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RESEARCH

Population responses of common lizards inside a predator-free dryland sanctuary

Samantha Turner[1,2] and Grant Norbury[3]*

[1]Wildlife Management Programme, University of Otago, PO Box 56, Dunedin, New Zealand
[2]Department of Conservation, 15 Wairepo Road, Twizel 7901, New Zealand
[3]Manaaki Whenua – Landcare Research, PO Box 176, Alexandra 9340, New Zealand
*Author for correspondence (Email: norburyg@landcareresearch.co.nz)

Abstract: Predator-free sanctuaries can assist the conservation of multiple endemic species, but quantitative evidence of these benefits is often lacking, especially for herpetofauna. We measured population responses of three common lizard species (schist geckos, *Woodworthia* 'Central Otago'; McCann's skinks, *Oligosoma maccanni*; and southern grass skinks, *O.* aff. *polychroma* Clade 5) 1 year before and 5 years after mammalian predators were removed inside a mammal-proof fence in a dry grass/shrubland habitat with abundant schist rock in Central Otago, New Zealand. Gecko counts in 25 artificial retreats (ARs) along replicated, 250-m long transects inside the fence increased from 5.2 to 17.7 per transect, compared with 3.3 to 5.4 per transect outside the fence where predators were present. This indicates a doubling of counts inside the fence when changes outside the fence are accounted for. The number of skinks seen along these transects increased from 1.3 to 2.3 per transect without predators, compared with 0.6 to 0.4 with predators. Lizard tracking tunnel indices (recorded only after predators were removed) were significantly higher without predators (62% geckos, 42% skinks) than where predators were present (12% geckos, 2% skinks). Tail loss in geckos was significantly higher where predators were present (24%) than where they were absent (1%). These data suggest that common lizard species are being suppressed to a significant degree by mammalian predators and demonstrate the potential for their recovery where predators are removed.

Keywords: conservation, ecosanctuary, gecko, lizard, predation, predator removal, skink

Introduction

Ninety-six percent of New Zealand's 124 endemic lizard species are either threatened, at risk, or data deficient according to the New Zealand Threat Classification System (Hitchmough et al. 2021). They are globally unique in that they are long-lived, mostly viviparous, and occupy a wide range of habitats from alpine to coastal areas (Lettink et al. 2010; Hare et al. 2016). Lizards also provide important ecosystem services such as pollination and seed dispersal of at least 23 native plant species (Wotton et al. 2016). The key threats to lizard conservation are habitat modification and introduced mammals with mammalian predation considered to be the principal threat (Hare et al. 2016; Nelson et al. 2016). Mammals exert both direct and indirect effects through predation and resource competition for food or refugia (Norbury 2001; Towns et al. 2016).

Predator suppression at landscape scales, and predator removal on offshore islands have widespread positive outcomes for many native species (e.g. Reardon et al. 2012; Hare et al. 2016; Gollin et al. 2021). Predator-free mainland sanctuaries potentially benefit multiple species, but quantitative data for lizard responses is lacking (Nelson et al. 2016). Fenced sanctuaries are expensive, so it is important to understand their benefits to justify their establishment (Scofield et al. 2011; Nelson et al. 2016).

Assessing the responses of relatively common species to management is important for wider conservation. There is an unstated assumption that relatively common lizard species are more resilient to predation than those that are less abundant or threatened. This is true to a degree, given their relatively high abundance and widespread distribution, but whether predators suppress their numbers, and to what extent, is less clear. This study aimed to evaluate the population response of three common lizard species to predator removal in a mainland sanctuary. It was hypothesised that, because they are relatively common, there would be little or no numerical response to predator removal. We used a BACI (before-after-control-impact) design to compare lizard population indices inside and outside the predator-free sanctuary before and after it was constructed and the predators removed. We also tested the hypotheses that predation increases tail loss in lizards and alters the age structure of the population. Age structure can potentially be altered either by greater predation of young or older lizards or by non-lethal effects on reproducing adults.

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Methods

Study site and species

The study took place in the Aldinga Conservation Area (45°17′S, 169°17′E), near Alexandra, New Zealand. The Mokomoko Dryland Sanctuary (14 ha), a predator-proof fenced area, was constructed inside this reserve in March 2015 by the Central Otago Ecological Trust (www.mokomokosanctuary.com). A major gully runs through the sanctuary, surrounded by schist rock outcrops. Vegetation includes a mixture of native (e.g. matagouri *Discaria toumatou*, mingimingi *Coprosma propinqua*, small-leaved pōhuehue *Muehlenbeckia complexa*, and *Olearia bullata*) and exotic plants (e.g. introduced thyme *Thymus vulgaris*, sweet briar *Rosa rubigonosa*, and everlasting pea *Lathyrus latifolius*). All introduced mammalian predators (ferrets *Mustela furo*, stoats *Mustela erminea*, cats *Felis catus*, hedgehogs *Erinaceus europaeus*, mice *Mus musculus*) and herbivores (rabbits *Oryctolagus cuniculus*, hares *Lepus europaeus*) were eradicated by kill trapping, brodifacoum poisoning using bait stations, or shooting. Rats (*Rattus* spp.) were not present. Mice have breached the fence on three occasions but were eradicated on each occasion approximately one month after first detection. Ongoing deployment of sentinel traps has confirmed the absence of mustelids, cats, rats, and hedgehogs. Outside the fence, predators were not controlled, rabbits and hares were suppressed by shooting and intermittent livestock grazing occurred under a special grazing permit.

Common lizard species at the study site were schist gecko *Woodworthia* 'Central Otago', southern grass skink *Oligosoma* aff. *polychroma* Clade 5, and McCann's skink *O. maccanni*. McCann's skinks are not threatened, but the schist gecko and southern grass skink are both classified as At Risk – Declining (Hitchmough et al. 2021).

Study design

Lizards were monitored along ten 250-m long transects spaced approximately 30 m apart. Five of the transects were located inside the sanctuary and five outside. Along each transect lizard monitoring devices (see below) were placed at 25 points spaced about 10 m apart. Each transect intersected the major gully at right angles encompassing a variety of habitats including schist rock, shrubland, and grassland. Surveys were carried out between 8 am and mid-afternoon under optimal weather conditions for skink detectability (i.e. mostly clear skies between 12 °C and 22 °C) (Hoare et al. 2009).

Artificial retreats

At each monitoring point an artificial retreat (AR) was deployed consisting of a stack of three 38 × 28 × 4 cm pieces of brown, corrugated Onduline roofing sheets. These Onduline sheets are made from organic fibres saturated with bitumen (www.onduline.com). *Woodworthia* geckos are especially attracted to ARs because of their structural and thermal properties (Lettink & Cree 2007; Thierry et al. 2009). The top and middle layers were separated with spacers to allow enough room for lizards to fit into the gaps. ARs were placed in sunny spots, with surrounding vegetation cover or rocks if available. ARs exposed to high wind were weighed down with a rock and left undisturbed for 6 weeks to allow time for lizard colonisation (Lettink & Cree 2007).

Three lizard monitoring sessions took place, both before (September–November 2014) and after (December 2019–March 2020) the predator exclusion fence was constructed and predators removed. The effects of sampling in different seasons were accounted for by the BACI experimental design. During each monitoring session, ARs were sampled once over a 2–3 consecutive day period. Each AR layer was carefully lifted, and the number of geckos present was recorded. Gecko counts were summed for the 25 ARs along each transect and averaged over the three sessions for the before and after periods. The duration of surveys was similar each day and spread evenly across the inside and outside transects so that similar weather conditions prevailed across the two predator treatments on any given day.

To test the hypothesis that predation increases tail loss (Hare & Miller 2010; Knox 2014) and alters the age structure of the population, we scored tail loss (including regenerating tail tips) and the number of young (< 40 mm snout to vent length) for geckos during the after period. This involved capturing geckos by hand and marking them on the back using a silver, xylene-free pen and releasing them, followed by a second sampling 7–8 days later to count the number of marked and unmarked geckos. Tail loss and recruitment of young were expressed as a proportion of the population based on the number of geckos in the first sampling with tail loss or that were young (all of which were marked), plus the number of unmarked geckos in the second sampling with the same characteristics, divided by the sum of total geckos caught in the first samplingand the number of unmarked geckos in the second sampling.

Because skinks rarely occupied the ARs in this study, we recorded the number of skinks seen incidentally while looking down at the ground as we walked slowly along the transects. Skinks were mostly seen amongst the grass and occasionally basking on top of ARs.

Tracking tunnels

A second lizard abundance index was recorded (only during the after period) using tracking tunnels (500 × 100 × 100 mm; Black Trakka™ tunnels from Gotcha Traps Ltf) with an ink card placed inside. Two hundred tunnels were placed 10 m apart along eight of the ten transects in between the AR points, half inside the fence and half outside. Tunnels were pinned down with metal pegs to reduce displacement by livestock or brushtail possums (*Trichosurus vulpecula*) outside the fence. Ink cards were collected one week later and the number of cards with lizard prints recorded. Skink prints are narrow with pointed toes, whereas gecko prints are broad with padded toes (Jarvie & Monks 2014). The two skink species could not be separated based on prints so were grouped together. The same observer scored all prints. Tunnels were baited with peanut butter to check for mouse prints (Watts et al. 2011).

Vegetation cover

Because the fence also excluded herbivores, vegetation cover was measured inside and outside the sanctuary to check there were no significant differences that could have confounded lizard abundance. A point quadrat survey was conducted in March 2020 using a 97 cm diameter wheel-point apparatus consisting of metal spikes, with one spike coloured (Tidmarsh & Havenga 1955). The apparatus was wheeled alongside each lizard transect and as the coloured spike approached the ground its contact with herbaceous vegetation or bare ground was recorded. Vegetation cover was calculated by dividing the number of vegetation contacts by the total number of contacts (points that landed on rock were excluded from the denominator because rock does not support vegetation; the proportion of rock cover was similar inside and outside the fence).

Statistical analysis

To check whether air temperature needed to be included in the BACI model, we fitted two generalised linear mixed models (GLMMs) in R (R Core Team 2020). The first GLMM with temperature alone as a predictor of gecko counts and the second with temperature and predator treatment included. Mean hourly temperatures were retrieved from the National Climate Database (cliflo.niwa.co.nz/) from the Alexandra Compact Weather Station (agent number 36592). Only gecko counts from January and February 2020 were included in this analysis because the December 2019 data lacked accurate start and finish times to match with the climate data.

To evaluate the effect of predation on gecko counts, we analysed the interaction between period and predator treatment (Underwood 1993; CJ Schwarz, pers. comm.) using a GLMM. We used a Poisson regression model for counts and a log link function using the *glmer* function from the lme4 package (Bates et al. 2015). The response variable was the mean number of geckos counted per transect. The fixed variables were predator treatment (control-impact), period (before-after), and their interaction term. An observation-level variable (monitoring session nested in transect) was added as a random effect to account for repeated measures of the transects and to compensate for overdispersion (Harrison 2015). Model fit was checked by calculating marginal (R^2_m) and conditional (R^2_c) pseudo-R^2 values (Nakagawa & Schielzeth 2013). Marginal pseudo-R^2 values describe how much of the variation in the data is explained by the fixed variables, while R^2_c describes how much variation is explained by both the random and fixed variables. The statistical significance of the interaction term (period*predator treatment) was assessed using a likelihood ratio test.

Tunnel tracking rates were analysed using a GLMM where the response (proportion of cards tracked) was assumed to be binomially distributed and a logit link function was used. Predator treatment was included as a fixed predictor variable and transect ID as a random variable; the effect of predator treatment was assessed using a likelihood ratio test.

Chi-squared 2 × 2 contingency tables were used to compare the proportion of tail loss and population age structure using the proportion of young geckos per adult in the presence and absence of predators using data from the after fence construction period. Standard errors were based on mean proportions per transect, averaged across months.

Low sightings of skinks precluded statistical analysis, so only sighting frequencies are reported.

Results

Temperature did not have a significant effect on gecko counts in the ARs (Chi sq = 0.444, df = 1, $P = 0.505$). The temperature-only model had a low R^2_m (0.002), explaining very little variance. The model with both temperature and predator treatment also had no significant effects of temperature on counts (Chi sq = 0.633, df = 1, $P = 0.426$), but predator treatment alone did have a significant effect (Chi sq = 20.396, df = 1, $P < 0.001$). Therefore, temperature was not included in the final model.

The mean daily count of geckos per transect increased between 2014 and 2019/20 from 5.2 (95% CI 2.5–7.0) to 17.7 (CI 7.2–36.9) where predators were removed compared with 3.3 (95% CI 1.6–4.5) to 5.4 (CI 2.1–11.5) where predators were present (Fig. 1) indicating a doubling of counts inside the fence when changes outside the fence are accounted for. The period*predator treatment interaction term was highly significant (likelihood ratio test: $X^2 = 11.394$, df = 1, $P < 0.001$), indicating a good model fit with most of the variance explained by the predator treatment and period alone ($R^2_m = 0.488$), although random effects also helped explained some of the variation in counts ($R^2_c = 0.879$). The mean number of skinks seen along each transect increased from 1.3 to 2.3 without predators, compared with a decrease of 0.6 to 0.4 with predators.

Gecko tracking rates were significantly higher where predators were removed (63.0%, 95% CI 50.0–75.2) than where they were present (12.1%, 95% CI 6.7–24.0) (Chi sq = 26.491, df = 1, $P < 0.001$; effect size coefficient of predator presence = −2.449, SE 0.476) (Fig. 2). Similarly, skink tracking rates were significantly higher where predators were removed

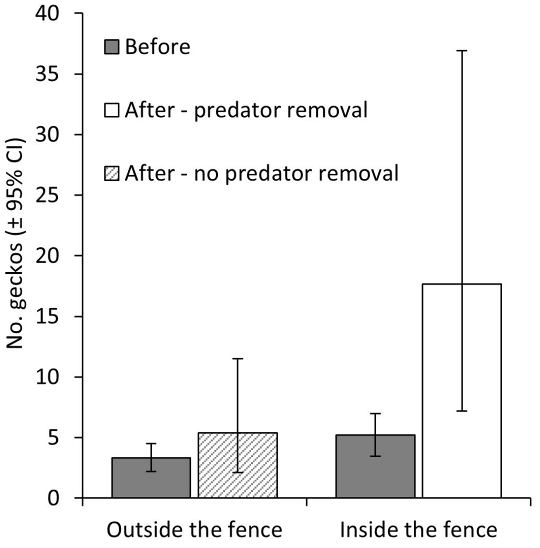

Figure 1. Average counts (± 95% CIs, *n* = 5) of schist geckos per transect (25 ARs) inside and outside the fence before (2014) and after (2019) predators were removed inside the fence.

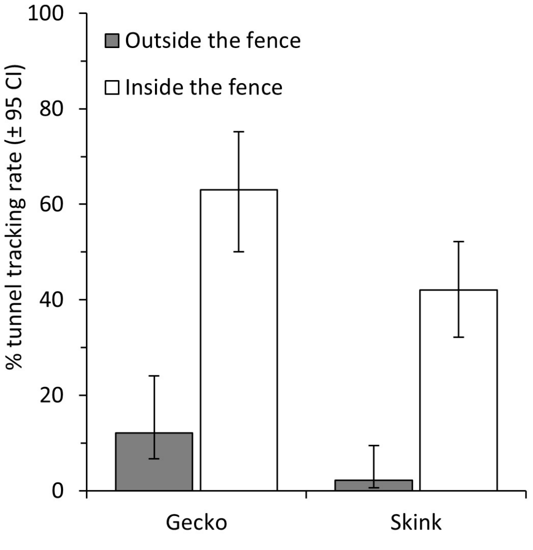

Figure 2. Tunnel tracking rates (± 95% CIs, *n* = 4) for schist geckos and skinks (McCann's and southern grass) in the presence (outside fence) and absence (inside fence) of predators.

(42.0%, 95% CI 32.1–52.5) than where they were present (2.2%, 95% CI 0.6–9.4) (Chi sq = 19.885, df = 1, $P < 0.001$; effect size coefficient of predator presence = −3.363, 0.754 SE) (Fig. 2). Predator removal explained most of the variation in tracking rates ($R^2_m = 0.321$ and $R^2_c = 0.324$). Mouse prints were present in almost half (49.3%) of the tunnels outside the fence, but no prints were detected inside the fence.

The proportion of tail loss in geckos was significantly higher where predators were present (23.8%) than where they were absent (1.4%) ($\chi^2_{(1,\, n\, =\, 487)} = 49.311$, $P < 0.001$). Age structure of the population did not differ between predator treatments ($\chi^2_{(1,\, n\, =\, 487)} = 0.105$, $P = 0.745$), with 13.9% young in the absence of predators, compared with 10.7% young with predators. Vegetation cover was similar inside and outside the fence (90.9% and 87.3%, respectively).

Discussion

Lizard abundance indices derived from artificial retreats, tracking tunnels, and sightings revealed a strong positive response of schist geckos, McCann's skinks, and southern grass skinks to predator removal inside the sanctuary providing empirical evidence of the benefits of predator removal for lizard conservation. The artificial retreat data for geckos are arguably the most robust because they were based on a BACI design and counts within retreats. Given that the skinks and geckos studied here sexually mature at approximately 3–4 years of age, an exponential population increase may be expected in the future with second generation recruitment.

The combination of lower tail loss in geckos and relatively high gecko abundance where predators were removed suggests that predation, not social interactions between conspecifics, was the primary driver of tail loss in this species (Hare & Miller 2010). This same conclusion was inferred for jewelled geckos (*Naultinus gemmeus*) on Otago Peninsula, where sites with higher predator tracking tunnel rates had greater tail loss in adult geckos (Knox 2014).

The age structure of the gecko population was similar between predator treatments suggesting that predation was not biased towards a particular age group, or recruitment rates were not indirectly affected by predation (via reduced basking or higher intraspecific competition for food).

Ground cover of vegetation inside and outside the fence was similar suggesting this was not a contributor to the observed differences in lizard indices. Although plant species composition was not measured we noted no major differences inside and outside the fence and the proportion of habitat types (gully, shrubs, and grassland) were similar. While exclusion of herbivores may allow fruit bearing shrubs to grow faster (geckos in particular consume fruits), we would not expect significant changes in these shrubs in the short timeframe of this study to the extent that it would drive the observed differences in lizard populations.

Similar increases in lizard abundance in a predator-free fenced sanctuary have been demonstrated for ornate skinks (*O. ornatum*) in the Zealandia Wildlife Sanctuary (Nelson et al. 2016). Romijn (2013) showed that during a period of mouse irruption, skink population growth was slower and male skinks were smaller. Given sufficient intensity and scale, suppression of predator populations without fencing can also increase lizard populations. At Macraes Flat, a higher rainfall site in the same region as our study, predator suppression over 2100 ha resulted in increased grand (*O. grande*) and Otago skink (*O. otagense*) abundance, as was the case following predator removal inside 9-ha and 18-ha predator-free fences (Reardon et al. 2012). Similarly, predator suppression over a 650-ha area close to Macraes Flat resulted in increased populations of korero gecko (*Woodworthia* 'Otago/ Southland'), McCann's skink (*O. maccanni*), and southern grass skink (*O.* aff. *polychroma* Clade 5) compared with no lizard increases in a paired area without predator suppression (Norbury et al. 2013).

Because we removed all mammalian predator species inside the fence we could not discern species-specific impacts. Several studies have inferred that mice are important predators of small indigenous lizards (Newman 1994; Towns & Elliott 1996; Lettink & Cree 2006; Hoare et al. 2007; Knox et al. 2012), including some of the larger species (> 25 cm) such as Otago skinks (Norbury et al. 2014). The mouse tracking rates recorded outside the fence in our study were quite high and well above the 5% threshold identified for recovery of small lizard populations (Norbury et al. 2022). However, mouse control can be costly and its feasibility on the mainland is currently problematic, particularly over large enough areas to elicit a population response in lizards. Mice are often not controlled in sanctuaries and mouse incursions happen quite frequently (Innes et al. 2012; Watts et al. 2017). In the absence of predators mouse numbers can increase dramatically and have negative impacts on lizard populations (Nelson et al. 2016). Given the risk they pose to small lizards, methods for sustainable reductions in mouse populations on the mainland are required.

Notwithstanding the high cost of predator proof-fences, our study suggests that predator eradication inside such a fence is a useful conservation tool to improve the viability of relatively common lizards. Even supposedly common lizard species may still be declining in number and may become threatened in the future if not carefully managed. Gathering longer-term data on lizard population trajectories without predators would be beneficial for understanding the densities they can reach. Measuring other lizard response variables would be useful, such as body condition, size, and sex. Gravid McCann's skinks, for example, are known to be more vulnerable to cat predation than non-gravid skinks, but further research is needed (Newman 1994; Chapple 2016). Such differences in vulnerability have important implications for the future persistence of lizard populations (Hare et al. 2016). Similarly, more research is needed to understand the relationships between lizard abundance indices and densities, such as those measured in this study. In the case of ARs they provide reasonably robust indices of lizard abundance in optimal weather conditions (Lettink et al. 2011; Wilson et al. 2017), suggesting that the difference in lizard counts in ARs reported here reflected abundance rather than a change in lizard behaviour.

The level of change in lizard abundance indices was higher for tracking tunnel rates compared with gecko counts in ARs and incidental skink sightings. Possible explanations for high tracking tunnel rates inside the fence include freedom from non-lethal predator effects on lizard activity. With predator removal lizards may be less cautious and therefore more mobile. Additionally, single-detection methods (ARs and visual sightings) differ from tracking tunnels where one lizard could be detected in multiple tunnels thereby inflating tracking indices. This was demonstrated by Lettink et al. (2022) who showed that tracking tunnels were approximately twice as likely to detect skinks than pitfall traps which, like ARs, are single-catch devices.

Despite their supposedly higher resilience to predation, we show that populations of common geckos, and perhaps common skinks, are being suppressed to a significant degree by mammalian predators. This demonstrates the potential for their recovery where predators are removed and has important implications for other lizard species whose populations may also be prevented from reaching higher densities.

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Author contributions: ST and GN conceived the project, interpreted the data, and wrote the original draft, along with further writing, editing, and reviewing. ST carried out the field work and analysed the data.

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Data and code availability: ST has access to data and code from this article, which can be shared upon request.

Ethics: Animal manipulations were approved by the Manaaki Whenua – Landcare Research Animal Ethics Committee.

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

References

Bates D, Mächler M, Bolker B, Walker S 2015. Fitting linear mixed-effects models using lme4. Journal of Statistical Software 67: 1–48.

Chapple DG 2016. The future of New Zealand lizards research. In: Chapple DG ed. New Zealand lizards. Springer International Publishing. Pp. 361–375.

Gollin JF, Gorman N, Armstrong DP 2021. Twenty years on: changes in lizard encounter rates following eradication of rats from Kāpiti Island. New Zealand Journal of Ecology 45: 3423.

Hare KM, Miller KA 2010. Frequency of tail loss does not reflect innate predisposition in temperate New Zealand lizards. Naturwissenschaften 97: 197–203.

Hare KM, Chapple DG, Towns DR, van Winkel D 2016. The ecology of New Zealand's lizards. In: Chapple DG ed. New Zealand lizards. Springer International Publishing. Pp. 133–168.

Harrison XA 2015. A comparison of observation-level random effect and beta-binomial models for modelling overdispersion in binomial data in ecology & evolution. PeerJ 3: e1114.

Hitchmough R, Barr B, Knox C, Lettink M, Monks JM, Patterson GB, Reardon JT, van Winkel D, Rolfe J, Michel P 2021. Conservation status of New Zealand reptiles, 2021. New Zealand threat classification series 35. Wellington, Department of Conservation. 23 p.

Hoare JM, Adams LK, Bull LS, Towns DR 2007. Attempting to manage complex predator–prey interactions fails to avert imminent extinction of a threatened New Zealand skink population. Journal of Wildlife Management 71: 1576–1584.

Hoare JM, O'Donnell CFJ, Westbrooke I, Hodapp D, Lettink M 2009. Optimising the sampling of skinks using artificial retreats based on weather conditions and time of day. Applied Herpetology 6: 379–390.

Innes J, Lee WG, Burns B, Campbell-Hunt C, Watts C, Phipps H, Stephens T 2012. Role of predator-proof fences in restoring New Zealand's biodiversity: a response to Scofield et al. (2011). New Zealand Journal of Ecology 36: 232–238.

Jarvie S, Monks JM 2014. Step on it: can footprints from tracking tunnels be used to identify lizard species? New Zealand Journal of Zoology 41: 210–217.

Knox C 2014. Tail loss as an indicator of predation pressure on *Naultinus* populations. Biogecko 2: 29–30.

Knox CD, Cree A, Seddon PJ 2012. Direct and indirect effects of grazing by introduced mammals on a native, arboreal gecko (*Naultinus gemmeus*). Journal of Herpetology 46: 145–152.

Lettink M, Cree A 2006. Predation, by the feral house mouse (*Mus musculus*), of McCann's skinks (*Oligosoma maccanni*) constrained in pitfall traps. Herpetofauna 36: 61–62.

Lettink M, Cree A 2007. Relative use of three types of artificial retreats by terrestrial lizards in grazed coastal shrubland, New Zealand. Applied Herpetology 4: 227–243.

Lettink M, Norbury G, Cree A, Seddon PJ, Duncan RP, Schwarz CJ 2010. Removal of introduced predators, but not artificial refuge supplementation, increases skink survival in coastal duneland. Biological Conservation 143: 72–77.

Lettink M, O'Donnell CF, Hoare JM 2011. Accuracy and precision of skink counts from artificial retreats. New Zealand Journal of Ecology 35: 236–246.

Lettink M, Young J, Monks JM 2022. Comparison of footprint tracking and pitfall trapping for detecting skinks. New Zealand Journal of Ecology 46: 3478.

Nakagawa S, Schielzeth H 2013. A general and simple method for obtaining R^2 from generalized linear mixed-effects models. Methods in Ecology and Evolution 4: 133–142.

Nelson NJ, Romijn RL, Dumont T, Reardon JT, Monks JM, Hitchmough RA, Empson R, Briskie JV 2016. Lizard conservation in mainland sanctuaries. In: Chapple DG ed. New Zealand lizards. Springer International Publishing. Pp. 321–339.

Newman DG 1994. Effects of a mouse, *Mus musculus*, eradication programme and habitat change on lizard populations of Mana Island, New Zealand, with special reference to McGregor's skink, *Cyclodina macgregori*. New Zealand Journal of Zoology 21: 443–456.

Norbury G 2001. Conserving dryland lizards by reducing predator-mediated apparent competition and direct competition with introduced rabbits. Journal of Applied Ecology 38: 1350–1361.

Norbury G, Byrom A, Pech R, Smith J, Clarke D, Anderson D, Forrester G 2013. Invasive mammals and habitat modification interact to generate unforeseen outcomes for indigenous fauna. Ecological Applications 23: 1707–1721.

Norbury G, van den Munckhof M, Neitzel S, Hutcheon AD, Reardon JT, Ludwig K 2014. Impacts of invasive house mice on post-release survival of translocated lizards. New Zealand Journal of Ecology 38: 322–327.

Norbury G, Wilson DJ, Clarke D, Hayman E, Smith J, Howard S 2022. Density-impact functions for invasive house mice (*Mus musculus*) effects on indigenous lizards and invertebrates. Biological Invasions 25: 801–815.

R Core Team 2020. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Reardon JT, Whitmore N, Holmes KM, Judd LM, Hutcheon AD, Norbury G, MacKenzie DI 2012. Predator control allows critically endangered lizards to recover on mainland New Zealand. New Zealand Journal of Ecology 36: 141–150.

Romijn RL 2013. Can skinks recover in the presence of mice? Unpublished BSc Honours thesis, Victoria University of Wellington, Wellington, New Zealand.

Scofield RP, Cullen R, Wang M 2011. Are predator-proof fences the answer to New Zealand's terrestrial faunal biodiversity crisis? New Zealand Journal of Ecology 35: 312–317.

Thierry A, Lettink M, Besson A, Cree A 2009. Thermal properties of artificial refuges and their implications for retreat-site selection in lizards. Applied Herpetology 6: 307–326.

Tidmarsh CEM, Havenga CM 1955. The wheel-point method of survey and measurement of semi-open grassland and Karoo vegetation in South Africa. Botanical Survey of Southern Africa Memoir 29. Government Printer, Pretoria. 53 p.

Towns D, Elliott G 1996. Effects of habitat structure on distribution and abundance of lizards at Pukerua Bay, Wellington, New Zealand. New Zealand Journal of Ecology 20: 191–206.

Towns DR, Hitchmough RA, Perrott J 2016. Conservation of New Zealand lizards: a fauna not forgotten but undervalued? In: Chapple DG ed. New Zealand lizards. Springer International Publishing. Pp. 293–320.

Underwood AJ 1993. The mechanics of spatially replicated sampling programmes to detect environmental impacts in a variable world. Austral Ecology 18: 99–116.

Watts CH, Armstrong DP, Innes J, Thornburrow D 2011. Dramatic increases in weta (Orthoptera) following mammal eradication on Maungatautari – evidence from pitfalls and tracking tunnels. New Zealand Journal of Ecology 35: 261–272.

Watts C, Innes J, Wilson D, Fitzgerald N, Bartlam S, Thornburrow D, Smale M, Barker G 2017. Impacts of mice alone on biodiversity: final report of a Waikato field trial. Hamilton, Landcare Research. 43 p.

Wilson DJ, Mulvey RL, Clarke DA, Reardon JT 2017. Assessing and comparing population densities and indices of skinks under three predator management regimes. New Zealand Journal of Ecology 41: 84–97.

Wotton DM, Drake DR, Powlesland RG, Ladley JJ 2016. The role of lizards as seed dispersers in New Zealand. Journal of The Royal Society of New Zealand 46: 40–65.

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