

Effects on South Island robins (*Petroica australis*) from pest control using aerially applied 1080 poison

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Abstract: New Zealand robins are thought to be vulnerable to poisoning by sodium fluoroacetate (1080), because individual birds found dead after aerial pest control operations have tested positive for 1080. We investigated the impacts of an aerial 1080 operation (preceded by non-toxic prefeeding) to control brushtail possums (*Trichosurus vulpecula*) on the survival and breeding success of a robin population at Silver Peaks, Dunedin environs, South Island, New Zealand. We monitored the survival of individual marked robins and their nesting success before and after the 1080 application. Robins were also monitored at a non-treatment site at nearby Silverstream. The possum control operation at Silver Peaks reduced rats (*Rattus rattus*), possums and mice (*Mus musculus*) to very low numbers, which remained low for at least 3 months. In contrast, at Silverstream, pest numbers remained high throughout the study. All individually colour-banded robins monitored in the treatment area ($n = 19$) plus five unbanded territorial birds were resighted post-drop, indicating that no known birds died as a direct result of the poison application. Nesting success in the treatment area, measured as daily survival rate, was higher than in the non-treatment area, and did not significantly differ between the breeding seasons before and after the poison drop. Our study showed that the 1080 operation did not adversely affect the robin population at Silver Peaks, but did reduce important nest predators (rats and possums) to low levels. Although there was no evidence of a positive indirect impact on nesting success through the reduction in predators, rat and possum abundances were relatively low at Silver Peaks (cf. Silverstream) prior to the 1080 drop, and vary from year to year. We recommend continued monitoring over several seasons to ascertain whether 1080 operations have any long-term benefits for robins at Silver Peaks.

Keywords: nest survival; non-target mortality; prefeeding; poison baits; *Rattus rattus*; ship rat; sodium fluoroacetate; *Trichosurus vulpecula*

Introduction

Aerial application of sodium fluoroacetate (1080) is used for large-scale control of brushtail possums (*Trichosurus vulpecula*). Possums are a vector for bovine tuberculosis (*Mycobacterium bovis*) (Morris & Pfeiffer 1995; Ryan et al. 2006) and a conservation threat in New Zealand (Brown et al. 1993; Montague 2000). The same technique is also capable of controlling two other introduced predators: ship rats (*Rattus rattus*) and stoats (*Mustela erminea*), the latter species through secondary poisoning (Nugent et al. 2011). Despite being recognised as an efficient large-scale pest control tool (PCE 2011), the aerial application of 1080 remains controversial with the New Zealand public (Hansford 2009; Philp 2009; PCE 2011; Green & Rohan 2012). One of the main concerns is the unintended by-kill of native forest birds (ERMA 2007).

In the past, aerial 1080 operations used carrot baits, but their quality and handling procedures strongly affected non-target mortality of birds (Spurr & Powlesland 1997; Powlesland et al. 1999). The use of industrially produced cereal-based pellets has largely eliminated variability in bait quality (Veltman & Westbrooke 2011). However, Veltman and Westbrooke (2011) also identified a lack of studies of forest birds at risk from aerial pest control operations that follow the current recommended best-practice method of using cereal-based 1080 pellets, preceded by prefeeding with non-toxic bait (but see Greene et al. 2013). Prefeeding increases the amount

of toxic bait eaten by possums (Warburton et al. 2010), and Veltman and Westbrooke (2011) asked whether prefeeding could likewise affect bird foraging behaviour, thus increasing the risk of poisoning birds. Recently, Veltman et al. (2014) proposed six principles to help prioritise research on effects of aerial pest control on bird survival.

New Zealand robins (*Petroica* spp.) have been found dead after 1080 operations, with their carcasses testing positive for 1080, making them a high priority species for study (Spurr & Powlesland 1997; Veltman & Westbrooke 2011). Robins may ingest poison either by consuming toxic baits or by secondary poisoning, through eating arthropods that have fed on the baits (Lloyd & McQueen 2000). To date, no studies have investigated the overall effects of prefed cereal-bait 1080 operations on robins by using individually marked birds. The goal of this study was to determine the impact of prefed aerial 1080 operations on a population of individually banded South Island robins (*Petroica australis*). Our specific objectives were: (1) to monitor changes in rat and possum abundance before and after aerial 1080 application, and at a non-treatment site; (2) to investigate whether individual robins were killed by 1080; and (3) to investigate whether the reduction of introduced predators increased robin nesting success (Innes et al. 2010). We predicted that a reduction of nest predators would lead to increased nesting success following the aerial application of 1080.

Methods

Study areas

The study area included two sites near Dunedin (Fig. 1), which are the only local areas where robins are commonly seen. The Silver Peaks study population (100 ha) is in a production forest plantation owned by City Forests, located in the upper reaches of the Waikouaiti River South Branch catchment. It is dominated by Douglas-fir (*Pseudotsuga menziesii*) and includes smaller patches of Monterey pine (*Pinus radiata*). The non-treatment study population (120 ha) is in the Silverstream catchment, which is dominated by kānuka (*Kunzea ericoides*) native forest. The two sites are 11 km apart (Fig. 1) and separated by the Silver Peaks Range, with no evidence of movement of robins between the two sites (IGJ, unpubl. data).

Banding of robins and monitoring of nesting success started at Silver Peaks in 2007 and at Silverstream in 2008 (IGJ, unpubl. data). For the purposes of this study, we included data from the 2010–2011 and 2011–2012 breeding seasons only, when robin monitoring techniques were consistent between the two study sites and more intense than in previous seasons. Robin territory locations at both study areas were well known during both seasons, and the majority of territorial birds (~85%, $n = 40$) were banded with unique colour bands for individual identification.

Aerial possum and rodent control

Cereal prefeed and 1080 poison pellets were aerially broadcast over approximately 5100 ha, including the Silver Peaks study site, by helicopter using an under-slung bucket. GPS technology was used to apply the bait along parallel flight paths with a

swath width of 120 m. RS5 cereal pellets (Animal Control Products, Whanganui, NZ) with a cinnamon lure were spread at a rate of 1.5 kg per hectare as prefeed on 9 August 2011. The same bait type with a 1080 loading of 0.15% w/w was applied at 2 kg per hectare for the toxin application, on 5 September 2011.

Rodent and possum abundance monitoring

In order to establish the effectiveness of the 1080 poison drop in reducing pests, the relative abundances of possums and rodents (rats *Rattus* spp. and house mice *Mus musculus*) were monitored before and after the poison drop at both sites (before-after-control-impact (BACI) design). Monitoring was done at Silver Peaks 34 days before the poison drop (on 1 August 2011), 11 days afterwards (16 September), and in a 'follow-up' survey, 80 days post-drop, during the middle of the robins' nesting period (24 November). At the non-treatment Silverstream study site the same monitoring regime was done on 23 August, 5 October and 1 December 2011.

Chewtrack cards (chewcards) (Sweetapple & Nugent 2011; made by Connovation, Manukau, NZ) were used for all predator abundance monitoring. Cards (9 × 18 cm) were made of 3-mm-thick white corrugated plastic, with peanut-butter-based Connovation FeraFeed 213 Special Blend paste as bait in the flutes (internal channels of the plastic). Rat, mouse and possum detection was based on tooth impressions left on the baited cards. Rat and possum Chewtrack Card Indices (CCIs) are significantly correlated with other established indices of pest detection, such as tracking tunnel and trap-catch indices (Sweetapple & Nugent 2011). Because chewcards are sensitive to the presence of rodents, they are useful for monitoring low-density rat populations, as are expected after aerial 1080 operations (Sweetapple & Nugent 2011).

At each study site up to 71 chewcards were placed at 50-m intervals along a continuous tracking line connecting known robin territories. At Silverstream, the tracking line followed a network of public walking trails, and cards were placed 1–5 m off the track. Cards were nailed to tree trunks at approximately 30 cm above the ground. The tree was marked with flagging tape and its location coordinates were recorded using a GPS unit; the same locations were used in all monitoring sessions. Cards were retrieved after 10 (± 1) days and analysed for bite marks. Tooth impressions were examined with hand lenses and compared with colour reference photographs provided by P. Sweetapple (Landcare Research).

The percentage of cards bitten by any species was assumed to be positively related to the abundance of that species (Sweetapple & Nugent 2011). CCIs were not corrected for the variable 10 (± 1) days that cards were deployed, because in previous studies almost all detection occurred during the first 7 days (Nugent et al. 2011). Pre- and post-drop rodent and possum CCIs were compared within and between sites to identify changes in mammal abundance. Point estimates and 95% confidence intervals (95% CIs) for CCIs were calculated with the adjusted Wald method, which is most suitable for dealing with proportions from small sample sizes (Sauro & Lewis 2005; Lewis & Sauro 2006). This method gave point estimates slightly greater than 0 when actual tracking rates were 0. Statistically significant changes in mammal abundance were inferred when 95% CIs did not overlap. We chose this simple method of non-overlapping adjusted Wald CIs instead of statistical tests because we were interested primarily in detecting expected large changes in pest abundance after the poison drop.

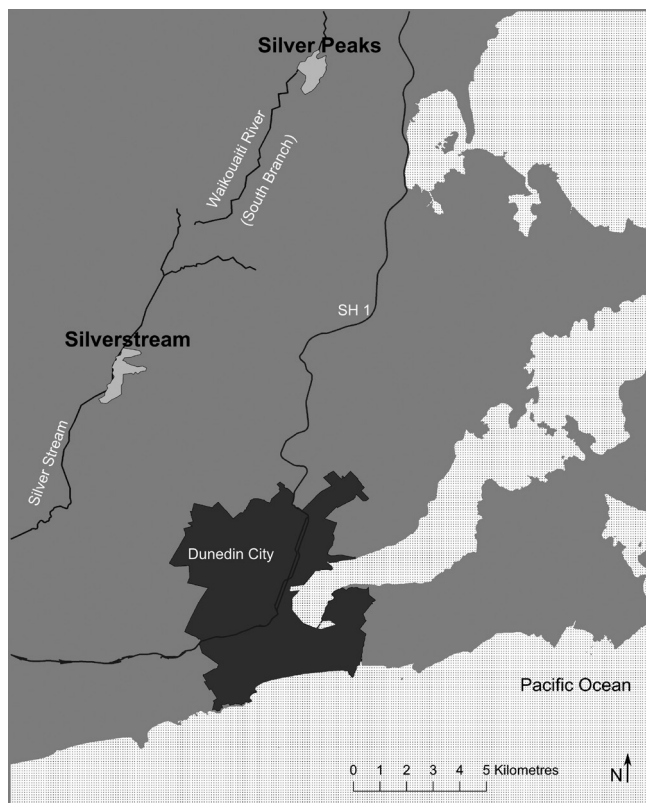


Figure 1. Silver Peaks (45°42' S, 170°31' E) and Silverstream (45°48' S, 170°25' E) study areas near Dunedin, South Island, New Zealand.

Monitoring of adult robin survival

Survival rates for 19 banded territorial adult robins (and five additional unbanded adults on five different territories) at Silver Peaks, and 15 banded (plus one unbanded) territorial robins at Silverstream were estimated based on a mark–recapture survey technique (Letting & Armstrong 2003). During two pre-drop surveys all banded birds encountered, plus other individuals that were identifiable by their territories, were recorded as marked individuals. Resightings after the poison drop were considered to be recaptures (Letting & Armstrong 2003).

Before the first survey, birds were trained to approach the sound of a researcher clapping their hands by offering a mealworm as a reward. If no robin was encountered within 5 min, a recorded territorial robin call was played. A maximum of 20 min was spent at each territory, and if no robin was detected within this time frame, it was recorded as ‘not sighted’. Two surveys were done at each site before the poison drop and two surveys after the drop (Table 1).

Monitoring of robin nesting success

In the pre- (2010–2011) and post- (2011–2012) 1080-drop breeding seasons we recorded the nesting success of up to two clutches laid by each territorial pair. Walk-through nest surveys were carried out from the start of each breeding season in early September. Nests were found by offering the territorial male a mealworm. The male then called the incubating female off the nest to feed her, and by following the female it was possible to find the nests of all breeding pairs. Location coordinates of nest trees were recorded using a GPS unit, and a small piece of flagging tape was placed on an adjacent tree. Each active breeding pair was visited approximately weekly and their nesting activity (incubating; feeding nestlings; feeding fledglings) recorded. A nest was considered successful if at least one chick fledged. A maximum of two clutches per pair were monitored, typically the first two clutches of the season.

Because of the difference in forest types between the two sites, and potential differences in their predator guilds, we did not expect pre-drop nesting success to be identical at the two sites. Nesting success was defined as the percentage of nests that survived from initiation to completion (one or more fledglings produced). However, this measure of observed nest success is biased depending on the stage in the nesting cycle when the nest was found; that is, the longer a nest is monitored, the more likely it is a nest failure will be observed (Mayfield 1975). Recently, Armstrong et al. (2002) substantiated this bias in a nesting success study with North Island robins (*Petroica longipes*).

Table 1. Dates of mark–resight surveys of South Island robins (*Petroica australis*) before (pre-drop) and after (post-drop) aerial 1080 operations to control possums (*Trichosurus vulpecula*) at Silver Peaks and Silverstream, South Island, New Zealand.

Survey	Silver Peaks	Silverstream
Pre-drop 1	24 Aug. 2011	22 Aug. 2011
Pre-drop 2	26 Aug. 2011	25 Aug. 2011
Post-drop 1	20/21 Sep. 2011	28 Sep. 2011
Post-drop 2	22 Sep. 2011	29 Sep. 2011

To correct for bias in observed nesting success relative to actual nesting success, we analysed the data using a nest survival model developed by Rotella (2013) for the program MARK (White & Burnham 1999), which gives maximum-likelihood estimates of daily survival rate (DSR) of the nest and its variance over the entire nesting period (Rotella 2013). We divided the robin nesting period into incubation (18 days) and nestling (21 days) periods (Powlesland 1997). The DSR is based on the Mayfield estimate that accounts for the number of exposure days for each nest and the time period between nest initiation and first detection, when nests were not under observation (Armstrong et al. 2002; Rotella 2013). Analysis was carried out in R (R Development Core Team 2011) using the *mark* function in the package *RMark* (Laake 2013), where the model type was specified as ‘Nest’, which fits a nest survival model using a logit-link function.

We fitted four models that we regarded as ecologically relevant, based on the explanatory variables site and year: Site; Year; an additive model, Site+Year; and an interactive model, Site×Year, for which we predicted that nesting success should increase at Silver Peaks in the season of the 1080 drop (because there should be fewer predators present), but remain constant across years at Silverstream. In addition, we fitted a constant (Intercept-only) DSR model, and a time-trend model (Time), in which DSR increased or decreased as the breeding season progressed (Rotella 2013). The model set was ranked using sample-size-corrected Akaike Information Criterion (AIC_c) (Burnham & Anderson 2002). The Akaike weights of all models in the set summed to one and the model with the highest weight was considered to be the closest to reality (i.e. had the best fit to the available data). Support for each model was evaluated by assessing the change in AIC_c from the best model (ΔAIC_c) (Burnham & Anderson 2002). Models with a change of <2 from the best model have substantial empirical support, models with a change of 4–7 have considerably less support, and models with a change of >10 have essentially no support (Burnham & Anderson 2002).

In order to estimate yearly DSR at each of the two sites, we performed model-averaging on those models with $\Delta AIC_c < 4$. Model coefficients were averaged using the *model.avg* function in the package *MuMIn* (Bartoń 2009). This function provides two model-averaged estimates, the ‘subset average’ (or natural average method) and the ‘full average’ (or the zero method). We present both sets of averaged model coefficients (Table 2). However, predictions of DSR are based on ‘full averages’, as these estimates are corrected for upward bias (i.e. shrinkage is applied) of coefficients that only feature in models with a low model weight (Grueber et al. 2011).

Table 2. Model-averaged estimates for the effects of site, year, and the site by year interaction on the daily survival rate of South Island robins (*Petroica australis*). Estimates were averaged from the three models with strongest AIC_c support ($\Delta AIC_c < 4$; Table 4). Estimates are given on the logit scale (as produced by *RMark*).

Parameter	Estimate	Unconditional SE	Estimate with shrinkage
Intercept	4.514	0.388	4.514
Site	−1.740	0.412	−1.740
Year	−0.448	0.448	−0.231
Site×Year	0.040	0.767	0.006

Results

Change in rodent and possum abundance

Two chewcards (from the ‘follow-up’ survey at Silver Peaks) out of a total of 421 had unidentifiable bite marks and we sent them to P. Sweetapple for further analysis. Rats or mice were ruled out, but not possums, although the marks were atypical of possum tooth impressions. Because of this uncertainty we removed the two cards from the possum dataset.

Before the 1080 drop, the CCIs for rats and possums were significantly higher at Silverstream than at Silver Peaks (based on non-overlapping 95% CIs) (Fig. 2a, b), while those for mice were similar at the two sites (Fig. 2c). The CCIs for all three pest species decreased significantly after poisoning at Silver Peaks (Fig. 2), but did not change substantially at Silverstream. Furthermore, there was no significant change in the first post-drop survey and the second post-drop survey 69 days later at Silver Peaks. The observed rate of chewcards with rat or possum marks was zero for both post-drop surveys.

Adult robin survival

No known robins died as a result of the poison drop. All marked robins that were found before the 1080 drop at Silver Peaks ($n = 19$) were resighted afterwards (100% resighting rate). In addition, several unbanded birds were noted in locations where unbanded birds had been seen prior to the 1080 drop

(one unbanded pair and three unbanded individuals paired with banded birds). All 15 banded robins that were monitored during the same period at Silverstream were also resighted in subsequent surveys, as was one additional unbanded bird.

Nesting success

Based on the nest survival model of Rotella (2013), estimated nesting success at Silver Peaks was 65.4% pre-drop and 58.6% post-drop, whereas that at Silverstream pre-drop was 9.4% and therefore much lower than at Silver Peaks (Table 3). During the post-drop monitoring, only 12 nests (six breeding pairs) were located at Silverstream (four additional territories had only single males), one of which was successful, with an estimated nesting success of 5.3% (Table 3).

Model comparison gave equal support to the Site and Site+Year models, as they both had $\Delta AIC_c < 2$ (Akaike weights 0.48 and 0.38, respectively; Table 4). There was also some support for the Site×Year model (Akaike weight 0.14; Table 4). These results suggest that there were differences in nest DSR between sites, with Silver Peaks recording much higher nesting success overall, as reflected in the above-mentioned differences in observed nesting success. In addition, nest DSR differed between years within sites but did not show the predicted increase in nesting success at Silver Peaks after the 1080 drop (Table 3).

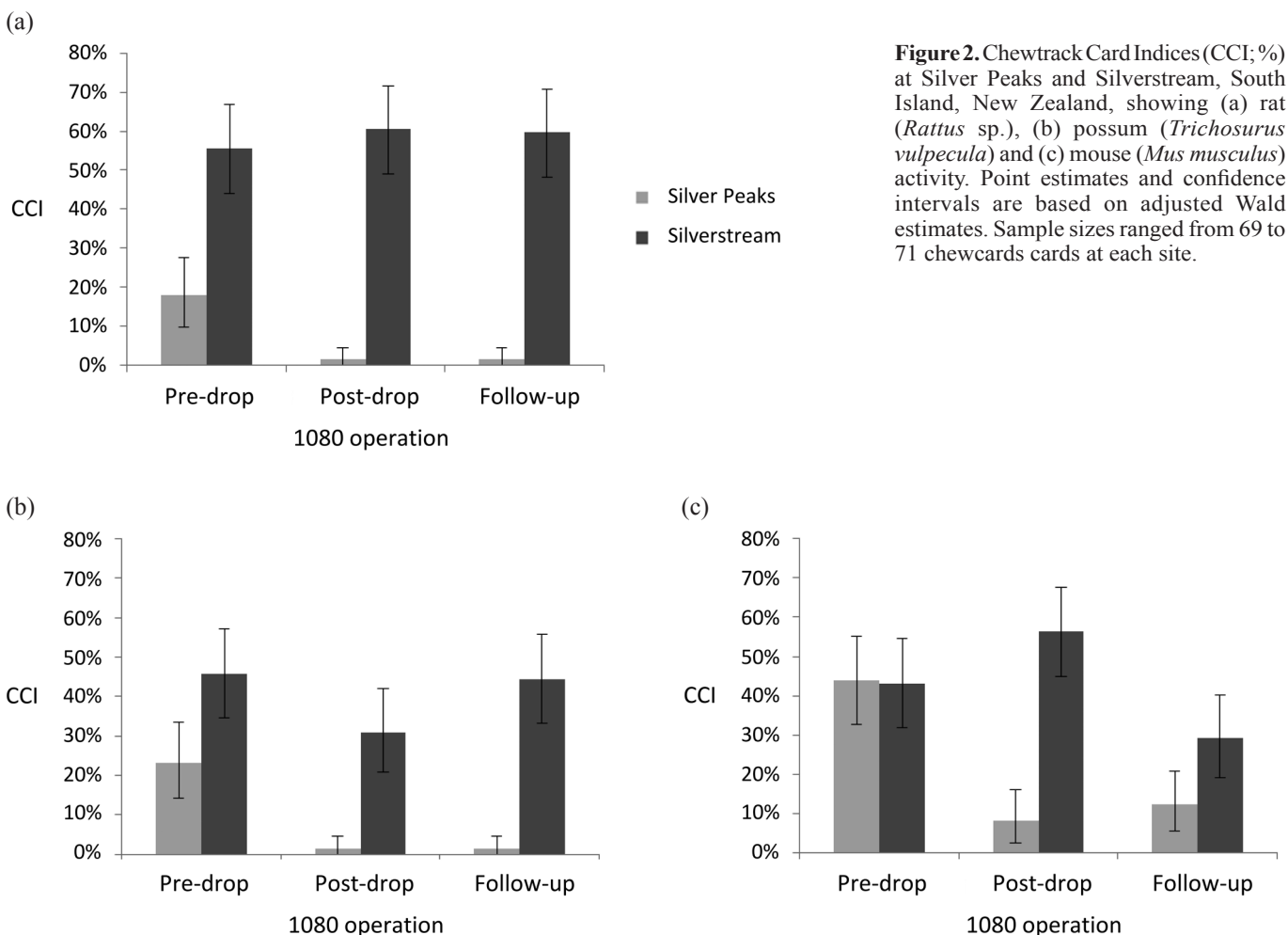


Figure 2. Chewtrack Card Indices (CCI; %) at Silver Peaks and Silverstream, South Island, New Zealand, showing (a) rat (*Rattus* sp.), (b) possum (*Trichosurus vulpecula*) and (c) mouse (*Mus musculus*) activity. Point estimates and confidence intervals are based on adjusted Wald estimates. Sample sizes ranged from 69 to 71 chewcards cards at each site.

Table 3. Total number of South Island robin (*Petroica australis*) pairs and nests monitored at two study sites (Silver Peaks and Silverstream, Dunedin environs, New Zealand) and observed nesting success. Daily survival rate was estimated from model-averaged parameters estimated with shrinkage (Table 2), and nest survival rate was estimated as daily survival rate raised to the power of 39 (days in an entire nesting period).

Site/Year	Pairs monitored	Nests monitored	Nesting success No. (%)	Daily survival rate (%)	Nest survival rate (%)
Silver Peaks					
2010	10	16	12 (75)	98.9	65.4
2011	12	21	14 (67)	98.6	58.6
Silverstream					
2010	10	16	3 (19)	94.1	9.4
2011	6	12	1 (8)	92.7	5.3

Table 4. Models used to explain variation in daily nest survival rates of South Island robins (*Petroica australis*) at Silver Peaks and Silverstream, Dunedin, New Zealand, during their 2010–2011 and 2011–2012 breeding seasons. K = no. parameters in each model; AIC_c = sample-size-corrected Akaike's Information Criterion; ΔAIC_c = difference in AIC_c value compared with best model (i.e. Site). Models with a change of c. <2 from the best model have substantial empirical support, models with a change of 4–7 have considerably less support, and models with a change of >10 have essentially no support (Burnham & Anderson 2002).

Model	K	AIC _c	ΔAIC _c	Model weight
Site	2	124.66	0.00	0.48
Site+Year	3	125.16	0.50	0.38
Site×Year	4	127.17	2.51	0.14
Time	2	142.33	17.67	0.00
Constant	1	145.68	21.02	0.00
Year	2	147.54	22.89	0.00

Discussion

Reduction in rodent and possum abundance

The aerial possum control operation at Silver Peaks was highly effective in reducing the rat and possum populations to very low levels (Fig. 2). House mouse numbers were reduced to very low levels as well (Fig. 2), which contrasts with an intensive study of aerial 1080 operations, where post-poison monitoring showed no significant reduction in mouse numbers (Nugent et al. 2011). In addition, a previous pen trial has shown that wild-caught mice predominantly avoided 1080 pellets, resulting in only 8% mortality (O'Connor et al. 2005). Another study showed that wild mice were able to identify and avoid cereal pellets containing 1080, and that prefeeding with non-toxic bait did not increase their subsequent intake of poisoned pellets (Fisher et al. 2009). Although mouse numbers can increase after poison operations, owing to competitive release as a result of the removal of rats (Nugent et al. 2011; Ruscoe et al. 2011), we observed no substantial increase in mouse CCIs by the end of the study. The above-cited studies were carried out in native bush, whereas the Silver Peaks study site is Douglas-fir-dominated exotic forest. Possibly, the availability of food for mice, such as invertebrates and seeds (Ruscoe & Murphy 2005), was low at Silver Peaks during the poison drop, increasing the bait acceptance by mice and leading to a high kill-rate.

Interactions between multiple pest species can impact on chewcard detection rates. In particular, rats can negatively affect the probability of detecting other species, particularly mice, because rats may gnaw off all baited parts of the cards (Sweetapple & Nugent 2011). The Silverstream CCI was

continuously high for rats throughout all three surveys, and some of the variation in the mouse CCI at Silverstream could result from rats reducing the detectability of mice, particularly in the follow-up survey.

Adult robin survival and nesting success

Multiple surveys have advantages over conducting just a single survey before and after a poison drop, as the additional data deliver more precise estimates and can be used to test the assumption that all animals have the same probability of being detected (Lettink & Armstrong 2003). As adult robins are highly territorial and sedentary once they settle (Taylor et al. 2005), detection probabilities often exceed 90% (Lettink & Armstrong 2003).

This is the first study to observe the responses of individually marked robins to aerial 1080 cereal pellet application preceded by prefeeding with non-toxic bait. Three previous studies examined the impact of aerially distributed cereal-based 1080 bait (without prefeed and at a substantially higher sowing rate of 5 kg per hectare) on individually marked robins (Veltman & Westbrooke 2011). In one of those studies (Saxon River, North-West Nelson) the sample size was two robins only, with both birds surviving. The other two studies had sample sizes of 17 (Long Ridge, Pureora Forest Park) and 35 robins (Pokeka, Wanganui National Park), with zero and nine individuals killed, respectively. The Long Ridge study was the only study that also had a non-treatment site ($n = 42$ birds), at which two birds died. These results led to the question of whether prefeeding would cause increased incidental robin mortality (Veltman & Westbrooke 2011; Veltman et al. 2014). At least in the case of our study, the answer appears to be no.

We found no evidence that the reduction of rats and possums to below detection level increased robin nesting success. This result may be explained by the fact that nesting success at Silver Peaks was already high before the poison drop (in 2010), perhaps because predator numbers were relatively low at that time, according to our chewcard results. Our CCIs for rats and possums at Silver Peaks were low compared with Silverstream even before the 1080 operation, but because we have no CCIs for rats and possums from previous breeding seasons, it is not possible to further assess the relationship between predator abundance and nesting success. However, in a previous study nesting success was relatively low at both Silver Peaks and Silverstream (between 2007 and 2009) when, on the basis of results from ink footprint tunnels, rats were relatively abundant at both sites in 2009/10 (Parker 2013). Owing to their arboreal foraging behaviour ship rats, together with possums, are likely to be key predators of robin nests (Brown 1997; Brown et al. 1998). The high nesting success rate and relatively low rat numbers that we recorded before the poison drop at Silver Peaks may have been unusual, but until further monitoring is done, we can only speculate that predator numbers and nesting success are likely to vary across years and across sites.

Finally, the benefits of one-off control of rats can be short-lived because rats can reinvade a treatment area relatively quickly, and rats that survive a 1080 operation can reproduce and quickly build up in numbers (Ruscoe et al. 2011). Therefore it is crucial to obtain an initial high kill-rate, and undertake follow-up control to prevent population recovery. How quickly rats will return to pre-poisoning numbers at Silver Peaks is unknown, but is one of the planned outcomes of an ongoing 5-year study. Despite public concern about the detrimental effects of 1080 on native wildlife, it appears from this study that a 1080 aerial operation had little negative impact on a robin population, although further replication is necessary.

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