NEW ZEALAND JOURNAL OF ECOLOGY

RESEARCH

Effects of untrapped land on the control of predators and associated monetary costs

Mark Hostetler[1]* and Dean P. Anderson[2]

[1]Department of Wildlife Ecology & Conservation, IFAS, University of Florida, PO Box 110430, Gainesville, FL 32611-0430, U.S.A.
[2]Manaaki Whenua Landcare Research, P.O. Box 69040, Lincoln 7640, New Zealand
*Author for correspondence (Email: hostetm@ufl.edu)

Published online: 30 June 2025

Abstract: In an area targeted for predator control, untrapped land can act as a source of new predator recruits into the trapped area, reducing biodiversity outcomes. We investigated how increasing the size of untrapped land could drive up costs and also compromise the ability to control predators. Using an individual-based simulation model, we explored how increasing the size of untrapped land impacted rat, possum, and stoat densities and how monetary costs rise because of the need for extra trapping effort near the perimeter of the untrapped property. This was done for low, medium, and high carrying-capacity habitats, such as drylands, beech, and podocarp forests, respectively. Sizes of overall areas in the simulation were 576 ha (rats), 2304 ha (possums), and 32 400 ha (stoats). Using a 5% tracking/trapping rate as a density impact threshold at which we considered biodiversity outcomes would be significantly affected, we calculated the corresponding densities as 20 rats km^{-2}, 13 possums km^{-2} and 0.4 stoats km^{-2}, respectively. For medium carrying-capacity landscapes, threshold densities over the full extent (trapped and untrapped areas) were exceeded when the untrapped area was 36.9 ha for rats, 72.6 ha for possums, and 459 ha for stoats. The trapping costs at the density impact threshold were about 4–5% higher than at the baseline if all property owners participated. For low carrying-capacity landscapes, biodiversity thresholds were typically not exceeded even at the largest untrapped area modelled. For high carrying-capacity landscapes, modelling results indicated that untrapped areas had to be smaller to meet thresholds. Conservation organisations can use the results to evaluate whether and how to implement trapping programs in areas where there is not 100% cooperation among landowners.

Keywords: conservation, cost-effectiveness, eradication, invasive species, possums, predator control, rats, stoats

Introduction

Rats (*Rattus* spp.), stoats (*Mustela erminea*), and possums (*Trichosurus vulpecula*) are significant predators of New Zealand wildlife, and conservation programs and initiatives have sought to implement techniques of eradication and/or control (Parkes & Murphy 2003; Parkes et al. 2017; Leathwick & Byrom 2023). Given the goals of the Predator Free 2050 initiatives, many areas are being targeted for eradication and/or control of introduced possums, rats, and stoats (King 2023). Often when an organisation has funds to implement a predator control or eradication program, they typically spend most of the funding on buying traps and hiring people to administer the traps and/or monitor biodiversity outcomes (Parkes & Murphy 2003; Peters et al. 2016). However, in a targeted region, several landowners may not want trapping on their properties and conservation organisations may choose to just trap around these properties. These untrapped areas consequently become refugia for predators, and reinvasion is more likely into the nearby control/eradicated areas (Parkes et al. 2017). To combat reinvasion, increased trapping/monitoring efforts in buffer areas are often conducted along the borders of untrapped areas (Department of Conservation 2023). This can drive up costs, but by how much? What effects do untrapped areas have on predator control across the entire region? How do these untrapped areas affect costs and impact the efficacy of trapping in areas of interest?

Given a scenario where a few landowners will not allow predator control on their property, an organisation has several choices. They can decide to: (1) trap around the properties not participating and increase trapping/monitoring efforts along the perimeter; (2) not trap in the region at all because of the potential effects of non-participating properties; or (3) spend more money on marketing and engagement strategies/incentives to engage with reluctant landowners to gain permission to trap on their land.

In this paper, our goal was to explore how increasing the area of untrapped land influences pest control management, financial costs, and potential impacts on biodiversity outcomes. We developed an individual-based simulation model that is applicable for rats, possums, and stoats across a range of landscape types. For each species, we modelled the impact of increasing the untrapped area, around which trapping intensity was increased in a buffer area to capture invaders. The impact of

DOI: https://doi.org/10.20417/nzjecol.49.3063

this management problem could vary with habitat types because the carrying capacity should influence our ability to maintain the population at a low level in the trapped area as well as the number of potential dispersers from the un-trapped area into to trapped area. We addressed the following three questions for low, medium, and high carrying-capacity habitats, such as drylands, beech, and podocarp forests, respectively. First, how do annual trapping costs and residual predator density over the entire area of interest change with increasing size of the untrapped area? Second, how do residual predator densities increase in the trapped area outside of non-participating properties with increasing size of the untrapped area? Lastly, using evidence from density impact functions (Norbury et al. 2015), what is the minimum untrapped area above which we would expect a decline in benefits for native flora and fauna? Agencies can use the results to aid in decisions on how to allocate funds in an area targeted for predator control or whether to target an area at all. In certain situations, it may be more cost effective to spend time and money to engage with a landowner (e.g. Sketch et al. 2019) or to move trapping efforts to another region where there is sufficient landowner participation.

Methods

Here, we describe the simulation model; all parameter values and associated references are found in Table 1. We set the total area of the landscape and the maximum size of untrapped properties as relative to the expected home range size of each species at low population density to simulate the effect of active removal efforts. Home range size has been shown to increase with decreasing density (Efford et al. 2016; Anderson et al 2022). We calculated the 99% home range size as a circular area with a radius of 3 times the home range parameter, σ (Efford 2004; see Table 1 for σ values). This resulted in home range sizes of 4.5 ha for rats, 19.1 ha for possums, and 255.1 ha for stoats. The corresponding full landscape extents were 10 times the home range diameters, resulting in 576 ha for rats, 2304 ha for possums and 32 400 ha for stoats. A circular property of a non-participating landowner was centrally located in the landscape and varied in size from 0 ha (i.e. 100% landowner participation) to a maximum circular area with a radius four times the home range radius (Fig. 1). We increased the radius of untrapped properties in increments equal to half the standard trap spacing for each species (Table 1; Department of Conservation 2023). The maximum property sizes were 72 ha, 290 ha and 4072 ha for rats, possums, and stoats, respectively.

The standard trapping network for each species was simulated using standard spacing between parallel transects with traps set at specified distances (Table 1; Department of Conservation 2023). The traps and transect spacing were reduced in the buffer area around the untrapped property, which extended outward to the distance of one home range diameter (Table1, Fig. 1; Department of Conservation 2023).

A stochastic, individual-based simulation model, using the parameters in Table 1, was run for 4 years with a 1-year time step for each species independently in low, medium and high carrying-capacity habitats (*K*; Table 1; King & Powell 2007; Cowan & Glen 2021; Innes and Russell 2021; King & Veale 2021). The simulation for each species-habitat combination was repeated 200 times for each untrapped property size. This allowed us to calculate the mean and 95% confidence intervals of the resulting population.

The initial number of individuals in the landscape was a random variate from a Poisson distribution with mean density

Table 1. Parameter values for each species used in the simulations. The three values for *K* are for low, medium, and high carrying-capacity landscapes, respectively. Equations (Eq.) used to derive parameters are given in the main body of the text, as indicated in the second column.

Parameter	Eq.	Rats	Possums	Stoats
pNeo (probability of being neo-phobic)[i]	1	0.03	0.03	0.03
g_0 (max probability capture)[a,b,c]	2	0.05	0.10	0.02
σ (home range parameter)[a,b,]c	2	40	80	300
pFail (probability of trap failure)	2	0.02	0.02	0.02
S (max. probability of survival)[d,e,f,g]	3	0.450	0.779	0.607
σ (survival density dependent rate)	3	2.1	3.0	2.5
K (carrying capacity (ha-1))[d,e,f,g]	3, 4	1.5, 5.0, 15.0	2.0, 8.0, 9.0	0.75, 2.5, 3.5
λ (per-capita population growth)[d,e,f,g]	4	4.5	0.8	4.5
β (recruitment density dependent rate)	4	1.65	1.93	1.50
γ (dispersal standard deviation)[d,e,f,g]		300	500	1000
Standard trap spacing[h]		50	50	200
Standard transect spacing[h]		100	200	800
Buffer trap spacing[h]		25	25	100
Buffer transect spacing[h]		75	100	750
costPerTrap	5	5	5	25
trapsPerDay	5	100	100	50
sessionsPerYear	5	3	4	3
trapNightsPerSet	5	10	1	9

[a]Anderson et al. (2022); [b]Anderson et al. (2016); [c]Vattiato et al. (2023); [d]King and Powell (2007); [e]King and Veale (2021); [f]Cowan and Glen (2021); [g]Innes and Russell (2021); [h]Department of Conservation (2023); [i]Johnstone et al. (2024).

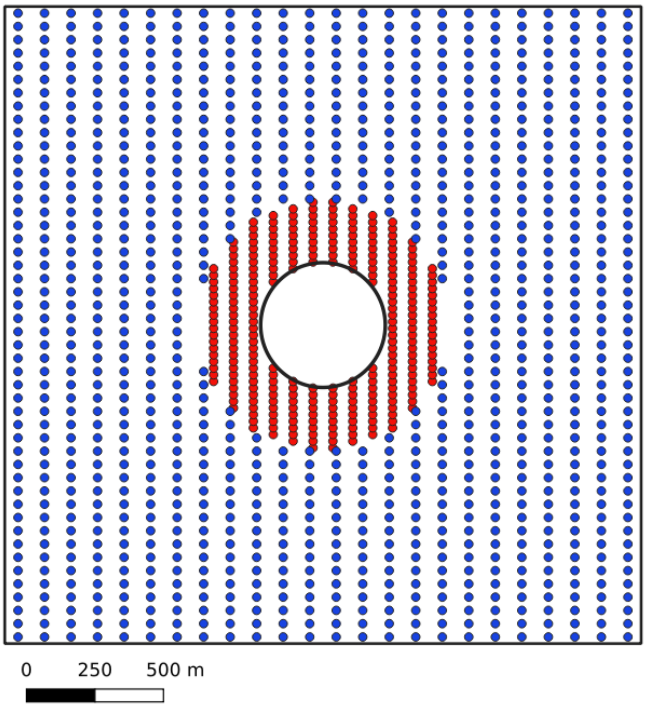

Figure 1. Spatial layout of modelled landscape for rats. The empty circular area in the middle represents a non-participating property that is 3.8 times the size of a rat home range. Blue points are rat traps spaced at 50 m along transects separated by 100 m. The red points are in the high intensity buffer area, with 25 m trap spacing along transects separated by 75 m.

of 90% of carrying capacity (0.9 *K*). The home range centres of individuals were distributed randomly across the landscape. The capture event of individual *i* of species *j* was the result of a random Bernouilli trial with probability P_{ij}, which was calculated as follows:

$$P_{ij} = \left(1 - \prod_{k=1}^{K}(1 - P_{ijk})^{nights_j}\right)(1 - pNeo_j) \quad (1)$$

where P_{ijk} was the probability of trap *k* capturing individual *i*, *nights* was the number of nights traps were deployed for species *j*, and $pNeo_j$ was the probability that an individual of species *j* would be neophobic and not interact with a device (Johnstone et al 2024; see Table 1 for species specific parameters). The P_{ijk} was given by:

$$P_{ijk} = g_{0,j} exp\left(\frac{-d_{ijk}^2}{2\sigma_j^2}\right)(1 - pFail) \quad (2)$$

where $g_{0,j}$ was the nightly probability of capture in a trap located at the centre of a home range (i.e. maximum probability of capture; Efford 2004), σ_j was the standard deviation of a normal distribution and a measure of home range size, d_{ijk} was the distance between individual *i* and trap *k*, and *pFail* was the probability of trap failure or capturing non-target species (see Table 1 for parameter values and references). The P_{ijk} decreases with increasing distance separating the home range centre and trap location (Appendix S1 in Supplementary Material). We assumed a home range radius was given by 3σ.

Following the removal of individuals, we modelled density-dependent survival of the remaining individuals. The probability of survival in year *t* ($pSurv_{ijt}$) was applied to all individuals and was calculated as follows:

$$pSurv_{ijt} = S_j exp\left(\frac{-n_i^2}{K_j^{\alpha_j}}\right) \quad (3)$$

where S_j was the maximum annual probability of survival for species *j*, n_i was the abundance in a circular area with radius $4.5\sigma_j$ around individual *i*, *K* was the carrying capacity density, and α_j was a rate parameter for the density-dependent effect on survival of species *j*. The survival event of each individual was the result of a random Bernoulli draw with $pSurv_{ijt}$. The realised probability of survival for an individual decreases with age (Appendix S2) and increasing local density (Appendix S3).

Each surviving individual contributed new recruits for *t+1* with a random Poisson variate with a mean density-dependent recruitment rate as follows:

$$recruitRate_{ijt} = \lambda_j exp\left(\frac{-n_{it}^2}{K_j^{\beta_j}}\right) \quad (4)$$

where λ_j was the maximum per-capita growth rate and β_j was a rate parameter for the density-dependent effect on recruitment for species *j*. The realised recruitment rate decreased with increasing local density (Appendix S4). The combined species-specific survival, recruitment and density-dependent parameters resulted in an equilibrium carrying capacity of approximately 5.1, 8.2, and 0.032 individuals per hectare for rats, possums, and stoats, respectively (Appendix S5).

The modelled landscape was closed so that no new individuals could disperse into or out of the area. All juveniles dispersed within the landscape prior to *t+1* in a random direction, and with random displacement in the north/south and west/east directions (*dy* and *dx*, respectively). The *dy* and *dx* were random variates from a normal distribution with mean 0 and standard deviation of γ_j (Appendix S6).

We estimated the relative annual costs of trapping for each non-compliant property scenario for each species. Recognising that cost estimates can vary widely, our aim was to estimate costs that were proportional to real costs to facilitate the comparison of predator-specific control outcomes with changes in the relative cost. Rough cost estimates were based on the use of DOC200 or DOC150 traps for stoats, snap traps for rats, and leg-hold traps for possums. We assumed a uniform landscape with no hills or valleys so that there was uniform accessibility and habitat quality. The density of traps in buffer zones was increased based on recommendations about the amount of trapping necessary along the edges of untrapped areas (Department of Conservation 2023; King 2023). The relative annual cost for a scenario was calculated as the estimated cost divided by the cost with 100% compliance. The estimated cost was the sum of fixed costs and annual labour costs. The fixed cost was the cost per trap ($costPerTrap_j$) times the number of traps deployed times each session ($trapsPerSession_j$), which was determined by the trap and transect spacing parameters (Table 1). The annual labour cost for species *j* was calculated as follows:

$$labourCost_j = \frac{trapsPerSession_j \times sessionsPerYear_j \times dayRate}{trapsPerDay_j} \quad (5)$$

where $sessionsPerYear_j$ was the number of annual trapping sessions, $trapsPerDay_j$ was the number of traps a contractor can service per day, and *dayRate* was the daily contractor fee, which was set at $400 per day for all species.

To assess results, we graphed population density (km^{-2})

and proportional costs (ha^{-1}) against the ratio of the untrapped area to home range area. For example, an untrapped area to home range ratio of 5:1 for rats corresponds to an untrapped circular property of 4.5 ha × 5 = 22.5 ha. The proportional costs for a species were the calculated costs of a scenario compared to the baseline cost, in which there was 100% participation by landowners. We reported the predator density outcome for both the entire area and for the trapped area alone (i.e. excluding the individuals in the un-trapped area).

We identified the minimum untrapped to home range area ratio that we would expect to result in benefits for native flora and fauna. We used density impact thresholds of 5% trapping or 5% tracking rate (Innes et al. 2004; Norbury et al. 2015), above which biodiversity benefits would be expected to decline. Using the g_0 and σ parameters from this study (Table 1), this translates into densities of about 20 km^{-2}, 13 km^{-2}, and 0.4 km^{-2} for rats, possums, and stoats, respectively. We refer to these as the biodiversity impact thresholds.

Results

Rats

Low carrying-capacity landscape

The predator density increased with an increase in the untrapped area from 0 (complete participation) to 66 ha (untrapped to home-range area ratio of 15:1; Appendix S7). Rat trapping costs peaked at an 8:1 ratio. The density of rats (km^{-2}) did not exceed the biodiversity impact threshold (20 rats km^{-2}) even at the largest untrapped area (Appendix S7). Focusing only on areas outside of the untrapped property, the rat density (km^{-2}) likewise did not exceed the biodiversity impact threshold (Appendix S7).

Medium carrying-capacity landscape

The predator density increased dramatically with an increase in the untrapped area from 0 (complete participation) to 66 ha (untrapped to home-range area ratio of 15:1; Fig. 2). Rat trapping costs peaked at an 8:1 ratio. The density of rats (km^{-2}) exceeded the biodiversity impact threshold (20 rats km^{-2}) at a

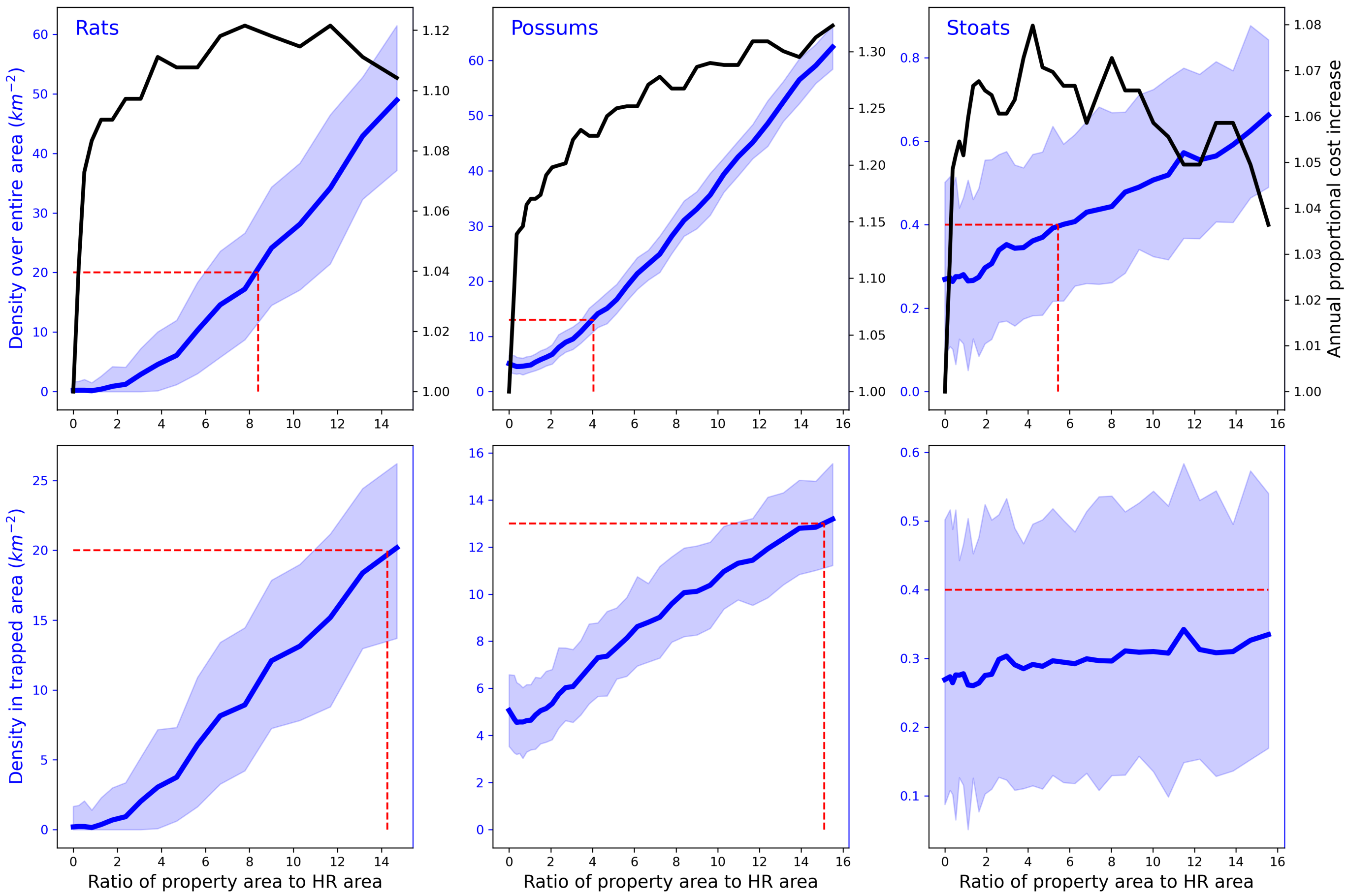

Figure 2. Population density (km^{-2}; blue line and left y axis) in a medium carrying-capacity landscape (e.g. beech forest), and cost results for rats (left column), possums (middle column), and stoats (right column). The top row of graphs shows the mean density (blue line) across the entire area at the end of four years of trapping across a range of property to home range area ratios. The light blue area represents the 95% confidence intervals. The black line in the top row shows the annual proportional cost increase over baseline costs (100% landowner compliance). The bottom row shows the mean and 95% confidence intervals of population density only in the trapped area (i.e. excluding the non-participating property). The horizontal dashed red line shows the population density that should result in a 5% tracking rate of detection devices with the species specific g_0 and σ used in this study. The vertical dashed red line demonstrates the ratio of non-participating property to home range area that will on average result in a population density that will exceed a 5% tracking rate and density impact threshold.

ratio of c. 8.2:1, or 36.9 ha of untrapped area (Fig. 2). Focusing only on areas outside of the untrapped property, the rat density (km^{-2}) exceeded the biodiversity impact threshold at a ratio of c. 14.3:1, or 64.4 ha of untrapped area (Fig. 2). Finally, the trapping costs at the biodiversity impact threshold were c. 4% higher than at the baseline where all property owners participated (Fig. 2).

High carrying-capacity landscape

The predator density increased with an increase in the untrapped area from 0 (complete participation) to 66 ha (untrapped to home-range area ratio of 15:1; Appendix S8). Rat trapping costs peaked at an 8:1 ratio. The density of rats (km^{-2}) exceeded the biodiversity impact threshold (20 rats km^{-2}) at a ratio of c. 5.2:1, or 23.4 ha of untrapped area (Appendix S8). Focusing only on areas outside of the untrapped property, the rat density (km^{-2}) exceeded the biodiversity impact threshold at a ratio of c. 6.2:1, or 27.9 ha of untrapped area (Appendix S8). Finally, the trapping costs at the biodiversity impact threshold were c. 2.0% higher than at the baseline where all property owners participated (Appendix S8).

Possums

Low carrying-capacity landscape

Possum density increased linearly with increasing untrapped area (Appendix S7). Across the entire 2304 ha, trapping costs peaked at a ratio of c. 16:1. The density of possums (km^{-2}) exceeded the biodiversity impact threshold (13 possums km^{-2}) at a 16:1 ratio, or 305.6 ha of untrapped area. Focusing only on areas outside of the untrapped property, possum density did not exceed the biodiversity impact threshold. Finally, the trapping costs at the biodiversity impact threshold were c. 27% higher than at the baseline where all property owners participated (Appendix S7).

Medium carrying-capacity landscape

Possum density increased linearly with increasing untrapped area (Fig. 2). Across the entire 2304 ha, trapping costs peaked at a ratio of c. 15.5:1. The density of possums (km^{-2}) exceeded the biodiversity impact threshold (13 possums km^{-2}) at a 3.8:1 ratio, or 72.6 ha of untrapped area. Focusing only on areas outside of the untrapped property, possum density exceeded the biodiversity impact threshold at a 15.5:1 ratio, or 296 ha of untrapped area. Finally, the trapping costs at the biodiversity impact threshold were c. 5% higher than at the baseline where all property owners participated (Fig. 2).

High carrying-capacity landscape

Possum density increased linearly with increasing untrapped area (Appendix S8). Across the entire 2304 ha, trapping costs peaked at a ratio of c. 16:1. The density of possums (km^{-2}) exceeded the biodiversity impact threshold (13 possums km^{-2}) at a 3.7:1 ratio, or 70.7 ha of untrapped area. Focusing only on areas outside of the untrapped property, possum density exceeded the biodiversity impact threshold at a 12:1 ratio, or 229.2 ha of untrapped area. Finally, the trapping costs at the biodiversity impact threshold were c. 5.6% higher than at the baseline where all property owners participated (Appendix S8).

Stoats

Low carrying-capacity landscape

The density of stoats increased linearly across the entire 32 400 ha, and trapping costs peaked at a 4.7:1 ratio of untrapped area to home range size (Appendix S7). The density of stoats did not exceed the biodiversity impact threshold (0.4 stoats km^{-2}), even at the largest untrapped area. Focusing only on areas outside of the untrapped property, the stoat density (km^{-2}) likewise did not exceed the biodiversity impact threshold (Appendix S7).

Medium carrying-capacity landscape

The density of stoats increased linearly across the entire 32 400 ha, and trapping costs peaked at a 4.7:1 ratio of untrapped area to home range size (Fig. 2). The biodiversity impact threshold of 0.4 stoats km^{-2} was surpassed for the full extent when the untrapped area to home range size ratio was 8:1, or approximately 452 ha of untrapped area. Focusing only on areas outside of the untrapped property, the stoat density (km^{-2}) exceeded the biodiversity impact threshold at a ratio of 2.3:1, or 586.6 ha of untrapped area. Finally, the trapping costs at the biodiversity impact threshold were c. 4% higher than at the baseline where all property owners participated (Fig. 2).

High carrying-capacity landscape

The density of stoats increased linearly across the entire 32 400 ha, and trapping costs peaked at a 4.7:1 ratio of untrapped area to home range size (Appendix S8). The biodiversity impact threshold of 0.4 stoats km^{-2} was exceeded at a 1:1 ratio, or 255 ha of untrapped area. Focusing only on areas outside of the untrapped property, the stoat density (km^{-2}) again exceeded the biodiversity impact threshold at a 1:1 ratio, or 255 ha of untrapped area (Appendix S8).

Discussion

Conservationists have suggested that a realistic strategy for predator control is long-term management to keep predator densities at low levels, because effective eradication is too difficult (e.g. King 2023). One could have management strategies to keep predator density to a low enough threshold that native flora and fauna benefit (Norbury et al. 2015). Below, we focused our discussion only on our modeling results for medium carrying-capacity landscapes. For a medium carrying-capacity landscape, the ratio of untrapped land to home range size cannot surpass 8.2, 3.8, and 1.8 for rats, possums, and stoats, respectively, to obtain positive outcomes for biodiversity (Fig. 2). For managed areas that are 576 ha for rats, 2304 ha for possums, and 32 400 ha for stoats, untrapped land cannot surpass 36.9 ha for rats, 72.6 ha for possums, and 459 ha for stoats. Beyond these thresholds, biodiversity outcomes may decline, even with increased trapping along the borders of the untrapped property. This conclusion comes with several caveats. We only modelled large circles of untrapped areas; if the actual untrapped area is more rectangular and not as wide as the home range of the predator in question, then the untrapped area may not have as much of an impact on native flora and fauna because the predator has a chance of encountering a trap outside the untrapped area. Also, the boundaries of the overall simulation assumed no immigration from outside areas, which if present would require increased trapping intensity along the outside periphery. Another caveat is that we only increased the size of one untrapped area in our model. Several landowners in an area of interest may not allow trapping, and the ways that multiple untrapped areas would affect outcomes are not clear.

Interestingly, even with a doubling of trapping effort around untrapped areas in medium carrying-capacity landscapes, extra costs only go up a little (e.g. 4–5% more expensive for untrapped areas with sizes at the biodiversity threshold). As the untrapped area grows, at some point overall costs reduce, due to larger areas remaining untrapped. Of course, one is not meeting biodiversity goals with larger and larger untrapped areas. An increased trapping effort is necessary at the perimeter of untrapped areas to minimise the number of predators going into the trapped area, but it is not perfect. Our medium carrying-capacity landscape model demonstrates that large untrapped areas are unsustainable for biodiversity outcomes; if one is trying to control all three predators, assuming no immigration from outside the area of interest, untrapped areas should not be larger than 36.9 ha (i.e. 8.2 times the home range size of rats). However, the 4–5% extra cost may be a significant amount of money for larger trapped areas. Although we did not model actual costs, because costs for a trapped area would be unique and have different constraints (e.g. terrain, accessibility, types of traps used, etc.), conservation initiatives could approximate the additional costs for untrapped areas based on the percentages from the model outputs. For example, if a region of interest is extremely heterogeneous and mountainous (compared to a uniform, flat region), one could have the higher cost estimates of the mountainous area factored into the 4–5% extra cost.

We noted some general trends in the low and high carrying-capacity landscapes. With low carrying-capacity landscapes (e.g. drylands), even as untrapped areas increased to the maximum size, densities of all three predators did not exceed biodiversity thresholds. Possums, though, had a biodiversity threshold at the largest untrapped area modelled (305 ha) at a 33% increase in cost from the baseline. This could indicate that trapping efforts in low carrying-capacity landscapes could still keep predator densities low enough to have good biodiversity outcomes, even when relatively large areas of untrapped land occur in the region. Conversely, in high carrying-capacity landscapes, the untrapped areas had to be smaller than in medium carrying-capacity landscapes to achieve good biodiversity outcomes. Because of the high growth potential, untrapped areas produce more predators and potential immigrants into the trapped area (relative to lower carrying-capacity landscapes), and survivors in the trapped area can produce more recruits.

Conservation organisations have to make decisions about the extra costs associated with untrapped land. For example, if it costs an extra $20,000 per year (compared to 100% cooperation), a possible strategy would be to spend money on marketing and other engagement activities to convince landowners to participate. This marketing strategy will probably vary from location to location. For example, urban citizens use social media more than rural landowners (Pick et al. 2015), and social media may work with urban citizens whereas multiple in-person visits and community meetings are better with rural landowners. In urban neighborhoods, educational signage could educate and engage residents to trap predators and implement conservation practices (Hostetler et al. 2008). Other options include paying the landowner to get access to the property, but this amount would need to be split among all participating landowners. Also, one could spend more money on the buffer area, increasing the size of the buffer or increasing the monitoring or trapping efforts in the buffer. All of these options need to be monitored and evaluated for success to determine if the money was well spent. Finally, one could consider not trapping the region and moving to an area with 100% cooperation or only small areas of untrapped properties.

In some situations, it may be important to focus on control of the predators in the trapped area while limiting the number of predators coming from the untrapped area. For example, conservation organisations may be deploying traps in forests surrounding a small city where urban trapping does not occur on 100% of all properties. In this type of situation, the model results indicate that certain sizes of untrapped urban area may still yield adequate biodiversity outcomes (i.e. in medium carrying-capacity landscapes with untrapped land to home range ratios of 14.3:1 for rats, 15.5:1 for possums, and 2.3:1 for stoats).

Overall, effective predator control and biodiversity outcomes are contingent on many factors and vary from one region to the next. Our modelling suggests that the probability of keeping predators at low levels depends on limiting the sizes of the untrapped properties. While the density impact thresholds of a 5% trapping or 5% tracking rate are based on a review of multiple studies, biodiversity outcomes could be quite variable depending on the flora and fauna of interest. Further, we assumed a uniform landscape with no hills or valleys so that there was uniform accessibility and habitat quality. Although we report on the greatest extent of untrapped land possible for good biodiversity outcomes, future research is needed to see how outcomes compare across a range of untrapped areas with more complex topography and habitat. The model and parameters presented in this paper will help decision makers evaluate how to allocate predator control budgets.

Acknowledgements

Graham Hickling provided comments to improve an early draft of the manuscript.

Additional information and declarations

Author contributions: MH and DPA conceptualised the project and undertook the investigation; DPA wrote computer code and ran simulations; MH and DPA analysed results and prepared the manuscript.

Conflicts of interest: The authors declare no conflicts of interest.

Funding: There was no specific funding associated with this project.

Data and code availability: Computer code is available at the Manaaki Whenua Landcare Research DataStore: https://doi.org/10.7931/2vcd-5208.

Ethics: Ethics approval was not required for this research.

References

Anderson DP, McMurtrie P, Edge KA, Baxter PWJ, Byrom AE 2016. Inferential and forward projection modeling to evaluate options for controlling invasive mammals on islands. Ecological Applications 26: 2548–2559.

Anderson DP, Rouco C, Latham MC, Warburton B 2022. Understanding spatially explicit capture–recapture parameters for informing invasive animal management. Ecosphere 13: e4269.

Cowan PE, Glen AS 2021. *Trichosurus vulpecula*. In: King

CM, Forsyth DM eds. The handbook of New Zealand mammals. 3rd edn. Melbourne, CSIRO Publishing. Pp. 43–77.
Department of Conservation 2023. Practical guide to trapping. https://www.doc.govt.nz/globalassets/documents/conservation/threats-and-impacts/pf2050/trapping-guide-pf2050.pdf (accessed 23 September 2024).
Efford M 2004. Density estimation in live-trapping studies. Oikos 106: 598–610.
Efford MG, Dawson DK, Jhala YV, Qureshi Q 2016. Density-dependent home-range size revealed by spatially explicit capture-recapture. Ecography 39: 676–688.
Hostetler M, Swiman E, Prizzia A, Noiseux K 2008. Reaching residents of green communities: Evaluation of a unique environmental education program. Applied Environmental Education & Communication 7(3): 114–124.
Innes J, Nugent G, Prime K, Spurr EB 2004. Responses of kukupa (*Hemiphaga novaeseelandiae*) and other birds to mammal pest control at Motatau, Northland. New Zealand Journal of Ecology 28(1): 73–81.
Innes JG, Russell JC 2021. *Rattus rattus*. In: King CM, Forsyth DM eds. The handbook of New Zealand mammals. 3rd edn. Melbourne, CSIRO Publishing. Pp. 161–240.
Johnstone KC, Price C, Garvey PM 2024. Personality, sex and capture biases: Challenges for predator monitoring and management. Journal of Applied Ecology 61(9): 2207–2218.
King CK 2023. Asking the right questions about Predator Free New Zealand. New Zealand Journal of Ecology 47(1): 3558.
King CM, Powell RA 2007. The natural history of weasels and stoats: ecology, behavior and management. 2nd edn. Oxford, Oxford University Press. 446 p.
King CM, Veale AJ 2021. *Mustela erminea.* In: King CM, Forsyth DM eds. The handbook of New Zealand mammals. 3rd edn. Melbourne, CSIRO Publishing. Pp. 285–341.
Leathwick JR, Byrom, AE 2023. The rise and rise of predator control: a panacea, or a distraction from conservation goals? New Zealand Journal of Ecology 47(1): 3515.
Norbury GL, Pech RP, Byrom AE, Innes J 2015. Density-impact functions for terrestrial vertebrate pests and indigenous biota: Guidelines for conservation managers. Biological Conservation 191: 409–420.
Parkes JP, Murphy E 2003. Management of introduced mammals in New Zealand. New Zealand Journal of Zoology 30: 335–359.
Parkes JP, Graham N, Forsyth DM, Byrom AE, Pech RP, Warburton B, Choquenot D 2017. Past, present and two potential futures for managing New Zealand's mammalian pests. New Zealand Journal of Ecology 41(1): 151–161.
Peters MA, Hamilton D, Eames C, Innes J, Mason NWH 2016. The current state of community-based environmental monitoring in New Zealand. New Zealand Journal of Ecology 40(3): 279–288.
Pick JB, Sarkar A, Rosales J 2015. A spatial and regression analysis of social media in the United States counties. 2015 Proceedings of the pre-ICIS 2015 Conference Workshop 3. http://aisel.aisnet.org/siggis2015/3.
Sketch, M, Dayer AA, Metcalf AL 2019. Engaging landowners in the conservation conversation through landowner-listening workshops. Society & Natural Resources 33(5): 669–680.
Vattiato G, Binny RN, Davidson SJ, Byrom AE, Anderson DP, Plank MJ, Carpenter JK, James A 2023. Detectability of ten invasive mammal pests in New Zealand: a synthesis of spatial detection parameters. New Zealand Journal of Ecology 47(1): 3552.

Received: 10 October 2024; accepted: 30 April 2025
Editorial board member: Zachary Carter

Supplementary material

Additional supporting information may be found in the online version of this article.

Appendix S1. Probability of detection or trapping decreases with increasing distance from a trap as a function of σ_j for each species.

Appendix S2. Mean age specific survival for the three species.

Appendix S3. Density dependent annual probability of survival for the three species.

Appendix S4. Density dependent per-capita recruitment rate for each species.

Appendix S5. Simulated non-trapped population growth for each species, which stabilises at the carrying capacity.

Appendix S6. Frequency distributions of dispersing juveniles for each species, with 95% confidence intervals.

Appendix S7. Population density in a low carrying-capacity landscape.

Appendix S8. Population density in a high carrying-capacity landscape.

The New Zealand Journal of Ecology provides online supporting information supplied by the authors where this may assist readers. Such materials are peer-reviewed and copy-edited but any issues relating to this information (other than missing files) should be addressed to the authors.