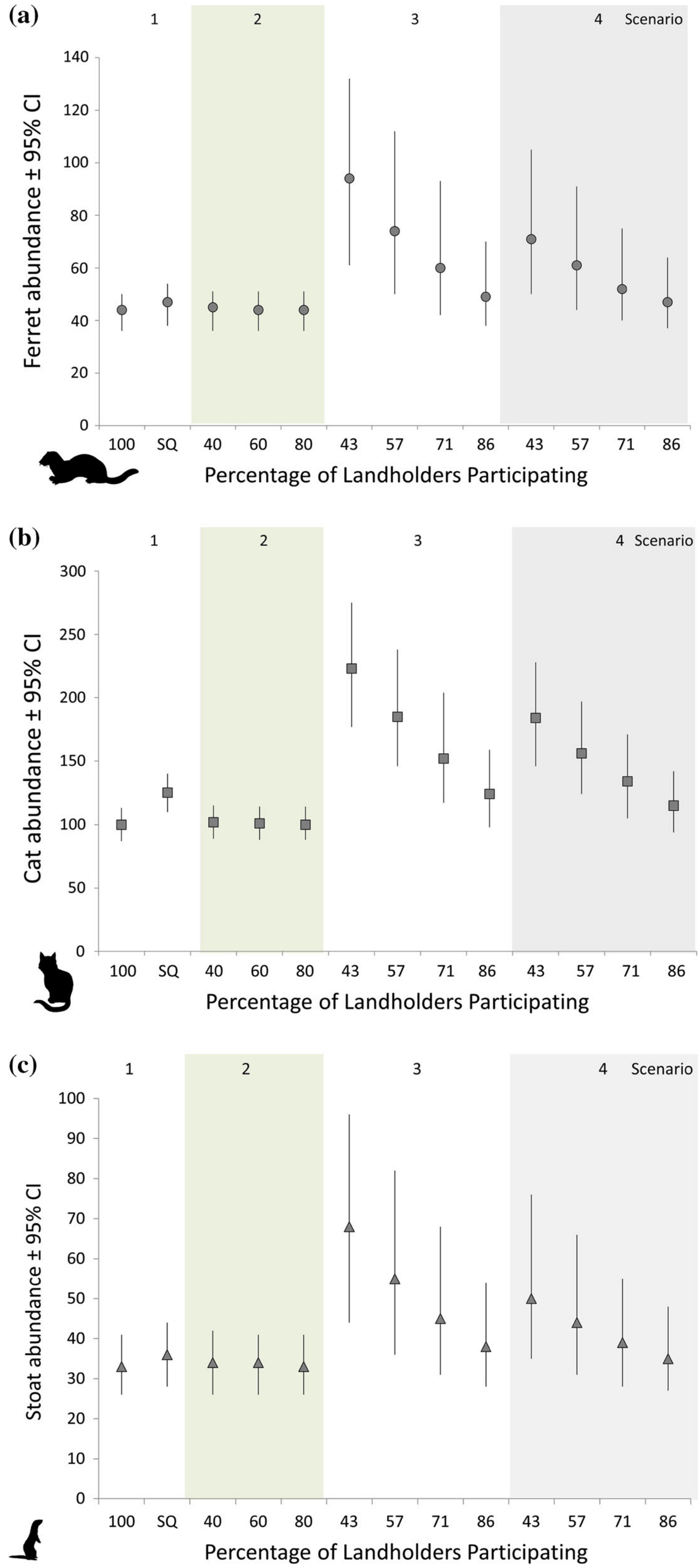


Fig. 2 Predicted median numbers (with 2.5 and 97.5 percentiles) of **a** ferrets (*Mustela furo*), **b** feral cats (*Felis catus*) and **c** stoats (*Mustela erminea*) remaining after 6 years of simulated predator control under each of four scenarios with mean initial knockdown levels (Table 1): (1) Status quo (SQ): participation by all but two properties adjacent to Cape Sanctuary; (2) non-participation by clusters of landholders with small 'lifestyle' properties; (3) non-participation by landholders with large properties; (4) relocation of traps from non-participating large properties to neighbouring properties. For each scenario, a range of levels of landholder participation is presented. For comparison, the left-most point shows predicted predator numbers when there is 100 % landholder participation

level of initial knock-down had minimal effect on predicted predator abundance after 6 years (Appendix S3).

Under Scenario 1 (*Status quo*), 1332 traps (91 % of the planned total) fell within participating properties. Predator populations recovered slowly after initial knock-down; after 6 years, abundance of all three predator species was much lower than before the initial knock-down (Fig. 2).

The *degree of participation of lifestyle landholders* (Scenario 2) had no noticeable effect on predator population trends (Fig. 2), and the uncertainty around the estimates of population size after 6 years was small. This suggests that, regardless of which clusters of lifestyle properties participated, the reduction in predators achieved is very similar. Further, only 3 % of the *Cape to City* area is occupied by lifestyle properties. Thus, even if all lifestyle landholders decided not to participate, only 34 traps (2 %) would be lost from the trapping network.

Under Scenario 3 (*Failure of landholders with large properties to participate*), the percentage of the study area participating in trapping ranged from 77 to 91 %. For all three predator species, participation rate had a substantial effect on the predicted abundance (Fig. 2). However, even at the lowest participation rate the model predicted predators would remain well below their initial abundance after 6 years.

Relocating traps from non-participating properties to neighbouring ones (Scenario 4) increased the effectiveness of predator control. Predator populations were lower for this scenario than for Scenario 3, in which the total number of traps was reduced by non-participation (Fig. 2). Further, although effectiveness of predator control was higher under scenario 4 than under scenario 3, the difference at the higher level of participation (86 %) was very small, suggesting that there is little gain from relocating traps when there is a high level of buy-in from landholders of large properties.

Discussion

To our knowledge, this is the first spatially explicit population model developed to predict the effectiveness of invasive predator control across a large, multi-tenure landscape. Given appropriate data on species biology, property boundaries, and willingness of landholders to participate, our model can easily be adapted to simulate the results of managing other invasive species, or to other landscapes, worldwide.

In our case study, we have shown that the control of invasive predators proposed in the *Cape to City* conservation initiative is likely to reduce the abundance of feral cats, stoats and ferrets under a range of plausible scenarios for landowner participation. Under the most likely *status quo* scenario, numbers of all three predators are predicted to recover gradually after the initial knock-down, but remain substantially below starting levels for at least 6 years.

Predator control would be less effective if one or more large properties opt out of the trapping program. However, the effects of non-participation by some of these landholders would be largely negated if the traps intended for their properties were shifted to neighbouring properties, as in Scenario 4. Many of the proposed trap lines run along property boundaries, and landholders on both sides of the boundary would presumably have to opt out in order to influence the proposed layout of traps. In modelling Scenarios 1–3, if a property was excised from the program all traps within it were removed but in reality, these traps may simply be able to be set a few metres away across the property boundary. Thus, the model's predictions are likely to be pessimistic when predicting the influence of individual properties opting out of the trapping effort. Our model also predicts that non-participation by owners of small properties would have a negligible effect. These results give a high level of confidence that predator control in the *Cape to City* area will be effective even if some landholders choose not to participate.

Based on the 6-year end points of simulated population trajectories, Scenarios 1 (*Status quo*) and 2 (*Limited participation by lifestyle landholders*) outperformed all other scenarios (Fig. 2). Comparison of trajectory endpoints for Scenarios 3 (*Failure of landholders with large properties to participate*) and 4 (*Traps allocated to the properties of non-participants relocated to neighbouring properties*) indicated that when large properties were not included, redeployment of traps onto neighbouring properties would compensate, resulting in better suppression of all predator populations.

Results from our spatial model suggest that support for the predator control program by owners of large properties will be important. Also, our modelling demonstrates that, with the exception of very small properties, continued high levels of landholder participation will be needed to capitalise on gains achieved through the initial broad-scale knock-down of predators. Further social data are needed to determine landholders' attitudes towards pest mammals and to pest control, and the factors that might influence the likelihood that a landholder will support and/or participate in broad-scale predator control. Behavioral theories from social psychology research can help inform what these factors may be (MacLeod et al. 2015).

Participants in environmental decision making and management can have differing expectations depending on cultural backgrounds (e.g. indigenous cf. non-indigenous peoples), occupation (e.g. farmers, horticulturalists) and location (urban, peri-urban or rural) (Karali et al. 2014). Landholder participation in pest management can depend on the cost, the type of management (e.g. poison vs. traps), each person's or group's interest in the desired outcome, and the number and characteristics of participating landholders (Bandura 1998; Fowler and Christakis 2010; Montanari and Saberi 2010). For example, if individual landholders perceive that not enough large landholders are participating, they may believe that their own control efforts will be futile.

Further, the number and characteristics of potential participants may determine the period of greatest need for investment in public engagement; for example, investing early to draw in a number of key social 'hubs' may allow for a baseline number of individuals to be recruited initially to establish social norms and enhance efficacy beliefs, which may then attract others to the program (Fowler and Christakis 2010). Conversely, landholders might drop out of a pest control program due to 'burn out' (which may be reduced, for example, by employing contractors to do the control), a change in land use (e.g. a change to production unaffected by pest animals) or if control is very efficient and pests are no longer perceived to be a problem (Russell et al. 2015).

Finally, models can be designed to address more general questions such as: How do new perceptions of pest control, or of the value of native biodiversity, spread through and persist in communities? (see, for example, Kendal and Laland 2000; Fowler and Christakis 2010; Montanari and Saberi 2010). This might generate insights into how communities both in New Zealand and internationally might learn from the *Cape to City* experience.

Although our model predicts reduced predator abundances as a result of trapping in *Cape to City*, it remains to be seen whether these reductions will be sufficient to allow recovery of native species and ecological processes. Some species and ecological processes may respond in a linear fashion to predator control, whereas others may not respond unless predator abundance is reduced below some critical threshold (Norbury et al. 2015). It is essential that long-term monitoring measures both the *results* of predator removal (changes in predator numbers) and the *outcomes* (e.g. changes in abundance or distribution of native prey species) (Clayton and Cowan 2010). Such information might, in turn, influence landholders' attitudes about participation in *Cape to City*. As more information becomes available (e.g. confirmation of participating properties and information on the drivers and motivations for participation by landowners), our model could be adapted to predict more accurately the likelihood of success in terms of the desired outcomes for native species.

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