

## Impacts of invasive house mice on post-release survival of translocated lizards

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Published online: 9 April 2014

**Abstract:** Invasive house mice (*Mus musculus*) have detrimental effects on biodiversity, but their impacts can be difficult to detect and are often unquantified. We measured their effects on survival of a translocated population of an endangered lizard in New Zealand. Twelve captive-reared Otago skinks (*Oligosoma otagense*) were translocated to a 0.3-ha area of grassland/shrubland cleared of invasive mammals and surrounded by a mammal-resistant fence. Sixteen more skinks were released 2 years later but this was followed by an incursion of mice for c. 160 days. Peak mouse density was at least 63 per hectare, and they were seen attacking adult skinks (> 25 cm in length), which is previously undocumented for this lizard species. Using photo/re-sight methods and Program MARK, we estimated skink survival ( $\phi$ ) and detectability ( $p$ ) in the presence of mice (second cohort:  $\phi = 0.15$  per annum, 95% Confidence Interval (CI) 0.01 – 0.48;  $p = 0.28, 0.20 - 0.38$ ) and in their absence (first cohort:  $\phi = 0.44$  p.a., 95% CI 0.03 – 0.82;  $p = 0.29, 0.22 - 0.39$ ). Survival of skinks from the first cohort during the mouse incursion was unaffected, presumably because they were already established and had access to familiar or more optimal refugia. Their survival over the entire 3 years of monitoring (0.83, 95% CI 0.60 – 0.93) compared favourably with published estimates for viable populations in the wild, protected from all invasive mammals. This suggests it may be feasible to re-establish captive-reared lizards in the wild, but mice should be considered a limiting factor, at least during the initial translocation phase.

**Keywords:** Central Otago; invasive species; *Oligosoma otagense*; predation; reintroduction; Otago skink

### Introduction

Invasive house mice (*Mus musculus*) are distributed worldwide and are increasingly recognised for their unwanted impacts on indigenous island species and ecosystems (e.g. Wilson et al. 2007; St Clair 2011; Wanless et al. 2012). However, their impacts can be difficult to detect and are often unquantified (St Clair 2011). Eradicating mice is expensive and problematic where reinvasion is likely, so having a clear understanding of the potential ecological gains of mouse control is important. Given their frequent incursions, mice are often the only mammalian pest remaining inside fenced sanctuaries in New Zealand (Innes et al. 2012). A number of New Zealand studies have inferred that mice are harmful predators of small indigenous lizards (Newman 1994; Lettink & Cree 2006; Hoare et al. 2007; Knox et al. 2012). Mice may also compete with lizards for food and shelter, or reduce basking opportunities. No study has measured the effects of mice on vital rates of lizard populations.

We assessed the impacts of mice on survival rates of a translocated population of an endangered skink species, the Otago skink (*Oligosoma otagense*). Once widespread in Central Otago (South Island, New Zealand), these skinks have declined dramatically over the past century and now occupy only 8–10% of their former range (Whitaker & Loh 1995). The species is now classified as 'nationally endangered' (Hitchmough et al. 2013). The only extant populations are present near the boundaries of their former range in the Macraes Flat and Lindis/Hawea districts (Whitaker & Loh 1995). Declines have been

attributed to predation by invasive cats (*Felis catus*), weasels (*Mustela nivalis*), stoats (*M. erminea*) and ferrets (*M. furo*) (Reardon et al. 2012), which are top predators in New Zealand ecosystems. The effects of invasive house mice on Otago skinks, whether by predation and/or competition for food or refugia, are unknown. Our prediction was that skink survival would be lowered by predation and/or competition with mice.

All skinks in this study were at least third-generation captive-reared from an original founder population of 12 individuals taken from eastern Central Otago. The project therefore provided a further opportunity to assess whether survival of the translocated captive-reared population would be lower than published survival rates of wild skinks (in Reardon et al. 2012).

### Materials and methods

#### Study site

In 2009, a community conservation group, the Central Otago Ecological Trust, began a pilot study to test the feasibility of reintroducing captive-reared Otago skinks to an area the species formerly occupied. In August 2009, we enclosed a 0.3-ha area of grassland/shrubland habitat with a 1.9-m-high mammal-resistant fence (Pestproof Fences, Havelock North). This release site was chosen because it contained high cover of indigenous shrubs (e.g. *Coprosma propinqua*, *Meliccytis alpinus*, *Discaria toumatou*, *Muehlenbeckia complexa*) that provided suitable food and refuge for skinks. The site

also contained numerous schist rock outcrops with deep horizontal cracks that skinks use for refuge. The elevation of the site is 340 m (taken from Google Earth imagery), and average annual rainfall (in the town of Alexandra, 7 km away) is 363 mm (New Zealand National Climate Database of the National Institute of Water and Atmospheric Research; see <http://cliflo.niwa.co.nz>). Before releasing the skinks, to ensure their adequate thermal protection from freezing, we measured minimum temperatures in winter 2008 (a reasonably average winter), using temperature probes (DS1922L-F5# Thermochron iButton) placed 1–3 m inside six rock crevices. Minimum temperatures did not fall below 3°C. Given these favourable characteristics, we believed the site could support at least 60–80 skinks.

### Mammal eradication and monitoring

To eradicate any mammals inside the enclosure before translocating the skinks (in November 2009) we used 20 kill-traps (spring-loaded metal-jawed traps: ‘DOC-250’ (Department of Conservation 2014) and ‘Timms’ traps ([www.philproof.co.nz/gen\\_showproduct.php?cat=1](http://www.philproof.co.nz/gen_showproduct.php?cat=1))); baited with fresh rabbit meat over a 3-month period. Twenty footprint-tracking tunnels (using ‘Black Trakka’ cards from Gotcha Traps, Warkworth) baited with peanut butter (Cunningham & Moors 1996) were deployed to detect rodents. To ensure no mammals were present, traps and tunnels remained in place for 6 months before skinks were released. Kill-traps were deactivated when skinks were released to avoid accidental capture of skinks, but tunnels remained active for the duration of the study to monitor continuously for the presence of rodents. Tunnels were checked approximately every month and the ink replenished. No mammals or their scats were detected or observed during regular visits to the site, until January 2012, when mice were seen inside the fence and tracking tunnels were marked with mouse prints (Table 1). Suspected entry points through the fence were blocked 4 months later and the mice eradicated using 20 live-capture Elliott traps (Tasker & Dickman 2002) cleared daily for 5 days, followed by poisoning over 4 weeks using 16 bait stations elevated 30 cm above ground by wooden posts to make them less accessible to skinks. Bait stations contained brodifacoum-impregnated

cereal baits (Talon® Pellets rodenticide). There was no sign of mice inside the fence 4 weeks after control began.

### Skink releases and monitoring

Twelve adult skinks were taken from captivity in the North Island, held in a Department of Conservation quarantine facility in Otago for 8 weeks for disease screening, and released inside the fence in November 2009 (Hare et al. 2012). In December 2011, another 16 adult captive-reared skinks were quarantined and released. Because Otago skinks are diurnal, spend much of their time sun-basking, and have unique body markings, a non-invasive photo re-sight method can be used to identify individuals and estimate survival rates (Reardon et al. 2012). Skinks were monitored every 15 days on average (range, 1–94 days) on 75 occasions. Monitoring was less frequent during winter because skinks were less active then. A monitoring session involved one person (same person on 60% of occasions) walking quietly through the area for 1–2 h photographing the lateral surfaces (snout to foreleg) of skinks. Images were compared by eye with a photographic library of known individuals. Monitoring began 7–10 days after each translocation and occurred only during good conditions for observing skinks (i.e. warm, sunny, little or no wind) from November 2009 to November 2012. The first mouse was seen in January 2012, 10 days after the release of the second skink cohort (during the first follow-up visit to the site).

### Data analysis

Mouse density was estimated as the number removed by trapping until no more were captured, divided by the area enclosed by the fence. This calculation assumed that all of the mice present were captured, and that the population was ‘closed’ during the 5-day trapping period. The estimate does not account for mice that may have been poisoned later, and therefore represents a minimum estimate.

To estimate skink survival rates, we analysed the photo/re-sight data using the Cormack–Jolly–Seber model in Program MARK (version 6.0; White & Burnham 1999). We assumed no emigration (we have no knowledge of adult Otago skinks escaping through mammal-resistant fences), no differences in survival due to gender or sampling session (data were too

**Table 1.** Timeline of key events, showing three periods for which skink monitoring data were analysed.

Time	Event/data analysis
15 Aug. 2009	Fence completed
19 Aug. – 27 Nov. 2009	Predator eradication
28 Nov. 2009	First cohort of 12 skinks released into enclosure
28 Nov. 2009 – 6 May 2010	<i>Period</i> over which photo/re-sight data from first skink cohort (‘group 1’ in Methods) were analysed, before mice entered the enclosure ( $n = 159$ days)
22 Dec. 2011	Second cohort of 16 skinks released into enclosure
1 Jan. 2012	Mice detected inside enclosure
22 Dec. 2011 – 29 May 2012	<i>Period</i> over which photo/re-sight data from second skink cohort (group 2) and skinks remaining from first cohort (group 3) were analysed, after mice entered the enclosure ( $n = 159$ days)
20–25 Apr. 2012	Mice trapped to zero captures
4 May 2012 – 4 Jun. 2012	Mouse poison deployed in elevated bait stations
15 May 2012	No mice detected
5 Dec. 2009 – 20 Nov. 2012	<i>Period</i> over which photo/re-sight data from first cohort were analysed (i.e. the full 3 years of the study)

sparse for gender- and session-specific analyses), and no behavioural response to capture, as there was no capture or handling of skinks. Skinks were only occasionally repelled by the observer, but they invariably re-emerged from a refuge minutes later. Because the first cohort was present for 2 years before mice appeared, we compared null models in MARK with models in which both survival ( $\phi$ ) and capture probability ( $p$ ) parameters varied with each release cohort. The best model was selected on the basis of its Akaike Information Criterion (AICc) value, corrected for bias due to small sample sizes (Burnham & Anderson 2001). Survival was estimated by model-averaging the top models with AICc values that differed by  $<2$  (Burnham & Anderson 2002).

We compared survival and capture probability between three different groups of skinks: 'group 1': the first-release cohort of skinks ( $n = 12$ ), monitored during the 153-day period beginning 7 days after their release, before mice entered the enclosure; 'group 2': the second-release cohort of skinks ( $n = 16$ ), monitored during the 160-day period beginning 10 days after their release, after mice entered the enclosure; and 'group 3': skinks ( $n = 6$ ) remaining from the first cohort that were present when mice entered the enclosure (see Table 1 for timeline). Comparison of (1) and (2) allowed us to assess post-release differences in survival in the presence and absence of mice. Because the intervals between consecutive sampling sessions for (1) and (2) were not identical, we used only photo/re-sight sessions that occurred at about the same time of year (within a week). This allowed us to set mean between-session time intervals in MARK that applied to both groups, with some sessions omitted. A couple of individuals from each cohort disappeared immediately after their release and were never found again, but they were included in this analysis for consistency between the presence and absence of mice. Comparison of (2) and (3) tested whether survival of skinks in the presence of mice was influenced by whether skinks were established (first-release cohort) or not (second-release cohort) at the site. In this case, skinks were monitored at the same time so all data were used over the mouse incursion period. MARK calculated weekly survival rates, which we converted to annual rates.

If high overlap was present between the confidence intervals (CI) of the survival estimates for two cohorts, we applied a form of Monte Carlo uncertainty analysis (Caswell 2001) to estimate the probability that one survival rate was higher than another. We generated beta probability distributions of survival for each cohort on the basis of their means and standard errors of the mean (SE) (assuming they varied independently), drew 5000 random samples from each distribution, and computed the difference for each sample.

Finally, we compared survival of our translocated, captive-reared population with survival of skinks in the wild. The first-release cohort appeared to be unaffected by mice, so we used all data collected from this cohort over the full 3 years of the study to compare with wild populations at Macraes Flat protected from all invasive mammals, including mice (Reardon et al. 2012). Assuming the two individuals from this cohort that disappeared immediately after their release were casualties of the translocation process, we omitted them from this analysis to make a valid comparison with wild skinks in Reardon et al. (2012). Seven juveniles were born during the study but were excluded from analyses because (1) published survival estimates of wild skinks excluded juveniles of the current year, due to low detection probabilities compared with older cohorts, and violation of closed-population assumptions, and (2) early in the

study some new-born skinks may have escaped through small gaps in the fence that were subsequently blocked.

## Results

### Mouse abundance and attacks on lizards

No mice were detected for the first two years of the study. Mice were first detected by direct observation in January 2012, and 57% of tunnels were tracked by mice in February 2012. During April 2012, we observed two instances of adult mice attacking adult skinks (20–25 cm in length). The first attack was on a skink from the second cohort that was basking on rocks; it writhed vigorously and escaped. Several hours later, we observed the same skink basking again. Photographs revealed several bites that had punctured and torn the epidermis on the skink's head. This skink was observed several times over the next four weeks until the mice were eradicated, but it was not seen thereafter. The second attack was seen on the same day, also on a basking skink that was able to escape. Photographs were not obtained on this occasion so the extent of injury or survival of this individual could not be determined. After mice were detected in the enclosure, 19 were killed by traps and an unknown number were poisoned until no more were detected. The minimum estimate of mouse density inside the fence was approximately 63 per hectare.

### Survival of skinks with and without mice

The top model included skink group (groups 1 and 2, with and without mice) as a predictor of survival, but there was poor differentiation between models because most models differed by AICc  $< 2$ , and model weights were  $< 0.5$ , (Burnham & Anderson 2002) (Table 2). Model-averaged survival of the second cohort of skinks in the presence of mice (0.15 per annum, 95% CI 0.01–0.48) was lower than survival of the first cohort in the absence of mice (0.44 p.a., 95% CI 0.03–0.82) but there was considerable overlap in confidence intervals. By incorporating uncertainty in these survival estimates using a form of Monte Carlo uncertainty analysis (Caswell 2001), we estimated an 85% probability that survival was lower when mice were present. Capture probability of skinks was similar regardless of the presence of mice (Table 2).

### Survival of established and newly-released skinks with mice

The two top models in this analysis included skink group (groups 2 and 3) as a predictor of survival (Table 3). The weight of the top model was quite low (0.67) so we again used model-averaging to estimate survival. Survival of the six established skinks remaining from the first cohort was apparently unaffected by mice as their mean survival rate (0.91 p.a., 95% CI 0.46–1.00) during the mouse incursion far exceeded that of the second cohort (0.17 p.a., 95% CI 0.02–0.45) released shortly after the mouse incursion. Capture probabilities were similar for these groups (Table 3).

### Overall survival of skinks unaffected by mice

To compare survival of our captive-reared skinks with published estimates for wild skinks, we analysed the full 3-year dataset for the first cohort, for which no effect of mice had been detected. Survival of this cohort (excluding two possible translocation casualties to make a valid comparison with wild skinks) was high (0.83 p.a., 95% CI 0.60–0.93). Capture probability was 0.34 (95% CI 0.30–0.39).

**Table 2.** Four models based on photo/re-sight data from the first-release cohort of 12 skinks, monitored 2 years prior to the mouse incursion, and the second-release cohort of 16 skinks, monitored from when mice were initially detected, i.e. in both cases beginning 7 or 10 days immediately after release into the enclosure. Survival ( $\phi$ ) and capture probability ( $p$ ) were modelled as a function of skink cohort ( $g$ ). The null model is represented by (.). The highest-ranking models are indicated by the lowest corrected Akaike Information Criterion (AICc) score and the highest model weight. Delta AIC is the difference in the AIC value compared with the top model. ML = model likelihood.  $k$  = number of parameters. Estimates of  $\phi$  (annual rate) and  $p$  (between monitoring sessions) were derived from the averaged model. LCL = lower 95% confidence limit. UCL = upper 95% confidence limit.

Model	AICc	Delta AICc	AICc weights	ML	k
Phi(g) p(.)	299.6	0	0.454	1	3
Phi(.) p(.)	301.0	1.374	0.228	0.503	2
Phi(g) p(g)	301.6	1.945	0.171	0.378	4
Phi(.) p(g)	301.9	2.256	0.147	0.324	3

  

Parameter	Estimate	LCL	UCL
Phi (group 1, before mice)	0.4371	0.0341	0.8196
Phi (group 2, with mice)	0.1532	0.0093	0.4770
p (group 1, before mice)	0.2949	0.2177	0.3861
p (group 2, with mice)	0.2767	0.1956	0.3758

**Table 3.** Four models based on photo/re-sight data from the six established skinks surviving from the first cohort that were present when mice were initially detected, and the newly-released second cohort of 16 skinks; monitoring of both groups began when mice were first detected. Other details as for Table 2.

Model	AICc	Delta AICc	AICc weights	ML	k
Phi(g) p(.)	328.6	0	0.672	1	3
Phi(g) p(g)	330.7	2.075	0.238	0.354	4
Phi(.) p(.)	333.3	4.690	0.064	0.096	2
Phi(.) p(g)	335.2	6.569	0.025	0.038	3

  

Parameter	Estimate	LCL	UCL
Phi (group 3, established skinks with mice)	0.9077	0.4642	1.0000
Phi (group 2, newly-released skinks with mice)	0.1677	0.0228	0.4464
p (group 3, established skinks with mice)	0.2561	0.1959	0.3273
p (group 2, newly-released skinks with mice)	0.2505	0.1939	0.3172

## Discussion

### Mouse attacks on skinks

Several mammals are known to prey on Otago skinks (e.g. Whitaker & Loh 1995; Tocher 2006) but our study is the first account of house mice attacking this species. Predation by mice on numerous smaller lizard species, often juveniles, has been reported (Newman 1994; Towns & Elliott 1996; Hoare et al. 2007; Lettink & Cree 2006). The Otago skink is one of New Zealand's largest lizards (up to 30 cm in length), and both attacks we witnessed were on adults. The absence of other mammalian predators inside the fence may have allowed mice to be more aggressive towards skinks and to be active during the daytime, as mice are known to alter their foraging behaviour when top predators are absent (Arthur et al. 2004).

### Mouse effects on skink survival

Our prediction that skink survival would be reduced in the presence of mice was supported. Release of the second skink cohort coincided with the mouse incursion, and there appeared

to be a reduction in their survival compared with the cohort released without mice, but the precision of these estimates was poor due to sparse data; however, Monte Carlo uncertainty analysis further supported our conclusions. Nonetheless, we had only single skink releases to compare so we were careful to ensure similar conditions applied to both releases, such as the same month of release, similar intervals between release and the start of monitoring, and similar climatic conditions. The addition of mouse poison (in elevated bait stations that were less accessible to skinks) was a difference between the releases but poison was present only during the last few weeks of the mouse incursion so was unlikely to have affected skink survival. Also, if poisoning reduced survival of skinks, it should have been apparent for both cohorts. No other factors that might have confounded the results were immediately obvious. Otago skinks are long-lived (into their late 30s in captivity), slow to mature (4.8 years), and normally display high survival rates (0.83 – 0.94 p.a.) when protected from predators (Reardon et al. 2012). The survival rate of the second cohort was unusually low for this species.

Others have inferred predation as the cause of negative relationships between mice and lizards in other New Zealand ecosystems (Newman 1994; Hoare et al. 2007; Knox et al. 2012). Our observation of mice attacking skinks supports this hypothesis, but competition for food or refugia is another, as yet, unquantified process. Competition with the already-established skink population is another possibility but we believe this is unlikely as only six skinks of the original cohort were present when the second cohort was released, which is well below our estimate of the site's carrying capacity of 60–80 skinks. All six skinks remaining from the first cohort survived the mouse incursion. Unlike the second cohort, these skinks had been established for 2 years and would have had adequate time to explore the environment and select optimal refugia. The 'inexperience' of the second cohort may have made them more vulnerable to mice. These putative behavioural differences between skink cohorts in response to mice were not evident, however, in their capture probabilities.

### Survival of captive-reared skinks compared with wild populations

Our prediction that survival of the captive-reared population would be lower than that of wild populations was not supported. Survival of the skinks that were unaffected by mice (the first cohort; 0.83 p.a.) was similar to that of Otago skinks protected in the wild from mammals at Macraes Flat (0.83 p.a., 95% CI 0.62 – 0.92, in 2006–2007; and 0.94 p.a., 0.50 – 1.00, in 2007–2008; Reardon et al. 2012). These wild populations were below the carrying capacity of the habitat and displayed high rates of population growth. The similarly high survival rate of our translocated population therefore suggests that population growth would be expected, given more time. Indeed, recruitment of seven offspring was recorded during our study.

While translocations of bats, birds, reptiles and invertebrates have generally been successful in New Zealand (Sherley et al. 2010), herpetofaunal translocations elsewhere often have poor success compared with other vertebrate groups (Dodd & Seigel 1991; although see Germano & Bishop 2009). Success is often a function of the quality of founders, which has a strong influence on post-release survival (Snyder et al. 1996; Connolly & Cree 2008; Santos et al. 2009). Translocations often involve captive-reared founders (Fischer & Lindenmayer 2000; Santos et al. 2009). Their high survival rate in our study was unexpected, but insufficient time has elapsed to determine whether it will translate into high population growth and long-term persistence.

### Implications for conservation of lizard fauna

House mice have been shown to negatively affect a number of indigenous island species and ecosystems around the world (e.g. Wilson et al. 2007; St Clair 2011; Wanless et al. 2012), but for indigenous lizards, the evidence of impact is confined largely to New Zealand. Whether this reflects lack of coevolution with mice or simply a consequence of geographic bias in lizard research is unknown. The threat posed by mice in New Zealand raises concerns for a number of lizard species on the mainland that are not protected from invasive predators and currently listed as threatened with extinction (e.g. the Nationally Critical *Oligosoma* aff. *infrapunctatum* 'Chesterfield' and *O.* aff. *longipes* 'Rangitata', and the Nationally Endangered *O. judgei*, *O. pikitanga*, *O. burganae*, *Mokopirirakau* aff. *granulatus* 'Southern Forest', and *Toropuku* aff. *stephensi* 'Coromandel') (Hitchmough et al. 2013).

### Conclusion

There was some evidence that mouse predation reduced survival of Otago skinks, at least for skinks that had just been released into an unfamiliar enclosed environment free of other mammalian predator species. Predation by mice may be an important limitation to the success of lizard translocation programmes, particularly during their initial phase. Efforts to control mice are therefore probably warranted, at least before the release of captive-reared populations. Our study was unreplicated and lacked an experimental control, so we cannot be sure whether the results apply to other ecosystems. However, our findings supplement a growing body of evidence that invasive mice can potentially reduce the probability of translocation success of indigenous biodiversity.

### Acknowledgements

We are grateful to the many volunteers who have assisted with this project, to the Central Otago Ecological Trust trustees (Garry Nixon, Tom Lamb, Selwyn Tomkins, Jan Kelly, Gordon Rayner, Alistair Campbell, Francie Diver, Derek Chinn), to the Department of Conservation (particularly Tim Whittaker, Rob Wardle, Lesley Judd, Karina Sidaway and Nathan Whitmore), and to private donors. We also thank our corporate funders, Central Lakes Trust, Otago Community Trust, Department of Conservation, World Wildlife Fund, Transpower New Zealand, MWH Global, Contact Energy, TrustPower, Landcare Research, and Rotary Club of Alexandra. Jennyffer Cruz and Nathan Whitmore provided advice on modelling re-sight data, and Andrew Gormley conducted the uncertainty analysis. Roger Pech, Andrea Byrom, Deb Wilson, James Smith, Carlos Rouco, Tony Whitaker, Jo Monks, Anna Carter, and Hannah Buckley provided useful feedback on the manuscript.

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Received 20 December 2013; accepted 18 February 2014