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**Impacts of mice alone on biodiversity:
final report of a Waikato field trial**



Impacts of mice alone on biodiversity: final report of a Waikato field trial

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Summary

Project and Client

- Predator-proof fences have enabled multi-species eradications of mammals from areas up to 3,400 ha on mainland New Zealand. House mice (*Mus musculus*) are the only mammals remaining in many sanctuaries, and may reach high population density with unknown biodiversity impacts. This report summarises a 5-year study of mouse abundance and impacts on native biodiversity at predator-fenced sites at Maungatautari, Waikato. This research was funded by the Ministry of Business, Innovation and Employment's Science and Innovation Group Core funds to Landcare Research, and by Waikato Regional Council and Auckland Council.

Objectives

- To determine the abundance of mice in the absence of other mammals and their impacts on forest invertebrates, seedlings, fungi, and birds.

Methods

- We estimated the abundances of mice and some known mouse-vulnerable taxa, with identical methods in a forest block with initially many mice (up to 45 mice per hectare; known as Q block) and an adjacent forest with few mice (between 2 and 24 mice per hectare; M block), from April 2011, to test whether there were biodiversity differences between the two blocks potentially attributable to mice. No other pest mammal species were present at either site. In August 2013, Maungatautari Ecological Island staff eradicated mice from the Q block, while mouse control was withdrawn and mice allowed to increase in M block. Measures of mice and biodiversity continued for another 2 years to examine the effect of this treatment switch.
- We estimated mouse population density (mice per hectare) with quarterly capture-mark-recapture trapping in live-capture traps. We also obtained relative indices of mouse abundance from inked footprint tracking tunnels. Both methods were applied from April 2011 to February 2016. The capture-mark-recapture data were analysed with a spatially explicit capture recapture (SECR) method.
- We sampled invertebrates, focusing on taxa known to be vulnerable to mouse predation including beetles, spiders, wētā, and caterpillars, using pitfall trapping and leaf litter extraction.
- Wētā were also monitored using footprint tracking tunnels.
- Land snails were extracted from leaf litter samples.
- We estimated earthworm abundance, biomass and species richness by hand-searching leaf litter and soil to a depth of 10 cm in quadrats at 20 sampling points in both blocks.
- We recorded the density of small seedlings in 36 systematically-placed plots in each block.

- We filmed a fungi 'cafeteria' site available to foraging mice, and examined mouse faecal pellets for fungal DNA and hyphae.
- We placed 40 used real bird nests containing both real and artificial eggs in the Q block, and checked them daily for 5 days to see if any were preyed on. Three nests were filmed with infra-red lit digital video cameras after the third day.
- We placed baited wax-tags and footprint tracking tunnels up to 16 m high in trees in the Q block as a preliminary test of whether mice were present in the forest canopy. Later, we undertook more intensive study of mouse arboreality at Maungatautari, with Waikato University.
- To compare the similarity of forest structure and composition between Q and M blocks, the presence of vascular species in each of 6 fixed height tiers (<30 cm, 30 cm – 2 m, 2–5 m, 5–12 m, 12–20 m, and >20 m) was recorded at the 36 seedling plots.

Results

- In Q block, mouse population density reached 29–45 mice per hectare after summer breeding each year, until mice were eradicated in August 2013. In M block, mice were undetectable at first, but increased up to 24 per hectare after intensive mouse control stopped in late 2011.
- The relationship between mouse density and the percentage of ink footprint tracking tunnels tracked by mice was not statistically significant after accounting for temporal autocorrelation. Additional replicated study sites would be needed for a strong test of the relationship between these two measures.
- Ground-dwelling invertebrates (especially beetles, spiders, caterpillars, and wētā) were both more abundant (2.8 invertebrates caught per trap night on M block) and individuals were on average larger in M block during low mouse density than in the Q block with higher mouse density. Three to eight months after the treatment switch, invertebrate abundance was similar (1.5 invertebrates caught per trap night on M block) in the two blocks. However, 15 months after the treatment switch, more invertebrates were caught in Q block with no mice, than in M block with increasing mouse density. In addition, larger beetles and weta were found in Q block with no mice than in M block at that time. Trends were similar for abundance of beetles, spiders, and wētā and for beetle species richness.
- Tracking rates of both adult Auckland tree wētā and other weta in footprint tracking tunnels were inversely related to the tracking rates of mice.
- Land snail community structure differed between the two blocks, but there was no evidence that mice influenced snail abundance and mean size of individuals.
- Native earthworm abundance, biomass and species richness were significantly higher in M block than in Q block in November 2013, but this pattern reversed by November 2015, 2 years after mouse eradication in Q. In addition, exotic earthworm species were more frequently in Q block found after mouse eradication.

- There was no evidence that mice influenced total seedling density, total seedling species richness, or the density of individual plant species on the two blocks over the study period.
- Fungal DNA and spores were detected in small numbers from mouse faecal pellets, but the results suggest that fungi are not a major component of mouse diets.
- After 5 days' exposure, a third of the 40 artificial bird nests were preyed on, but fewer quail eggs (30 × 24 mm) were eaten than finch (14 × 9 mm) or canary eggs (16 × 11 mm). Mice visited two filmed nests and preyed on eggs at one.
- When mouse densities were high in Q block, mouse footprint tracking rates at ground level averaged 93% after one night. Mice tracked ten (67%) of tunnels placed on branches 2 m above the ground for 6 nights in November 2011, and two (15%) of 13 tunnels placed 8–20 m above ground for 7 nights in May 2012. In 2015, mice were detected at 17% of wax-tag-track devices 5 m above the ground at Maungatautari locations where mice were the sole mammal, while at nearby Te Tapui Scenic Reserve, where multiple pest mammals were present, few mice were detected, and all those were on the ground.
- Q and M block did not differ significantly in forest structure or in plant species composition.

Conclusions

- In the absence of other mammals, mouse densities (8–45 mice per hectare) in Q block were relatively high and similar to estimates in beech forest and alpine ecosystems after masting events. Densities were lower in M block throughout the study (6–24 per hectare), even in the absence of intensive mouse control during the final 2 years of the study.
- Predation by mice affects invertebrate (including earthworms) communities adversely at Maungatautari by reducing invertebrate abundance and removing larger individuals selectively. This conclusion is based on an inverse relation between mouse abundance and invertebrate abundance and body size. However, mouse impacts are in total likely to be much smaller than those of the full suite of pest mammals that were eradicated at Maungatautari, and highly vulnerable invertebrates probably disappeared centuries ago due to predation by kiore (*Rattus exulans*). From a conservation perspective, it is encouraging that the recovery of invertebrate abundance and increases in the body size of beetles and wētā occurred relatively quickly (within 15 months) after mouse eradication in the Q block.
- Predation by mice did not significantly influence plant regeneration.
- Mice will eat unattended small (<30 mm length) bird eggs and some fungi, but it is unknown whether they prey on eggs or chicks at attended nests, or influence fungal biomass or dynamics.
- Mice are excellent climbers, present in trees up to 11 m above ground level.

Recommendations

- When they are the only mammal species present, mice become abundant and are likely to have significant impacts both on invertebrates in their prey size range, such as wētā, and on litter invertebrate biomass. Mouse control may be worthwhile to improve or maintain these values.
- If high-value conservation taxa, such as kakapo, are translocated into Maungatautari or other fenced sanctuaries where mice are the only mammal species present, we recommend that mice should be regarded as potential predators until proven otherwise.
- Further research is required to study the likely impacts of mice on nesting birds, lizards, and frogs.

1 Introduction

House mice (*Mus musculus*) are among the most ubiquitous mammals, owing to their potentially rapid population growth rate, varied diet, and close association with humans (King 2005). Mice are the smallest (mean weight range 17–26 g) of the four rodent species introduced to New Zealand, arriving after Norway rats (*Rattus norvegicus*) and before ship rats (*R. rattus*), as stowaways on ships in the early 1820s. House mice quickly spread and by the early 1900s occupied most suitable habitats throughout North and South Islands (Ruscoe & Murphy 2005).

New Zealand studies have shown that mice eat a range of small invertebrates (3–12 mm long) and plant material. Caterpillars are often the most common invertebrate group eaten by mice in forests, followed by spiders, beetles, and wētā (Ruscoe & Murphy 2005; Jones & Toft 2006). Fitzgerald et al. (1996) examined the stomach contents of mice from podocarp-broadleaf forest in the Orongorongo Valley, southern North Island, and found that invertebrates accounted for 94% of the stomach contents; these included caterpillars (in 51% of stomachs), spiders (45%), beetles (27%), and wētā (17%). On Rangitoto Island, Miller and Miller (1995) also found that invertebrates were the major component of mouse diet, with wētā (*Hemideina thoracica*) being a dominant prey item. Mice also eat a range of plant seeds (Ruscoe & Murphy 2005), destroying most of them (Williams et al. 2000). In alpine habitat in Fiordland, South Island, the diet of mice was dominated by wētā, spiders, and grasshoppers (Wilson & Lee 2010). On Gough Island in the South Atlantic, Jones et al. (2002) found that during the winter, earthworms inhabiting the leaf litter or soil layers close to the surface became an important food item for mice.

Worldwide, as well as in New Zealand, mice in unmanaged urban, rural, and wild ecosystems are typically uncommon and inconspicuous when food is scarce and other competing and predatory mammal species are present. In New Zealand, larger predatory mammals such as ship rats and Norway rats, mustelids (stoat *Mustela erminea*, ferret *M. putorius*, and weasel *M. nivalis*) and cats (*Felis catus*) limit mouse abundance or activity and obscure their impacts (Innes et al. 1995; King et al. 1996; Ruscoe & Murphy 2005; Harper & Cabrera 2010; Bridgman et al. 2013).

In contrast, overseas examples show that mice alone on islands or in agricultural crops with abundant food and few or no predators can become extremely abundant and cause substantial damage to biodiversity and crop yields (Pech et al. 1999; Jones et al. 2003; Angel & Cooper 2006). The revelation that mice on Gough Island ate the undefended chicks of large seabirds including albatrosses (Cuthbert & Hilton 2004) greatly extended understanding of their potential impacts. This led, for example, to Angel et al. (2009) questioning whether impacts of mice alone on islands should be regarded as equivalent to those of rats.

Mouse populations also increase when mice are the only remaining terrestrial mammals on New Zealand islands (Newman 1994; Russell 2012) or when food supplies become plentiful, such as during mast seeding of beech trees (King 1983; Fitzgerald et al. 1996), podocarp trees (Ruscoe et al. 2004), and tussock grasses (Wilson & Lee 2010). Recently, mice have frequently been the only mammal species surviving in mainland pest-fenced sanctuaries (Innes et al. 2011), either because they survive eradication attempts or because they

subsequently reinvade through fences that exclude larger pest mammals. In this situation, as on oceanic islands, mice become very abundant and may prevent the achievement of the predator-free objective and biodiversity restoration goals (Goldwater et al. 2012).

This report describes research that examined impacts of mice on biodiversity inside such a pest-fenced sanctuary in the central Waikato Region, North Island, New Zealand. It updates an interim report (Innes et al. 2014) that summarised the research at the midpoint (2.5 years) of the project. Therefore, some sections of the text are repeated from Innes et al. (2014).

2 Background

Maungatautari (3400 ha) is an extinct andesitic volcano in the Waikato, North Island. It has varying forest types ranging from lowland rimu (*Dacrydium cupressinum*) and tawa (*Beilschmiedia tawa*) forest to montane forest dominated by tawari (*Ixerba brexioides*), kamahi (*Weinmannia racemosa*), and tawheowheo (*Quintinnia serrata*) (Clarkson et al. 2002).

Maungatautari was enclosed by a pest-proof fence (Day & MacGibbon 2007) in August 2006 and most pest mammals were eradicated inside the fence in a prolonged operation that started in November 2006 (Speedy et al. 2007). Mice became very scarce but were probably never eradicated entirely from the reserve. Since February 2012 no further mouse control has been attempted, and mice have become increasingly abundant. In this study, we used two independently fenced sites with contrasting mouse densities at Maungatautari.

Our 'M block', the more northern of our two study sites (Fig. 1), is a small (24 ha) part of the fenced Maungatautari reserve, forming a peninsula of forest surrounded in part by farmland. We detected mice there beginning in November 2011, but not in our first two trapping and tracking sessions in April and August 2011. Some trapping and poisoning continued near M block in an attempt to kill invading ship rats. Beginning in August 2013 (after the 'treatment switch' described in the next paragraph), only rat traps were used when fence breaches were detected. However, poison was laid in error on two occasions, each time for 2 days: (1) at one fence-post south of M block in November 2014; and (2) along the fence north of M block in January 2016.

Our 'Q block' study site, approximately 100 m south of M block, is a fenced 17-ha, private forest block covenanted to the Queen Elizabeth II National Trust and separated from the main Maungatautari reserve by the width of a vehicle track. This block is also a peninsula of forest surrounded in part by farmland. It was pest-fenced independently in 2006, as a trial, and all mammal pests except mice were removed in the following 2 years. Mice were eventually eradicated using hand-spread brodifacoum poison in May 2008 and the block remained pest-free until May–August 2009. Since then, 67–100% of tracking cards placed there have shown mouse tracks (our data), but no other mammals were found. In August 2013, Maungatautari staff again eradicated mice from the site, thus affecting a 'treatment switch' between the two blocks in terms of mouse abundance.

In April 2011, Waikato Regional Council agreed to partially fund Landcare Research to take advantage of the presence of the Q block as an opportunity to examine the biodiversity impacts of mice. This was an emerging issue both at Maungatautari reserve and also at other pest-fenced sanctuaries around New Zealand where persistent mice populations were being discovered. Primary funding was provided by Landcare Research's core funds to address this question, and Auckland Council funded some additional work on mouse impacts on fungi in 2011–12.

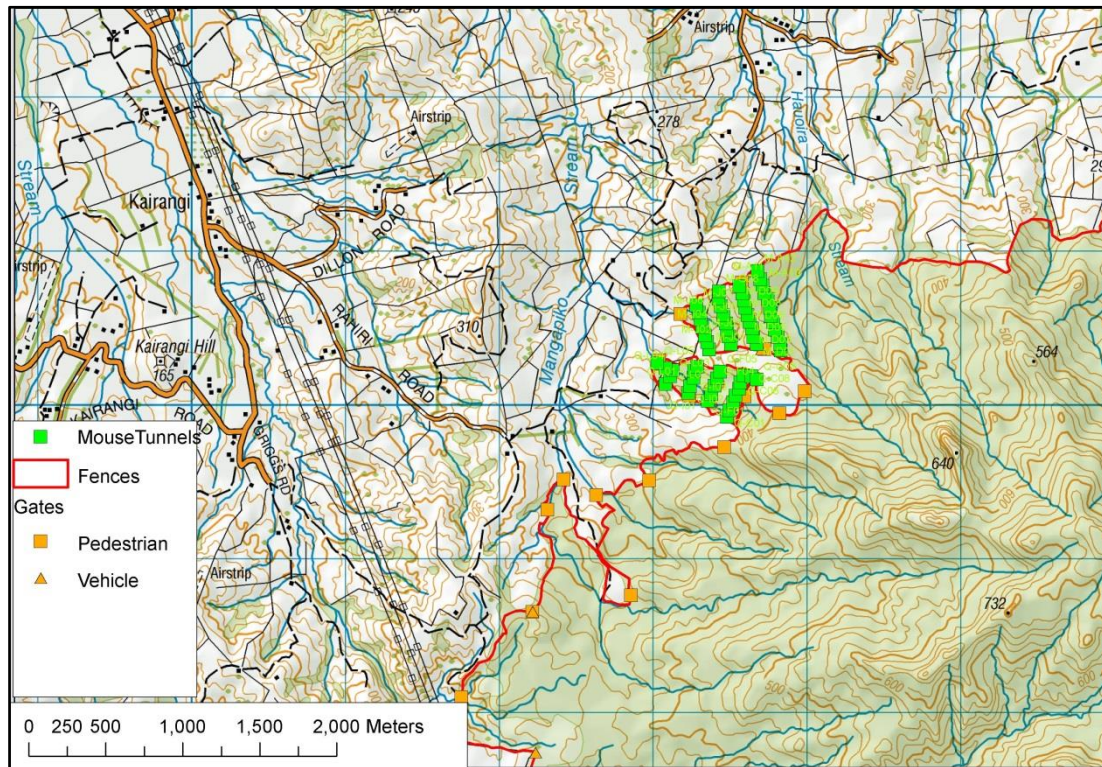


Figure 1. Locations of the two study sites on the northwest edge of 3400 ha Maungatautari, central Waikato. The Q block (Bill Garland's covenanted reserve) is the more southern. Green dots indicate footprint tracking tunnels placed to index mouse abundance.

3 Objectives

To determine the density of mice and their impacts on invertebrates, seedlings, fungi, and birds in a podocarp–broadleaved forest, in the absence of other introduced mammals.

4 Methods

The experiment began in April 2011, with ‘treatment’ (mouse abundance) reversals in August 2013. Throughout, impacts on forest invertebrates, seedlings, fungi, and birds were assessed. We estimated the densities of mice and the relative abundances of selected mouse-vulnerable taxa at two sites: i) Q block, with high (May 2011 – August 2013) followed by low (November 2013 – February 2016) mouse densities; we refer to these site-time interactions as QH and QL respectively (Fig. 4); ii) M block, with low and higher mouse densities in the time periods named above (in this report, ML and MH respectively).

4.1 Mouse abundance

4.1.1 Mouse tracking and trapping

We calculated relative indices of mouse abundance in both blocks with footprint tracking based on Department of Conservation standard procedures (Gillies & Williams 2008), but with systematic tunnel placement because of the small block sizes. Inked tracking tunnels (24 in Q block and 36 in M block) were placed 50 m apart on lines 125 m apart (Figs 2 and 3), and were baited with peanut butter for one night, in April 2011, August 2011, and then every 3 months until February 2016. Tracking rate (percentage of tunnels tracked by mice) was calculated for each occasion on each block.

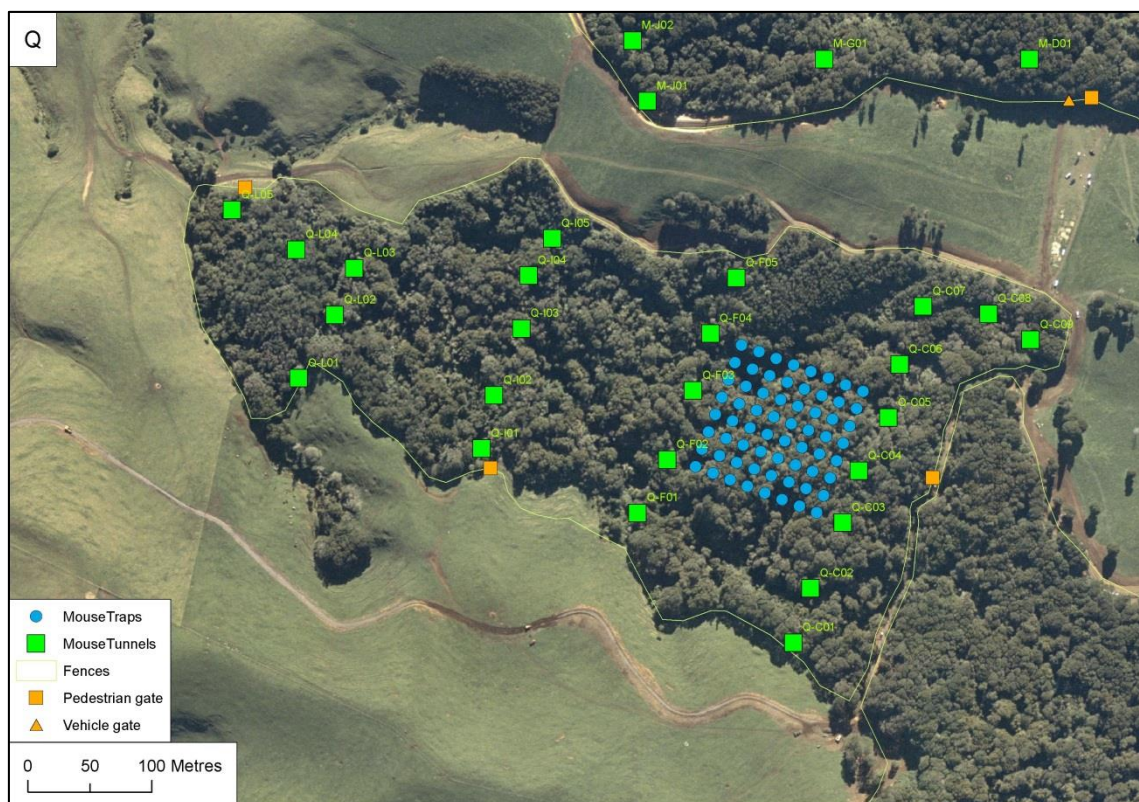


Figure 2. Layout of tracking tunnels and cage traps in the Q (Garland’s) block, Maungatautari.

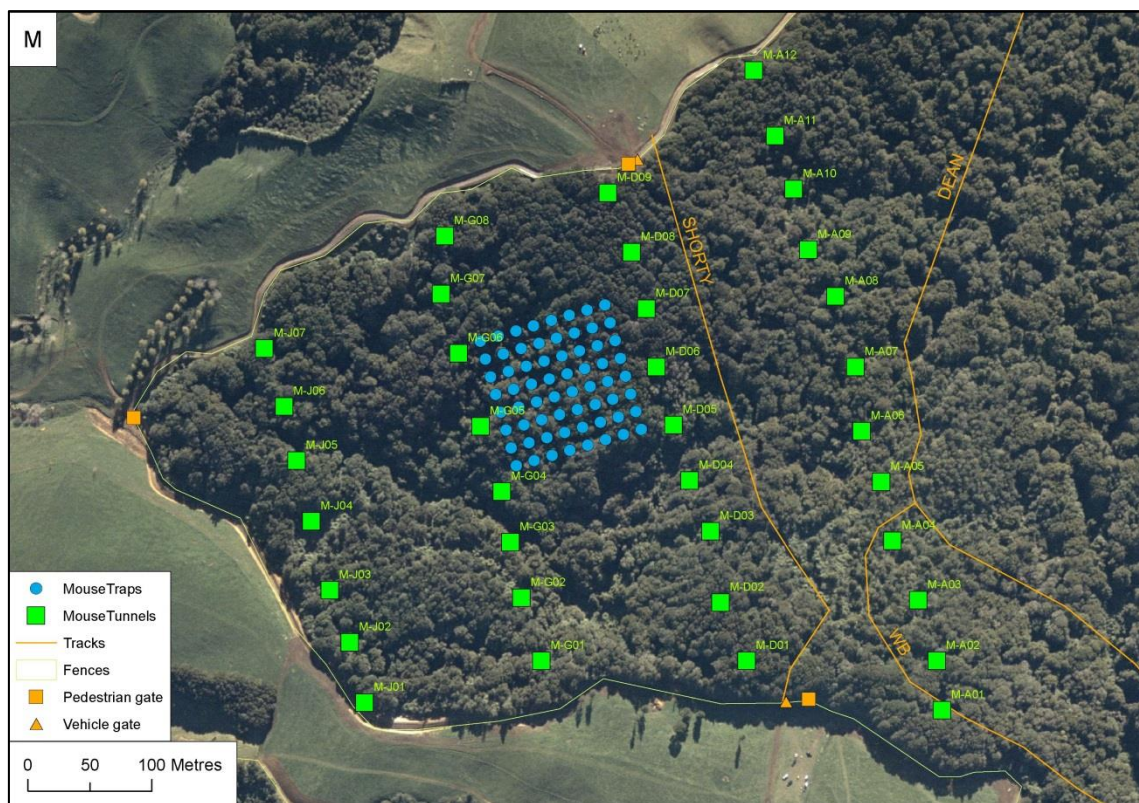


Figure 3. Layout of tracking tunnels and cage traps in M block, Maungatautari.

We estimated the population density of mice (mice per hectare) on the basis of capture-mark-recapture in live-capture traps. A grid of 64 Longworth traps (8 rows × 8 columns, with 15-m spacing) was placed in each block (Figs 2 & 3). Traps were baited with peanut butter and rolled oats and contained polyester fibre to keep captured mice warm. Traps were checked daily for 5 days, every 3 months. Each 5-day capture session began 1–7 days after tracking tunnels were checked. Captured mice were marked with numbered ear tags, weighed, sexed, and released at the point of capture.

Mouse population density estimates were obtained by analysing the data from each quarterly capture session in each block, with the secr package (spatially-explicit capture-recapture [SECR]; Borchers & Efford 2008; Efford et al. 2009) in program R (R Core Team 2016). We analysed combined data from all capture sessions, excluding the first three capture sessions in M block, which had ≤1 capture, and the last seven sessions in Q block, which had 0 captures. Combining data from multiple sessions allowed us to estimate density for sessions with few captures, i.e. many of the M block sessions (White 2005).

In the SECR method, two spatial detection parameters, g_0 and σ , are estimated from the trapping data; g_0 is the probability of capture in a trap at the centre of an animal's home range, and σ is a measure of home-range width (in metres). Spatial detection parameters were estimated using conditional likelihood, and mouse density in each capture session was then calculated as a derived parameter on the basis of captures in that session. We selected a set of alternative spatial detection models of variation in the two parameters, after

comparing the performance of alternative models during the first two years of the study on the basis of AIC_c (Burnham & Anderson 2002). We then compared alternative models fitted to all the data from 2011–2016, also on the basis of AIC_c . In the alternative model set, g_0 and σ were either both constant (i.e. the null model) or an additive function of treatment (Q block or M block) and/or season (spring, summer, autumn, winter). Models in which g_0 was also an additive function of mouse weight and/or a behavioural response to capture were also tested. Alternative types of behavioural responses to capture (Efford 2016) were considered: b , a permanent response in which an animal's probability of capture increased (a trap-happy response) or decreased (trap-shy) after its first capture; and bk , a trap-specific permanent response in which an animal became trap-happy or trap-shy in relation to a particular trap (e.g. Royle et al. 2011).

4.1.2 Relationship between mouse tracking and mouse population density

We tested whether the mouse tracking rate (the percentage of inked tracking tunnels tracked by mice) was related to mouse density with a linear model (function `gls` in package `nlme`) in program R (R Core Team 2016). This model included a correlation structure (an autoregressive process of order 1) to account for temporal autocorrelation, i.e. non-independence of sequential measures on the same block. The response variable was the percentage of tunnels tracked by mice in each block on each sampling date. Predictor variables were mouse density and block (M or Q) and their interaction. Percentages of tunnels tracked were transformed before analysis to normalise the distribution of residuals, by expressing them as proportions and calculating the arcsine of their square-roots (Zar 1996). Mouse density was \log_e -transformed before analysis, to linearise the relationship and limit effects of very high values. We omitted tracking tunnel data that were not paired with mouse density estimates, i.e. data from mouse capture sessions with ≤ 1 capture, when mouse density could not be estimated (see above). A predictor variable was considered statistically significant ($P < 0.05$) when the 95% CI (confidence interval) of the corresponding model coefficient excluded zero. We also tested for significance at $P < 0.1$ on the basis of 90% CIs.

4.2 Invertebrates

4.2.1 Sampling the invertebrate communities using pitfall traps

Ground-dwelling invertebrate fauna were sampled using pitfall traps, each a 100-mm-deep plastic cup (105-mm diameter) containing 100 ml of 50% monopropylene glycol. A pitfall trap was placed at 5-m intervals along a 45-m transect located between two tracking tunnels. Two transects were located randomly within each of the study blocks, giving a total of 20 pitfall traps in each block. Traps were set for one month in April 2011–2015, and for 3 months over summer (late November to late February) in 2011/12, 2012/3, 2013/14, 2014/15 and 2015/16. Specimens were preserved in 70% ethanol.

Captured invertebrates (>3 mm in length) were sorted and counted to Order level using a binocular microscope. We focused on caterpillars, spiders, beetles, and wētā as these have been identified previously as dominant prey items in diets of mice in the presence of other

mammals (see references in Ruscoe & Murphy 2005). Beetles were sorted on the basis of external morphology to recognised taxonomic units (hereafter referred to as species) and, where possible, given generic and species-level identifications by Stephen Thorpe (Research Associate, University of Auckland). Each beetle captured was measured (mm) and then average beetle length per trap was calculated. Average wētā size was calculated by measuring the width of the pronotum and then an average per trap was calculated.

Data are yet to be completely analysed, but preliminary results are presented as mean invertebrate abundance per trap night \pm 95% CI (confidence intervals) so that differences ($P < 0.05$) are apparent by inspection. Differences are considered probable if the CIs do not overlap.

4.2.2 Sampling the leaf-litter invertebrate communities

Litter-dwelling invertebrates were collected from one leaf-litter sample within a 33-cm diameter circular frame (0.086 m^2) at 32 sampling points within 3 m of a tracking tunnel within each block. The placement of each sampling point was chosen by using a random compass bearing and distance ($<5 \text{ m}$) from each tracking tunnel. Eight extra litter samples (2 from each line) were collected from the Q block to make equal total numbers of 32 samples from each block. Approximately every 7 days in April, in each of the sampling years, 8 litter samples were taken from each block (32 samples per block, each April for 5 years). At collection, all leaf litter and friable humus were scraped rapidly from the frame, placed in individual bags and transported back to the laboratory. Invertebrates were extracted from the leaf litter over a 72-hour period using Tullgren funnels.

The preservation, sorting and identification of invertebrates from the leaf-litter followed the same protocols used for the pitfall-trapped invertebrates. Litter weight was recorded per sample. Preliminary results are presented as mean invertebrate numbers per g of litter per $0.086 \text{ m}^2 \pm 95\%$ CI (confidence interval) so that significant differences ($P < 0.05$) are indicated by non-overlapping CIs.

4.2.3 Sampling wētā using tracking tunnels

Tracking tunnels used for monitoring mice also record the footprints of invertebrates including wētā. Each tracking card was examined for the presence of wētā footprints as these are readily recognisable. Protarsal, mesotarsal, and metatarsal prints longer than 2.5, 3.5, and 4.4 mm, respectively, indicate the presence of adult Auckland tree wētā (*H. thoracica*; Watts et al. 2011). Smaller wētā, including subadult and juvenile Auckland tree wētā, all age classes of ground wētā (*Hemiandrus* species) and all age classes of cave wētā (Rhaphidophoridae species) were recorded as 'other wētā'.

Tracking rates are expressed as the mean percentage ($\pm 1 \text{ SE}$) of tunnels tracked with adult Auckland tree wētā and other wētā footprints on the 4 lines (a total of 5–12 tunnels per line) of tracking tunnels in each block.

4.2.4 Land snails

After the 72-hour Berlese funnel extraction described in 4.2.2, land snails were extracted manually from the litter samples in autumn 2011 and 2012 from both blocks, then analysed in processes described in detail in Innes et al. (2014).

4.2.5 Earthworms

In November 2013 and 2015, we used a headlamp to hand-search leaf litter and soil to a depth of 10 cm from within 20 50 × 50 cm quadrats (0.25 m²) in each of the Q and M blocks. To avoid potential trampling effects, each sampling point was located 10 m west of a randomly selected pre-existing tracking tunnel station in 2013, and 10 m east of a tunnel in 2015.

Earthworms were immediately placed into labelled vials containing 70% ethanol; later they were dabbed dry and individuals weighed to 3 decimal places (of a gram). Identification to recognised taxonomic unit (RTU) was completed using a binocular microscope and the keys of Lee (1959) and Sims and Gerard (1986).

Because of non-normal distributions of the data, Wilcoxon rank sum tests were used for analysing the biomass, abundance and diversity data, in program R (R Core Team 2016).

4.3 Seedlings

We counted cotyledonary seedlings (with their first leaf), true-leaf seedlings (<15 cm tall) and mixed-leaf (both cotyledons and true leaves) seedlings in 36 circular plots, each 1 m², placed systematically along transects in each of Q and M blocks in April 2011 (QH; ML), April 2013 (QH; ML) and June 2016 (QL; MH). Plots were placed 5 m from tracking tunnels, measured at right angles on alternating sides of each transect. In the Q block, 12 additional plots were placed opposite other plots to make an equal total number (36) of plots in each block to achieve similar sampling densities. Total density of all seedlings (number per m²), total seedling species richness, and densities of cotyledonary, mixed-leaf and true-leaf seedlings were then calculated by species and size class.

Initial results are presented as densities of seedlings per m² ± 95% CI (confidence intervals) so that differences ($P < 0.05$) are apparent by inspection. If the CIs do not overlap then significant differences are likely.

4.4 Fungi

Mouse consumption of fungi was studied using cameras at a 'fungus cafeteria' feeding platform in July 2011, and also by DNA analysis of mouse faecal pellets collected in August 2011 and February 2012, as described in Innes et al. (2014).

4.5 Bird eggs

In February–March 2013, we and student Kelly Frogley (University of Otago) placed 40 used natural bird nests, each containing eggs of two sizes, on the ground in Q block (QH). Nests were checked after 5 days, and daily nest survival rates were estimated (see Innes et al. 2014 for details).

In 2015, Waikato University student Cat Kelly looked for bird nests at Maungatautari, and filmed at some nests to investigate possible mouse impacts (Kelly 2016).

4.6 Use of trees by mice

In order to determine mouse use of arboreal habitats we investigated their use of canopy trees. To find out whether mice were climbing trees, in December 2011 we placed 15 wax-tags and 15 baited tracking tunnels on branches 2 m above the ground for 6 nights in Q block (QH). In May 2012, we placed 13 baited footprint tracking tunnels in trees 8–20 m high for 7 nights in Q block (QH) to test whether mice were present in the forest canopy.

We and Waikato University MSc student Cat Kelly placed nine wax-tag and tracking devices at each of three heights ranging from the ground to the canopy, at each of 20 sites with high mice densities and no other mammals in the large Maungatautari reserve (not in M block). We repeated these methods at 19 sites at nearby Te Tapui Scenic Reserve where all widespread mainland pest mammal species were present, including ship rats, possums (*Trichosurus vulpecula*), cats and stoats, during January–June 2015. Devices were collected after a week.

4.7 Comparison of forest structure and composition

To compare the forest structure and composition between Q and M blocks, the presence of vascular species in each of 6 fixed height tiers (<30 cm, 30 cm–2 m, 2–5 m, 5–12 m, 12–20 m, and >20 m) was recorded at the 36 seedling plots described in Section 4.3.

PERMANOVA (Anderson 2001) was used to test for significant differences in canopy composition between plots in the Q and M blocks: (1) considering presence in all tiers simultaneously; and (2) considering each tier separately. Bray-Curtis distance was used to quantify dissimilarity between plots. PERMANOVA analyses were performed using the *adonis* function in the R (R Development Core Team 2009) package *vegan*. In addition, a detrended correspondence analysis (DCA) ordination (Hill & Gauch 1980) was performed to allow a graphical comparison of the species composition of plots in each block. This was carried out using the *decorana* function in the R package *vegan*.

5 Results

5.1 Mouse abundance

5.1.1 Mouse population density

In the best-supported model of mouse population density (i.e. the model with the lowest AIC_c), both g_0 (probability of capture of an animal in a trap at the centre of its home range) and σ (home range width) varied as a function of treatment block (Q or M) and season. g_0 also varied according to mouse weight and a behavioural response to capture b . Other models received much less support ($\Delta AIC_c > 8.8$).

Mouse population density fluctuated seasonally in both blocks, with relatively high summer or autumn densities following spring and summer reproduction, winter declines, and gradual increases during the next breeding season (Fig. 4). In QH, density estimates were 8–45 mice per hectare, until mice were eradicated in August 2013. In ML, density was apparently zero until summer 2012, when the first mouse was caught there. Thereafter, density estimates in MH increased to 2–24 per hectare but did not equal or exceed densities in QH.

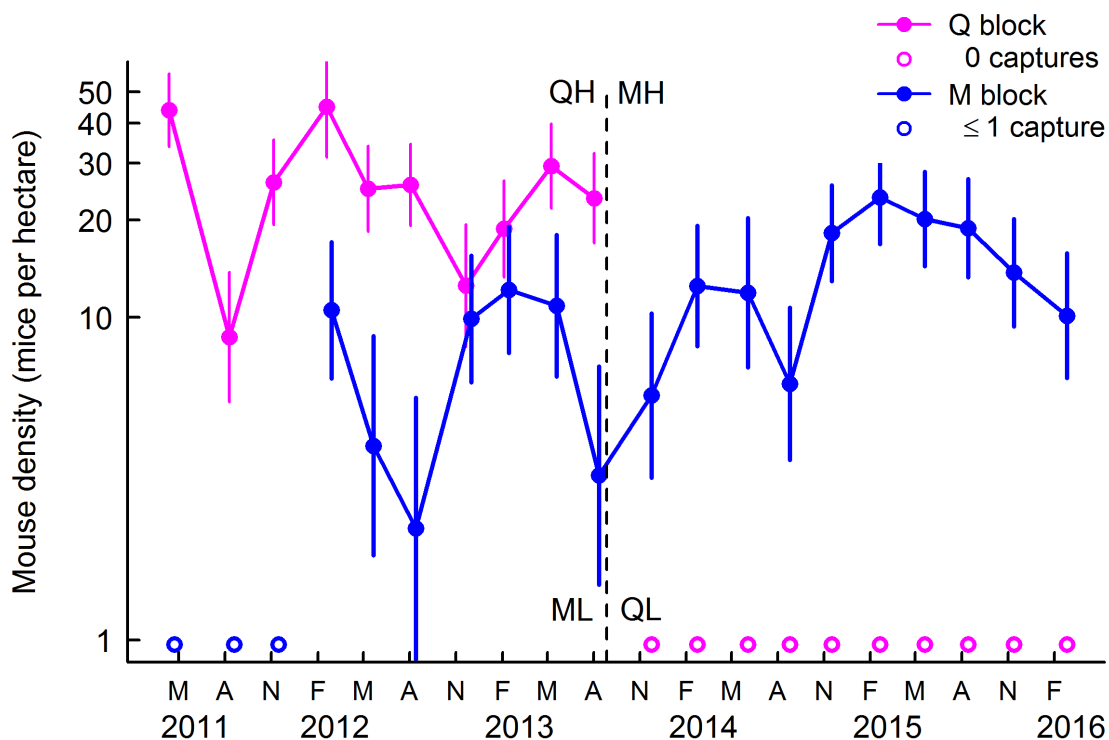


Figure 4. Estimated house mouse population density in Q and M blocks within Maungatautari Ecological Island. Vertical lines show 95% confidence intervals. Open circles show trapping sessions when density could not be estimated, i.e. when ≤ 1 mice were captured in M block (blue) early in the study, and when 0 mice were captured in Q block (pink) after eradication. Timing of the treatment switch in August 2013 is shown with a dashed vertical line. Density is plotted on a logarithmic scale. QH, QL, MH, and ML indicate mouse densities (High, Low) that switched between blocks (Q, M) at or around August 2013.

The estimated probability of capturing a mouse in a trap at the centre of its home range (parameter g_0) was higher in Q block than in M block (Fig. 5). The average home range of mice (based on parameter σ) was smaller in Q block than in M block, indicated by a more rapid decline in capture probability at increasing distance from the home range centre (Fig. 5). Because we did not model the shapes or utilisation of individual mouse home ranges, our analysis does not yield realistic home-range size estimates. Capture probability increased for mice recaptured during the same capture session (a trap-happy response; Fig. 5), and was related positively to mouse size (i.e. weight in grams). Both capture probability and home range width varied seasonally, with the capture probability point estimates highest in spring and lowest in summer, and home range width point estimates higher in summer than in other seasons.

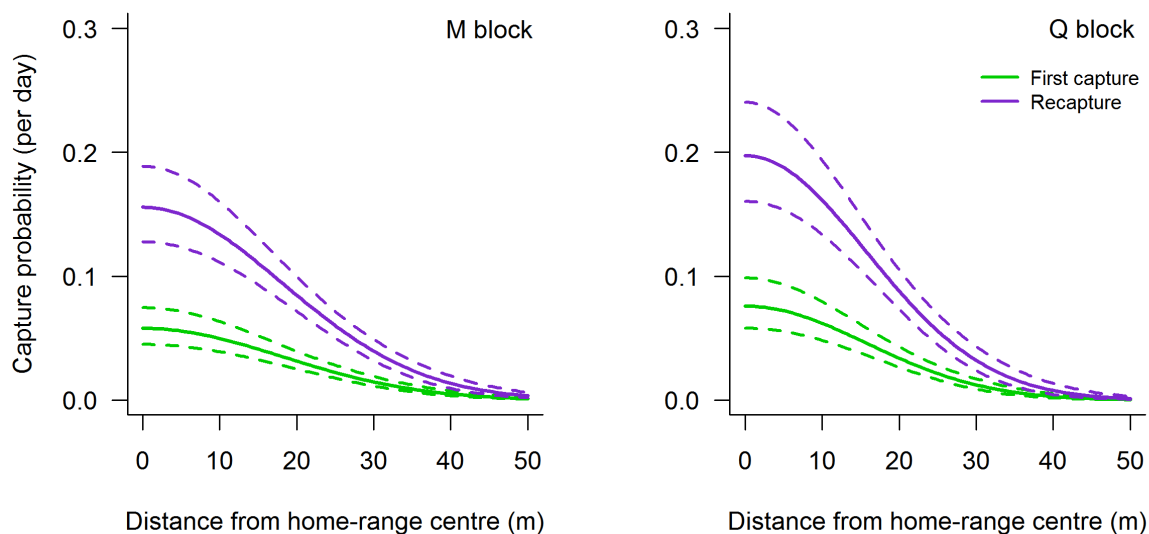


Figure 5. Modelled daily probability of capturing a mouse in a live-capture trap, as a function of distance of the trap from the centre of the home-range, in M block and Q blocks. Recapture probability refers to recapture during the same quarterly capture session. Capture probabilities for a 20-g mouse in autumn are shown. Dashed lines indicate 95% confidence intervals.

5.1.2 Relationship between mouse tracking and mouse abundance

Mouse tracking rates (percentage of inked tracking tunnels tracked by mice) were 0–11% where ≤ 1 mice were captured, i.e. in capture sessions in both Q block and M block when mouse density could not be estimated (Fig. 6). Tracking rates associated with capture sessions where >1 mice were caught ranged from 8–92% in M block and 67–100% in Q block (Fig. 6). The percentage of tunnels with mouse tracks was positively correlated with estimated mouse density in the corresponding block and capture session (Pearson's correlation coefficient $r = 0.69$, $n = 27$). However, in a linear model that accounted for temporal autocorrelation, estimated mouse density was not a statistically significant predictor of the percentage of tunnels with mouse tracks ($P > 0.1$). The fitted model explained 68% of variation in the data (R^2). Tracking rates did not differ significantly between the Q and M blocks ($P > 0.1$); nor was the interaction between tracking probability and block significant ($P > 0.1$).

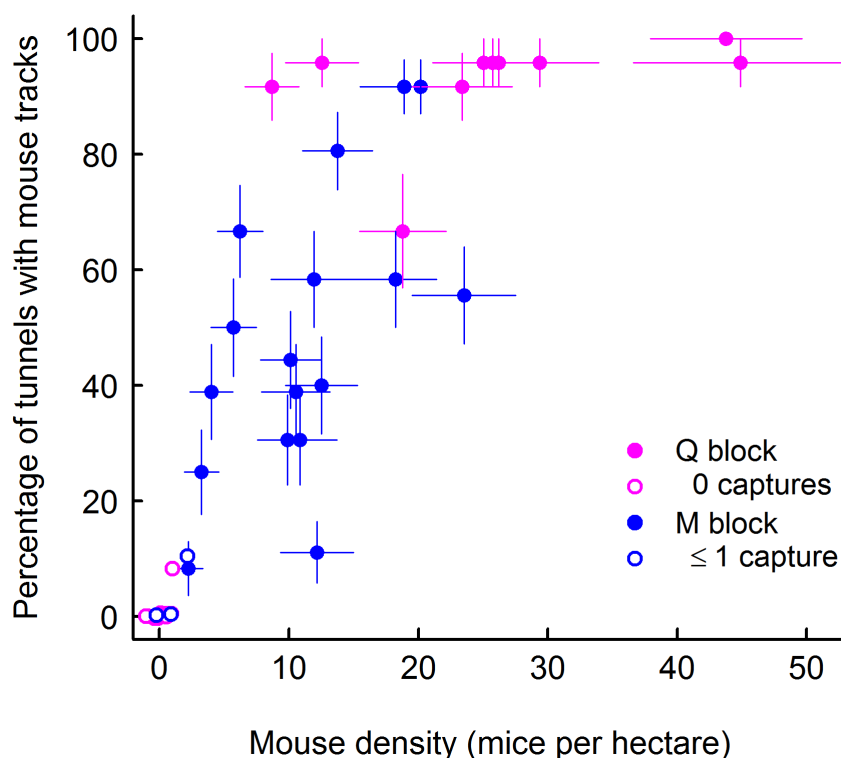


Figure 6. Mean mouse tracking rate (percentage of inked tracking tunnels tracked by mice) at the start of each trapping session, plotted as a function of estimated mouse density in Q block and M block. Vertical and horizontal lines indicate corresponding standard errors of each estimate. When there was ≤ 1 capture (open symbols), density could not be estimated and is plotted as the number of captures, with variation added (jittered) so that similar points are visible.

5.2 Invertebrates

5.2.1 Sampling the invertebrate communities using pitfall traps

A total of 42 003 invertebrates were caught in pitfall traps over the study. During the first two years of the study, more invertebrates were caught in pitfall traps in M block during low mouse densities (ML) than in the Q block with higher mouse densities (QH) (Fig. 7). At the first sampling (2013–14), 3–8 months after the treatment switch, invertebrate abundance was similar in the two blocks, and at times (Jan/Feb 2014 and Apr 2014) abundance did not differ significantly between the blocks (Fig. 7). However, 15–20 (2014–15) and 27–30 (2015–16) months after the treatment switch, more invertebrates were caught in QL than in MH (Fig. 7). Invertebrate numbers caught in Q block during this time (QL) approached the highest levels recorded in the M block with low mice (ML) before the treatment switch (Fig. 7). These abundance patterns were similar for beetles, spiders and wētā, and for beetle species richness.

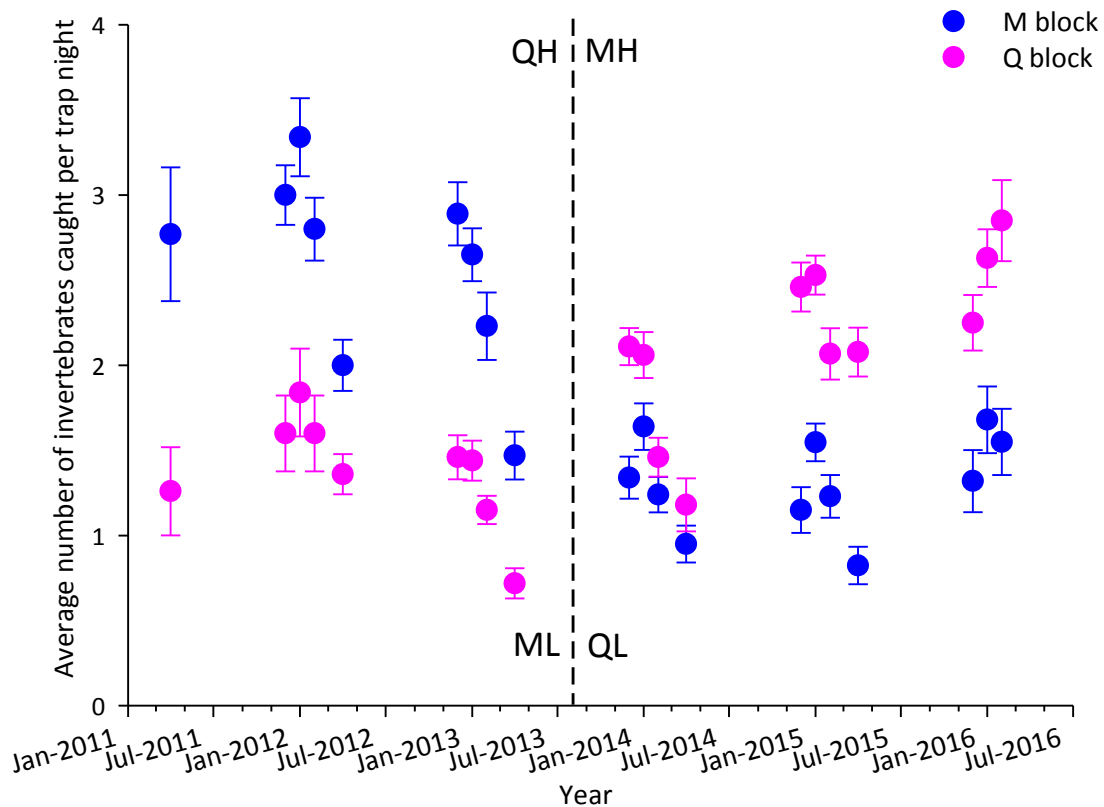


Figure 7. Average number ($\pm 95\%CI$) of invertebrates caught per trap night at Maungatautari. Time of the treatment switch in August 2013 is shown with a dashed vertical line. QH, QL, MH, and ML indicate mouse densities (High, Low) that switched between blocks (Q, M) at or around August 2013.

Before the treatment switch, beetles sampled from Q block with high mouse densities were on average half the size of those collected from M block (Fig. 8). At the first (2013–14) and second (2014–15) samplings, 3–8 and 15–20 months respectively after the treatment switch, beetle size was similar in QL and MH (Fig. 8). However, it appears that by 27–30 months after the treatment switch (2015–16 sampling) larger beetles were caught in QL than in MH (Fig. 8). Wētā showed similar results, except the sizes of wētā differed significantly between the two blocks earlier, at 15–20 months after the treatment switch.

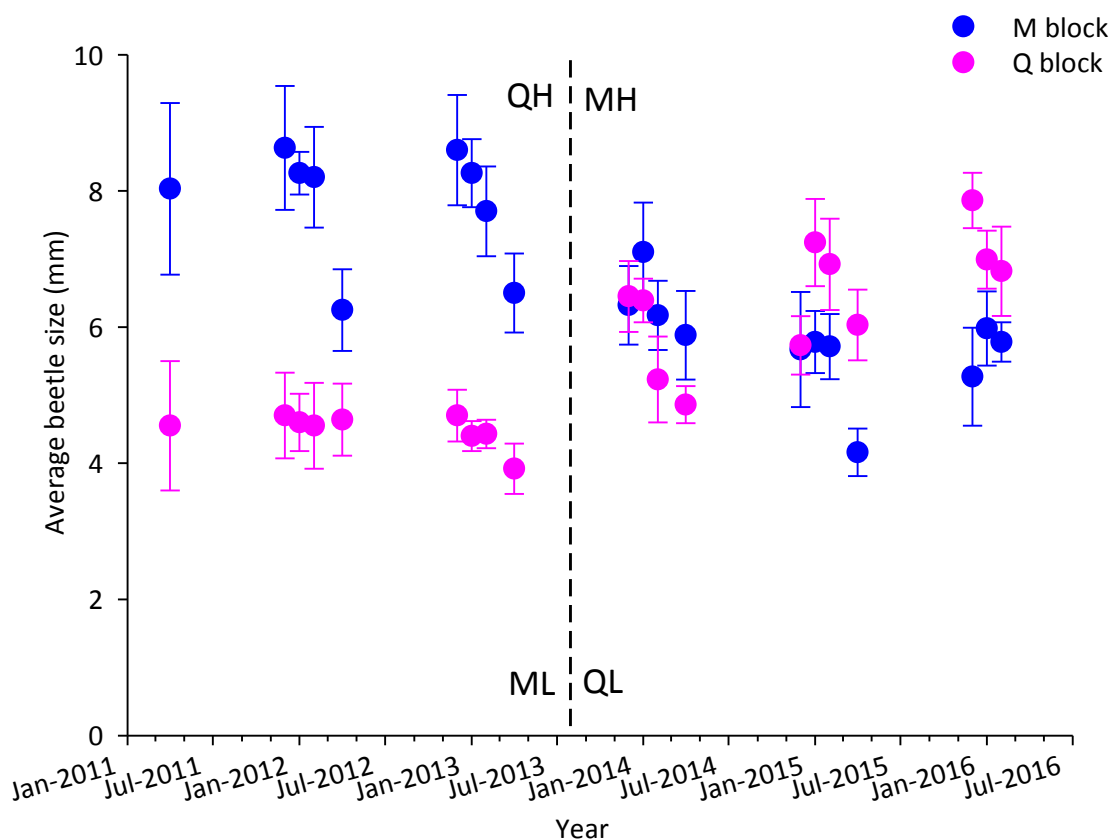


Figure 8. Average size ($\pm 95\%CI$) of beetles caught in pitfall traps at Maungatautari. Time of the treatment switch in August 2013 is shown with a dashed vertical line. QH, QL, MH, and ML indicate mouse densities (High, Low) that switched between blocks (Q, M) at or around August 2013.

5.2.2 Sampling the leaf-litter invertebrate communities

A total of 10 617 invertebrates were sampled from the leaf litter over the study. There was no difference in the average weight of litter collected between the two sites and over the duration of the study.

Litter samples showed similar results to the pitfall trap data. During April 2011, 2012 and 2013, the mean number of invertebrates collected from leaf litter in ML was significantly higher than in QH (Fig. 9), and tending to decline over time. After the treatment switch, the density of invertebrates found in the leaf litter continued to decline in the M block (Fig. 9). In contrast, 8 and 20 months after the eradication of mice in the Q block (QL), the density of leaf-litter invertebrates was increasing and was significantly higher than in M block (MH) (Fig. 9). Similar trends were observed for densities of caterpillars, spiders and beetles, and for beetle species richness.

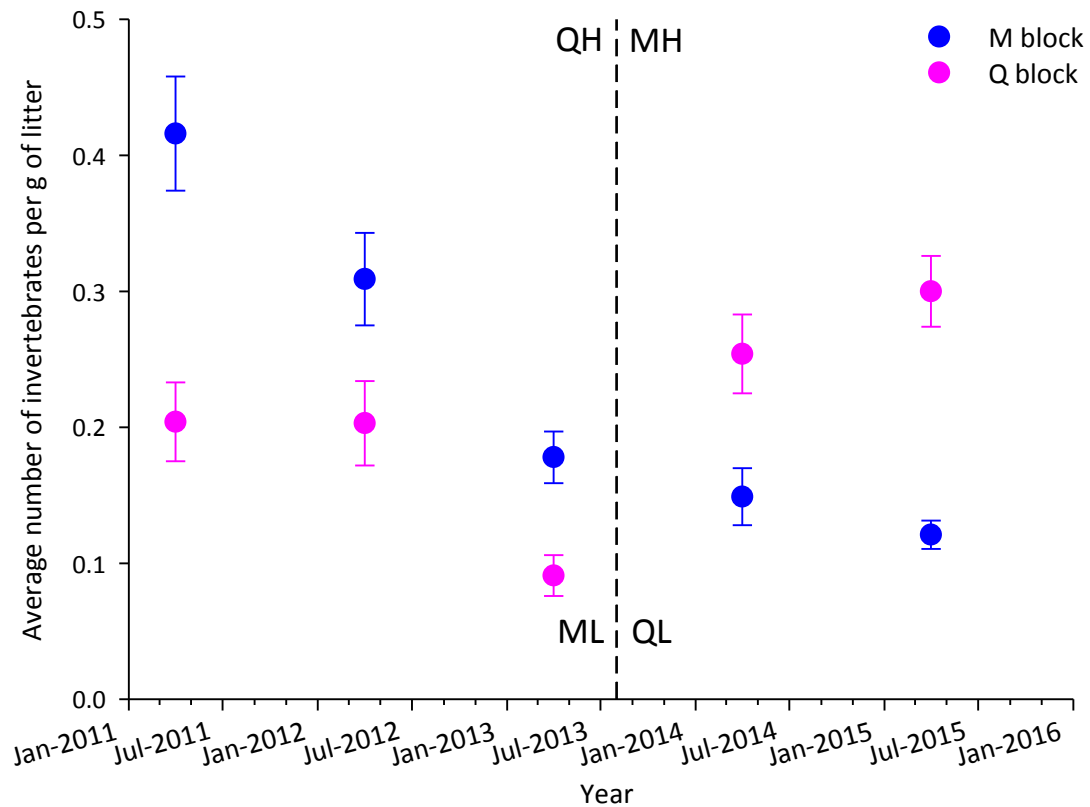


Figure 9. Average number ($\pm 95\%$ CI) of invertebrates caught in the leaf litter from Maungatautari. Time of the treatment switch in August 2013 is shown with a dashed vertical line. QH, QL, MH, and ML indicate mouse densities (High, Low) that switched between blocks (Q, M) at or around August 2013.

Before the treatment switch, beetles sampled from the leaf litter collected from ML were on average twice the size of those in QH (Fig. 10); with the beetles in M block tending to decrease in size over time. After the treatment switch, there was no significant difference in beetle size between the two blocks (Fig. 10).

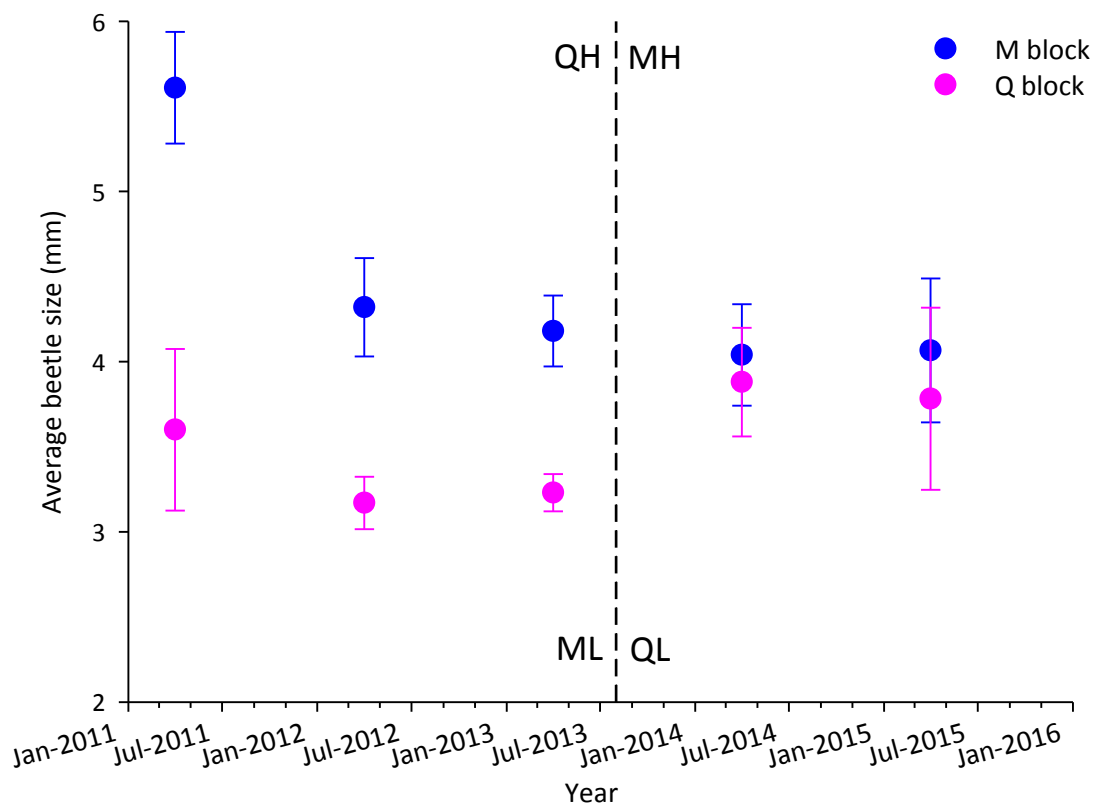


Figure 10. Average size ($\pm 95\%CI$) of beetles collected in leaf litter samples from Maungatautari. Time of the treatment switch in August 2013 is shown with a dashed vertical line. QH, QL, MH, and ML indicate mouse densities (High, Low) that switched between blocks (Q, M) at or around August 2013.

5.2.3 Sampling wētā using tracking tunnels

During the first two years of the study, mean tracking rates of adult Auckland tree wētā ranged from 38 to 61% in ML compared with tracking rates ranging from 8 to 31% in QH (Fig. 11). In the year after the treatment switch, mean tracking rates of adult Auckland tree wētā were similar in the two blocks. After that time, mean tracking rates of adult Auckland tree wētā increased to more than 50% in QL while in MH average tracking rates of adult Auckland tree wētā were always less than 20% (Fig. 11). Similar trends were observed for other wētā.

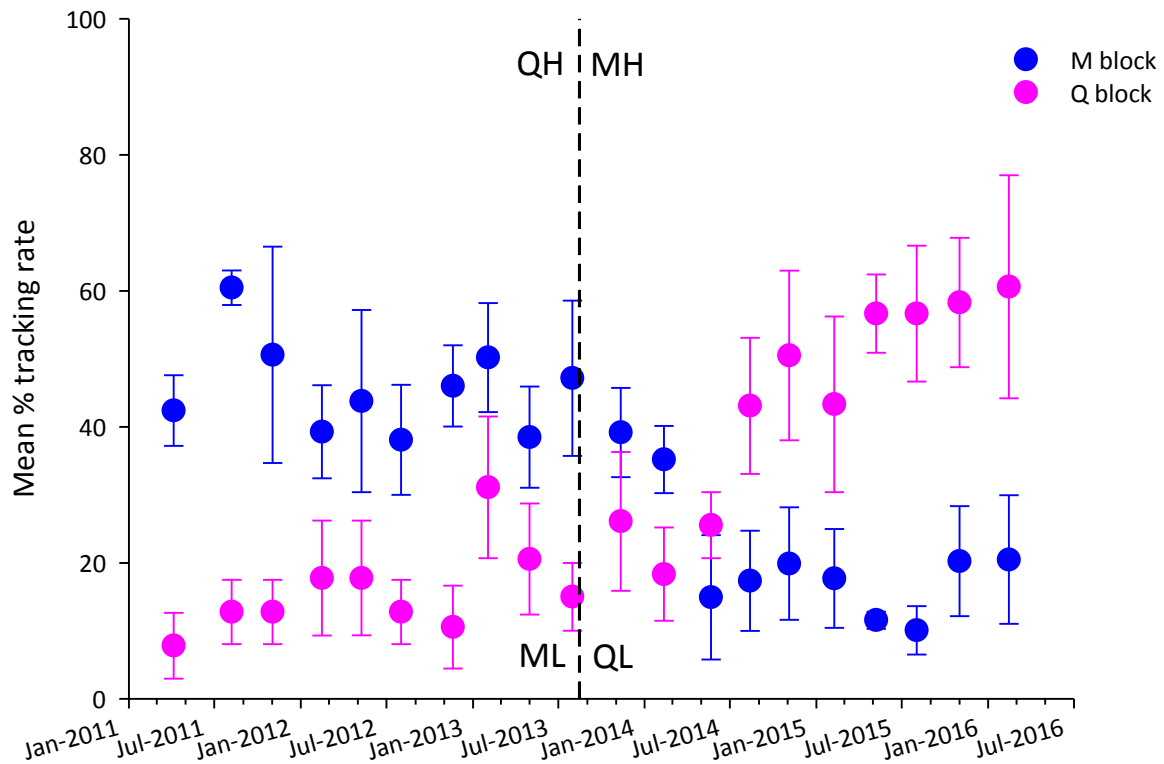


Figure 11. Mean adult Auckland tree wētā ($\pm 95\%$ CI) tracking rate (percentage of inked tracking tunnels tracked by wētā). Time of the treatment switch in August 2013 is shown with a dashed vertical line. QH, QL, MH, and ML indicate mouse densities (High, Low) that switched between blocks (Q, M) at or around August 2013.

Tracking rates of adult Auckland tree wētā were inversely related to the tracking rates of mice (Fig. 12). Again, other wētā showed comparable trends.

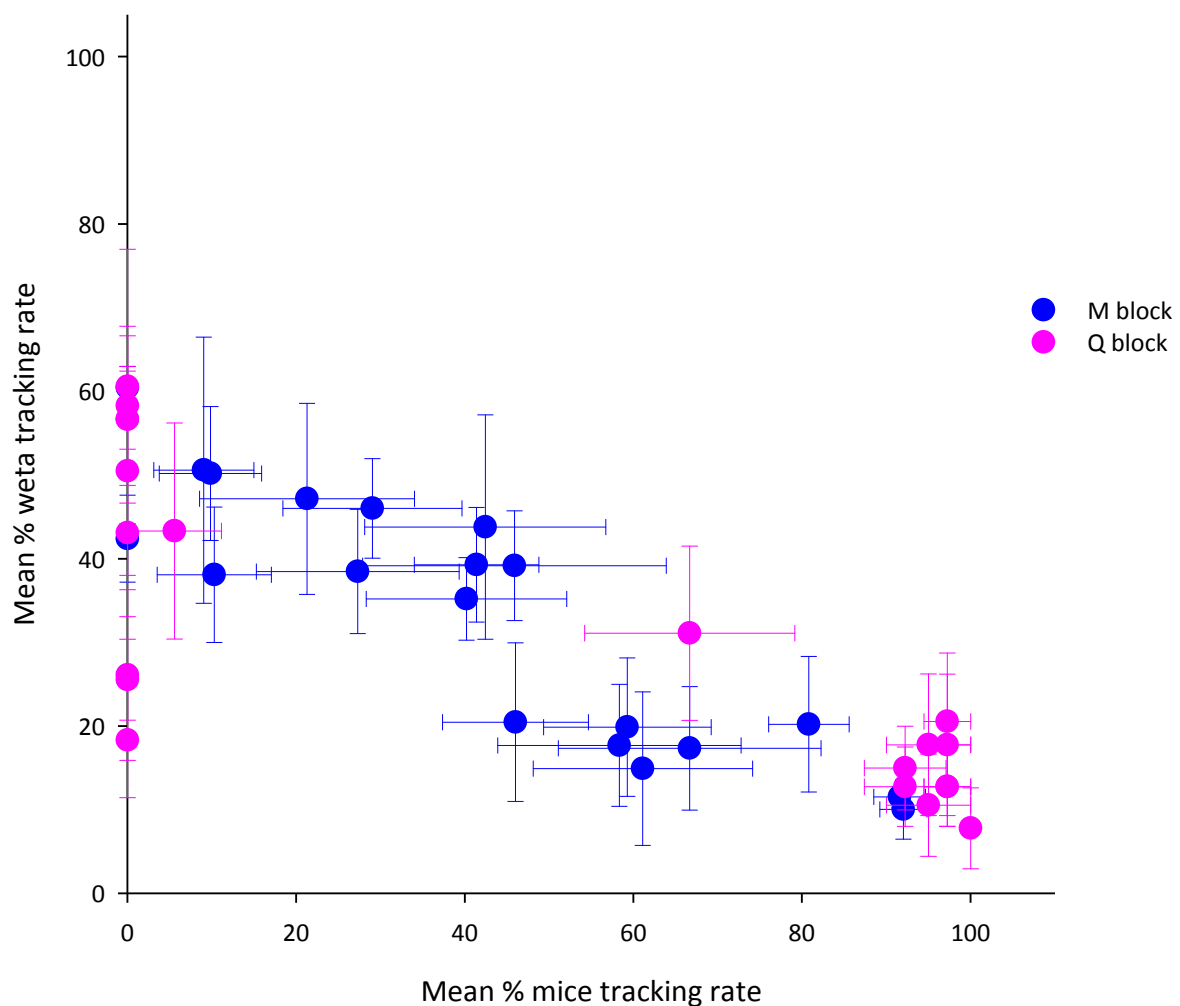


Figure 12. Relationship ($\pm 95\%CI$) between mean mouse and adult Auckland tree wētā tracking rate (percentage of inked tracking tunnels tracked by mice and weta).

5.2.4 Land snails

Significantly more snail species were found in ML than in QH in April 2011, but species richness was similar between treatment blocks in April 2012 (Innes et al. 2014). This, coupled with the absence of treatment (i.e. block) differences in abundances or mean sizes of snails (Innes et al. 2014), was interpreted as indicating inherent block differences rather than evidence of mouse effects.

5.2.5 Earthworms

There was a significant ($P = 0.015$) difference in average earthworm abundance between the two blocks, with an average earthworm abundance of $0.7 (\pm 0.36SE)$ per quadrat in Q block and $2.25 (\pm 0.69SE)$ in M block in November 2013 (Fig. 13). By November 2015, approximately 26 months after the treatment switch, the average earthworm abundance in Q block (QL) had significantly ($P < 0.001$) increased 7.4-fold to $5.15 (\pm 1.5SE)$. In M block,

there no change was detected (Fig. 13). Average earthworm biomass and RTU richness showed similar results. While M block showed little change in exotic earthworm taxa present, Q block showed an increase from no exotic earthworms in 2013 to seven taxa in 2015.

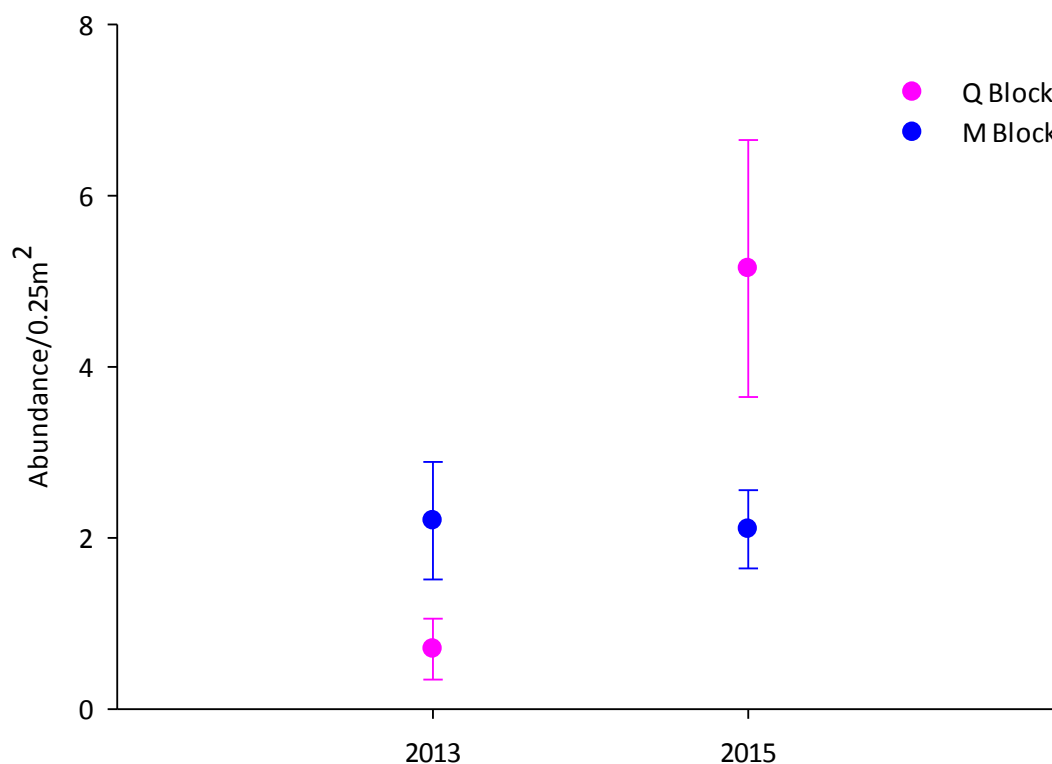


Figure 13. Average earthworm abundance per quadrat \pm 1SE.

5.3 Seedlings

A total of 4317 seedlings were measured during the study, comprising 28 indigenous species. There was no evidence that total seedling density (number per 1-m² plot), total species richness, or the density of individual plant species differed between the two blocks over the study period. Preliminary results suggest some significant differences in the density of mixed-leaf seedlings (combined cotyledonary and true-leaf stage seedlings) between sampling occasions, with more mixed-leaf seedlings recorded in Q block than in M block in 2016 (Fig. 14).

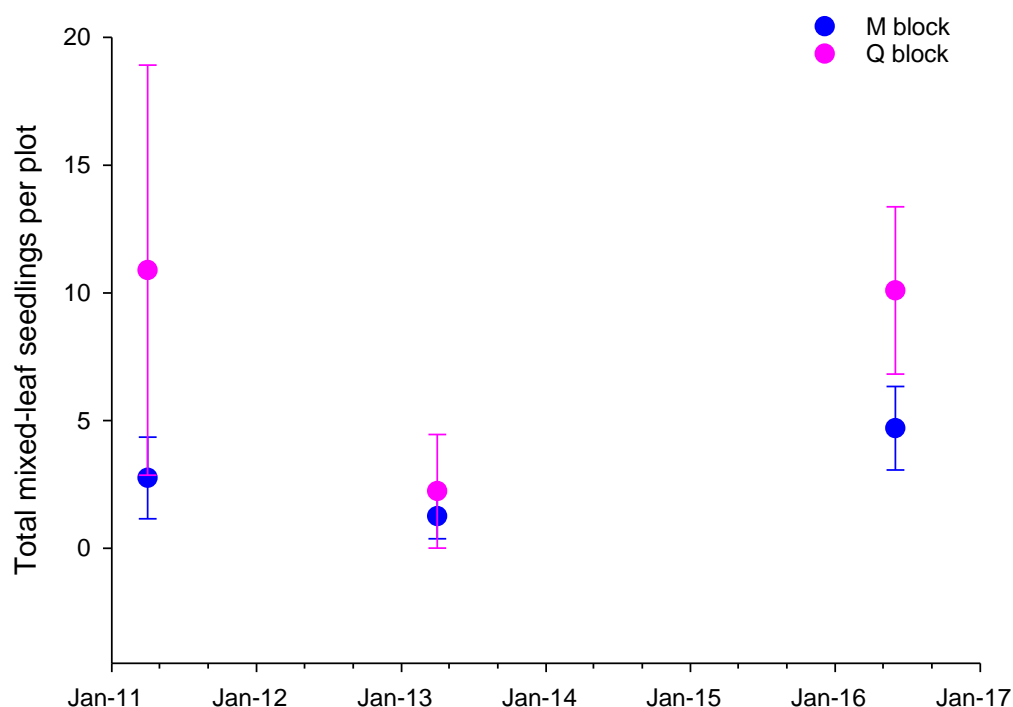


Figure 14. Total number of mixed-leaf seedlings in M and Q blocks at Maungatautari. Vertical solid lines indicate 95% CIs.

5.4 Fungi

No mice were filmed visiting fruiting bodies of known edible and other mushrooms in the 48 hours during which the fungi were presented to mice in the Q block in July 2011. Mouse density was estimated at 8 per hectare in August 2011, just after this trial (Fig. 4). Fungal DNA was amplified successfully from 14 of 54 examined faecal pellets, but good quality DNA sequence data were obtained from only three of these 14 pellets (Innes et al. 2014).

5.5 Bird eggs and nests

During the 5 days of exposure, 13 (32.5%) of the 40 used natural nests placed on the ground by Kelly Frogley had eggs preyed on by mice. Fewer relatively large eggs of Japanese quail *Coturnix japonica* (30 x 24 mm) were preyed on than either canary *Serinus canaria domestica* (16 x 11 mm) or zebra finch *Taeniopygia guttata* (14 x 9 mm) eggs.

Too few natural occupied nests were found by Cat Kelly in 2015 to draw strong conclusions about the likely impact of mice on them. She filmed one mouse at an abandoned thrush (*Turdus philomelos*) nest containing eggs (12 November 2015), but the mouse did not eat the eggs. This was the only mouse observed at any of the 6 nests with eggs, despite a total of 420 hours of filming.

5.6 Use of trees by mice

At ground level, footprint tunnel tracking rates were usually >90% in QH before mouse eradication (see also section 5.1.2). Ten (67%) of 15 tunnels placed 2 m high in trees for 6 nights were tracked by mice in QH in November 2011, and 2 (15%) of 13 tunnels placed at 8–20 m above ground for 7 nights in May 2012 were tracked. Of these high tunnels, the two that were tracked were at the low end of the height range, at 8–11 m above ground (Fig. 15). Two of 15 wax-tags placed 2 m above the ground in trees in QH were chewed by mice.

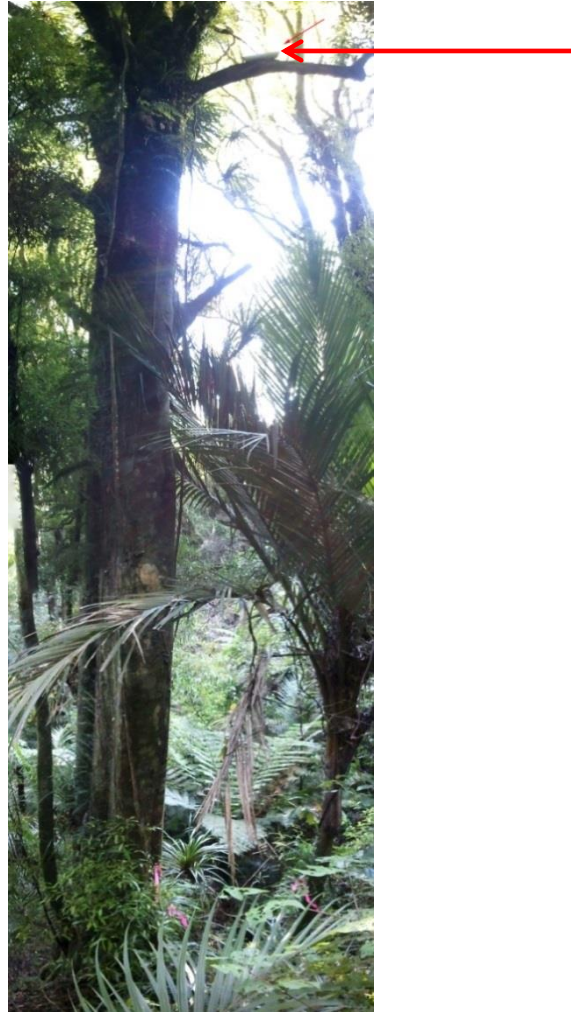


Figure 15. Baited tracking tunnel (indicated by the red arrow) 11 m above the ground in a tawa tree, tracked by a mouse in May 2012.

In January 2015, mouse detection rates with wax-tag-track devices in the main Maungatautari reserve were similar to tunnel tracking rates recorded earlier in QH: 93% of devices on the ground detected mice ($n=60$), 35% at shrub level (mean height above ground 1.6 m, $n=20$), 17% at subcanopy level (mean height above ground 5.0 m, $n=40$), and no detections at canopy level (mean height 9.2 m, $n=60$) (Kelly 2016). In contrast, at nearby Te Tapui Scenic Reserve where multiple mammalian species were present, mice were detected at only 3% of ground devices and no devices at higher levels. However, ship rats were

detected at Te Tapui at 75% of ground devices, 42% of shrub devices, 45% of subcanopy devices and 73% of canopy devices. Possums were also detected at all levels: 25% of devices on the ground, 10% in shrubs, 5% in the subcanopy and 1% in the canopy.

5.7 Comparison of forest structure and composition

There were no significant differences in vegetation composition between Q and M blocks, in terms either of presence of species in all tiers simultaneously ($P = 0.631$) or of individual tiers (Table 1).

Table 1. PERMANOVA analyses of individual height tiers analysed separately. DF indicates degrees of freedom. F-stat is the statistic used for significance testing, with the resulting P -value also given

Tier	DF Model	DF Error	F-stat	P -value (simulated)
Tier 1 (<30cm)	1	44	1.194	0.298
Tier 2 (30 cm – 2 m)	1	49	0.723	0.627
Tier 3 (2–5 m)	1	55	0.274	0.992
Tier 4 (5–12 m)	1	54	1.381	0.195
Tier 5 (12–20 m)	1	34	1.725	0.078
Tier 6 (>20 m)	1	55	1.003	0.43

DCA plot ordination did not show any discernible difference in vegetation composition between blocks (Fig. 16).

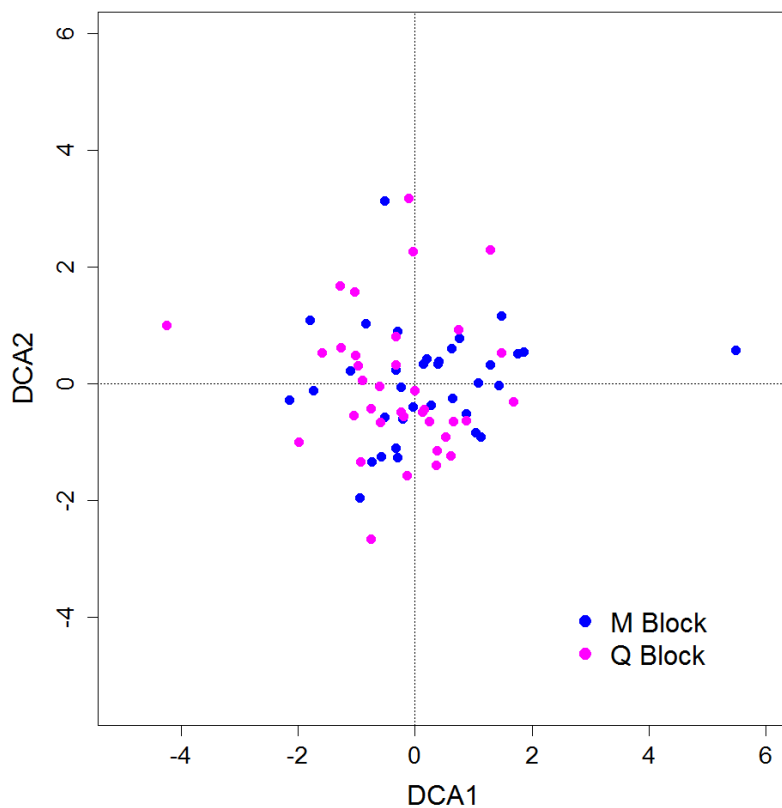


Figure 16. DCA ordination of vegetation composition in 36 plots in each of M and Q blocks on Maungatautari.

6 Discussion and conclusions

6.1 Mouse abundance

The highest population densities in Q block (up to 45 mice per hectare) were similar to estimates in forest and alpine ecosystems after masting events (Ruscoe et al. 2001, 2004; Wilson & Lee 2010), but lower than an estimate (160 per hectare) in rank kikuyu grass (*Pennisetum clandestinum*) in the partially fenced sanctuary on Tawharanui Peninsula, Auckland Region (Goldwater 2007; Goldwater et al. 2012).

Mouse density in MH (maximum 24 mice per hectare) gradually increased to levels similar to those observed in QH before eradication there, but never attained the highest densities estimated in QH in autumn 2011 and 2012. This difference between the blocks may be due to the large area of available habitat within the main fenced sanctuary where M block is located (3,400 ha), compared with the much smaller fenced area within Q block (17 ha), which may limit juvenile dispersal in Q block. Estimated mouse density in MH declined between February 2015 and February 2016, from its maximum of 24 per hectare to 10 per hectare. This decline may reflect the depletion of food for mice supplied by invertebrates. Our results suggest that invertebrate numbers built up in M block during years with very

few mice, but that predation by the growing mouse population has since depressed both abundance of invertebrates and the mean body sizes of some taxa (6.3 below).

The contrasting population trajectories within the two blocks in late winter and early spring (August–November) of 2012 (Fig. 4) provide evidence that the habitat may have been better for mice in the M block before the treatment switch. Density declined in Q block, but increased in M block during this period. This apparent density-dependence in rates of population change may be due to differences in the availability of food (invertebrates) in the two blocks (evident from our data) and possibly to other unknown effects of crowding in Q block. Mouse mortality may have been lower and/or reproduction higher in M block than in Q block during winter and early spring 2012.

The higher capture probability of mice in Q block compared with M block may also be the result of low food availability in Q block early in our study, making baited traps more attractive there. Finally, the smaller home ranges of mice in Q block (we did not estimate home-range sizes, only spatial differences in capture probability) may be attributable to the higher average mouse density restricting the movements of individuals.

6.2 Relationship between tracking and density

Our study was not designed to test whether the percentage of ink footprint tracking tunnels tracked by mice was a useful index of mouse density. The positive relationship between the two measures could have been the result of temporal autocorrelation (non-independence of sequential measures within the same block). For example, the mouse tracking rate on a given date may have been affected less by mouse density than by other factors that also determined the previous tracking rate. An alternative study design, with both tracking rate and density measured at multiple independent replicated sites, would allow a stronger test of the relationship between these two measures. Such a design was not feasible in this project.

6.3 Impacts on biodiversity

At Maungatautari, we did not detect a significant impact of mice on fungi, land snails (see Innes et al. 2014 for further details) or seedlings, although all are likely to be minor diet items, and the faecal pellets examined for fungi were not collected at the time of the main autumn fruiting. It is possible that there were mouse effects on land snails that went undetected because the sampled communities were in a poor condition as a legacy of prior mammalian predation or some other earlier disturbance. There was low snail species richness (6–16 species vs expected 40–70), low abundance (26–49 vs expected 100–1000 per litter sample) and low mean size (1.5 – 1.7 mm vs expected 2–5 mm). However, our data suggest mice may reduce the abundance and average size of other ground-dwelling invertebrates in their prey range (3–12 mm) by about half. The treatment switch between the two study blocks implemented at the midpoint of the project confirmed this inference. In addition, the average abundance of earthworms inhabiting the leaf litter or soil layers close to the surface increased 7.4-fold after the eradication of mice in Q block.

In New Zealand, invertebrates, particularly litter-dwelling caterpillars, beetles, spiders, and ground wētā, occur frequently in the diet of mice in a wide range of habitats, implying that these are significant food items (see Ruscoe & Murphy 2005). Studies have estimated that the proportion of the standing crop of invertebrates harvested daily by mice ranges from 0.7% to 2.9% (Rowe-Rowe et al. 1989; Crafford 1990; van Aarde et al. 1996). A more recent estimate by Innes et al. (2010) suggests that mice in the presence of other mammals consume 9 g of invertebrates per hectare per night in North Island podocarp-broadleaved forest. This amount is likely to be higher for mice alone at high densities. Therefore, it is not surprising that a number of studies in New Zealand have implicated mice in the decline of invertebrate populations (Bull 1967; Ramsay 1978; Brignall-Theyer 1998). For example, mouse predation has been cited as the most likely cause of the extinction of two insect species, a predatory carabid beetle (*Loxomerus* sp.) and an unidentified wētā species, on Antipodes Island (Marris 2000).

A comparison of invertebrates on islands with and without mice in the South Indian Ocean suggested that predation by mice caused a reduction in the mean body size of medium- to large-sized invertebrate taxa, significant negative effects on the populations of some invertebrate species, and disruption of the mating strategies of weevils (Crafford & Scholtz 1987; Crafford 1990; Chown & Smith 1993). When mice were eradicated from Allports Island in 1989, Fitzgerald et al. (unpubl. data) observed an increase in the abundance of eight common invertebrate species and found that a beetle species absent in 1986 became abundant by 1997/98. In contrast, van Aarde et al. (2004) constructed mouse-free exclosures on Marion Island, and found no significant effect of mice on any of eight invertebrate prey groups' abundance or biomass, or on community structure (diversity and composition). Collectively, these studies show that at some sites mice may limit invertebrates directly by predation or indirectly by competition for food such as seeds, fruits and other invertebrates.

In the present study, it appears that invertebrate abundance and body size were inversely related to mouse abundance. In addition, the inverse relationship between mean tracking rates of wētā and mice may indicate a predator- or competitor-release response of wētā to mice. Wētā are not only within the preferred prey size range of mice, but are mobile and may be particularly attractive to mice because their movements are readily detectable. From a conservation perspective, it is encouraging that the recovery of invertebrate abundance, increases in body sizes of beetles and wētā, and increase in wētā tracking rates occurred relatively rapidly (within 15 months) after mouse eradication in the Q block. Also on Maungatautari, Watts et al. (2011) found dramatic increases in wētā pitfall captures, wētā tracking rates, and the incidence of wētā footprinting per tracking card within 2 years after all mammals were initially eradicated. Overall it is encouraging that these large invertebrates remain after more than a century of predation by pest mammals.

In New Zealand, the risk of exotic earthworms invading native forests has been thought to be low (Lee 1959), although little research on the issue has been done. The increase in exotic earthworm species in QL could be due to exotic species capitalising on both available food resources and lowered competition after several years of mouse predation on native earthworm species. More survey work in future is required to reveal if these forest soil invaders are able to sustain these new higher population levels. In addition, beetle species

responses will be analysed to determine if exotic beetle taxa also increased in the beetle communities sampled.

Mice rapidly found and ate small canary and finch eggs (approximately 16 × 11 mm) and occasionally ate larger (30 × 24 mm) quail eggs, all of which were on the ground and undefended by a sitting adult bird (Innes et al. 2014). Too few attended natural nests were found at Maungatautari in 2015 for us to assess the risk of mouse predation, but the fact that only one mouse was filmed in 460 hours of nest observation suggests that it was low (Kelly 2016). Native forest species with eggs up to the size of quail eggs include hīhi (*Notiomystis cincta*), tomtit (*Petroica macrocephala*), whitehead (*Mohoua albicilla*), bellbird (*Anthornis melanura*), kākāriki (*Cyanoramphus novaezelandiae novaezelandiae*), robin (*Petroica australis*), saddleback (*Philesturnus carunculatus*), and tui (*Prothemadera novaeseelandiae*; Frogley 2013). Mice have been shown to eat eggs of Gough buntings (*Rowettia goughensis*; 28 mm egg length; Cuthbert & Hilton 2004; Ryan & Cuthbert 2008) and rock wrens (*Salpinctes obsoletus*; 22 mm egg length; O'Donnell et al. 2017) in the wild, but whether they will take eggs or young of native birds at Maungatautari remains unknown. Similarly, it is now well known that mice attack very large seabird chicks on some islands (Angel et al. 2009; Jones & Ryan 2010). It is therefore clearly plausible that they might also attack undefended chicks of ground-nesting species such as kakapo (*Strigops habroptilus*) if they were reintroduced to Maungatautari.

Mice were detected across nearly all devices set at ground level; at most devices up to 2 m above the ground, and at a few set up to 13 m high in trees (Innes et al. 2014; Kelly 2016). This finding broadens the potential of mouse impacts across ground-dwelling and arboreal invertebrates, and perhaps also to birds. However, we are aware of no accounts of mice being filmed at active bird nests in New Zealand forests in the last 20 years, even in beech (*Nothofagus* spp.) forest when mice are abundant (G. Elliott, Department of Conservation, pers. comm.).

6.4 Contextual perspectives

There are three important contextual perspectives when considering mouse impacts measured in this project. First, the indigenous biota we examined at Maungatautari has already survived c. 750 years of rodent predation and at least 150 years of impacts of other introduced pest mammals. Kiore (*Rattus exulans*) in particular would have done most damage to New Zealand invertebrates, being the first and smallest (mean weights 70–130 g at different locations; King 2005) rat to be introduced. Therefore, in this comparatively brief research project we are looking for rapid responses from the relatively resilient fauna that has already survived hundreds of years in the presence of mammalian predators (Gibbs 2010). Second, the impacts of mice (especially on vertebrates such as lizards and birds) at Maungatautari will likely be small compared with the total impacts of the other larger mammalian predators and browsers that have been removed successfully, such as stoats, ship rats, brushtail possums, hedgehogs (*Erinaceus europaeus*), cats, red deer (*Cervus elaphus*), and pigs (*Sus scrofa*; King 2005). Third, if mice can also be eradicated, and the original bird and lizard fauna can be restored by population growth and translocation, then there may be even greater predation pressure on invertebrates from native predators (Sinclair et al. 2005; Watts et al. 2014). The objective of restoration in sanctuaries is

primarily to restore original ecological interactions and processes (Lee et al. 2005) as far as possible, and not to increase the abundances of all taxa.

6.5 Study design

Our experimental design is different from that recommended by Jones and Toft (2006), who suggested that researchers use at least 20 enclosure plots measuring 10 × 10 m and spaced at 60–100 m for 6 years to determine how mice impact on invertebrate biodiversity. The high number of replicates was chosen to overcome the considerable spatial and temporal variation in invertebrate populations. Our large (17–24 ha) study sites from which we take multiple samples offer another approach to dealing with small-scale spatial variation in invertebrate populations. These large sites are also suitable for research on mouse density, use of trees, and behaviour at bird nests, for which we require devices to be spaced at appropriate distances relative to the width of a mouse home range.

The key disadvantage of having few large study blocks is a lack of replication, which limits inference about the cause of differences between the blocks, and the relevance of this research to other sites. Our design is improved by the implementation of different treatments over time in the same blocks. With this design, our results provide strong support for conclusions that invertebrate abundance and body size are inversely related to mouse abundance, irrespective of any potential block effects.

In this study we were also limited by having just one live-trapping grid for mice in each study block, and the resultant disparity between the small area (1 ha) of these grids in which mouse density can be calculated, compared with the c. 20× larger areas in which tracking tunnels were distributed. This design decision was driven by the high cost of live-trapping on multiple small grids per block.

6.6 Other aspects of mice alone in sanctuaries

We suggest there may be three additional negative outcomes of having abundant mice alone in sanctuaries, which are not direct biodiversity impacts. First, mice may interfere with important monitoring devices set for other species, in particular rats and mustelids, by stealing baits and obscuring footprints in tracking tunnels. Second, house mice are inveterate burrowers (Schmid-Holmes et al. 2001; Avenant & Smith 2003), and may provide conduits out of the sanctuary into adjacent mouse-free exclosures, or their burrows may enable weasels or other predator species back into the sanctuary. Finally, visitors and volunteers may be displeased to see mice when they visit a sanctuary that they think of as ‘pest-free’, especially in the daytime.

We can identify only one possible positive outcome of having mice alone in a sanctuary. Having mice present as prey may in the short term distract any larger predators that invade the sanctuary, especially stoats and weasels, from feeding on threatened fauna such as birds. When stoats invaded Orokonui Ecosanctuary (near Dunedin), where mice were absent, it is likely that they killed most of the South Island saddlebacks present (Elton Smith, Orokonui Ecosanctuary, 2016, pers. comm.). However, a stoat invading Tawharanui Open

Sanctuary near Auckland, where mice were present, was not known to have killed any native birds. In this circumstance the mice were exploited as vectors for anticoagulant poisoning to target the stoat (Matt Maitland, Tawharanui Open Sanctuary, pers. comm.). However, it is also possible that mouse presence in sanctuaries attracts mustelids in the first place, and in the longer term, abundant mice will support mustelid population growth following reinvasion.

7 Conclusions

This research adds to the complex body of knowledge that regional and district councils, wildlife sanctuary trusts, and the Department of Conservation use to manage sanctuaries like Maungatautari. The research suggests that mice limit populations of several indigenous invertebrate groups. However, in our view mice alone are definitely preferable to having all the other pest mammals back at Maungatautari. We hope that mouse control tools will steadily improve so that in the future mice can be eradicated from large rugged forest reserves such as Maungatautari.

8 Recommendations

- If the only mammal present, mice are likely to have significant impacts on threatened invertebrates in their prey size range, such as giant weta, and on litter invertebrate biomass. Mouse control may be worthwhile at any site to improve or maintain these values.
- If high-value conservation taxa, such as kakapo, are translocated into Maungatautari or other fenced sanctuaries where mice are the only mammal present, we recommend that mice should be regarded as potential chick predators until proven otherwise.
- Further research is required to study the likely impacts of mice on nesting birds, lizards, and frogs.

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