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Impacts of aerial 1080 predator control on nest success and adult survival of South Island robins

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Abstract: Little is known about the impact of aerial 1080 control on nesting success and abundance of birds. The South Island (SI) robin (*Petroica australis*) is vulnerable to predation by exotic mammals, with declining populations on the east coast of the South Island of New Zealand. We document the effect of an aerial 1080 operation carried out in 2011 on the relative abundance of brushtail possums (*Trichosurus vulpecula*), rats (*Rattus* spp.), house mice (*Mus musculus*), and on SI robin nest success, comparing a site receiving aerial 1080 (Silver Peaks) to a non-treated site (Silverstream) in the Dunedin region. Although control reduced predator numbers, the relative abundance of house mice had recovered to pre-control levels after five months, as had rats after 12 months and brushtail possums after 15 months. There were no significant differences in SI robin nest success and nest survival over time between the treated and untreated sites, although negative relationships between nesting success and possum and rat abundance were observed.

Keywords: biodiversity, invasive mammal impacts, mice, New Zealand; pest management, *Petroica australis*, possums, predation, rats

Introduction

The aerial spread of sodium fluoroacetate (1080) in cereal baits is the most cost-effective method to control several pest mammal species over large and remote areas in New Zealand (Brown & Urlich 2005; Parkes et al. 2017). Annually about 635 000 ha of remote forests are treated with 1080 to control introduced brushtail possums *(Trichosurus vulpecula)*, ship rats *(Rattus rattus)* and (by secondary poisoning) stoats *(Mustela erminea)*, which consume dead and dying possums and rats (Eason et al. 1993; Alterio 2000; Eason et al. 2011; Byrom et al. 2016; Parkes et al. 2017). The Department of Conservation repeats aerial 1080 operations every 2–7 years to protect forest canopies from possums, or as is required to minimise the impacts of introduced mammals on native species following masting events (Brown & Urlich 2005; Byrom et al. 2016; Parkes et al. 2017). TBfree New Zealand similarly applies 1080 every 4–10 years to reduce TB prevalence in possums (Brown & Urlich 2005; Byrom et al. 2016; Parkes et al. 2017), although the frequency of control can depend on possum trap catch following operations (Brown & Urlich 2005).

Improvements in the efficiency of aerial 1080 operations, such as the use of pre-feeding of non-toxic cereal baits to increase bait uptake (Coleman et al. 2007; Nugent et al. 2011), have led to high and generally reliable proportions of both possums (95%) and rats (90%) killed in areas treated with 1080 (Innes et al. 1995; Miller & Miller 1995; Coleman et al.

2007). However, rat abundances can recover to pre-control levels within six months of a 1080 operation (Innes et al. 1995, 1999), while possums can recover in 3–4 years (Innes et al. 1999). It is also possible that when one introduced predator is targeted, others can increase to pre-control levels due to their release from competition and predation (Sweetapple & Nugent 2007).

SI robins are an endemic passerine currently listed as not threatened (Robertson et al. 2016). Although common in parts of Fiordland, Arthur's Pass and Nelson Lakes National Parks, their distribution is patchy and declining (Higgins & Peters 2002). There are only two small populations on the east coast of the South Island, in Kaikoura and Dunedin, both of which are affected by predation by rats, possums, and stoats (Powlesland 1983; Jones 2016). There is strong evidence that SI Robins were common around Dunedin as recently as the mid-1990s (McEwen 1987; IG Jamieson, pers. obs.), but numbers have declined, and only three small sub-populations remain: one at Orokonui (a fenced predator-free sanctuary), one at Silverstream, and one at Silver Peaks where aerial 1080 predator control has occurred every six years since 2004 (Fig. 1).

Most research considering the aerial application of 1080 has focused on impact on predator numbers, with fewer evaluating biodiversity outcomes. A meta-analysis by Fea et al. (2020) explored population responses of a range of bird species to varying intensities of predator control (including periodic possum control using 1080) in relation to body size,
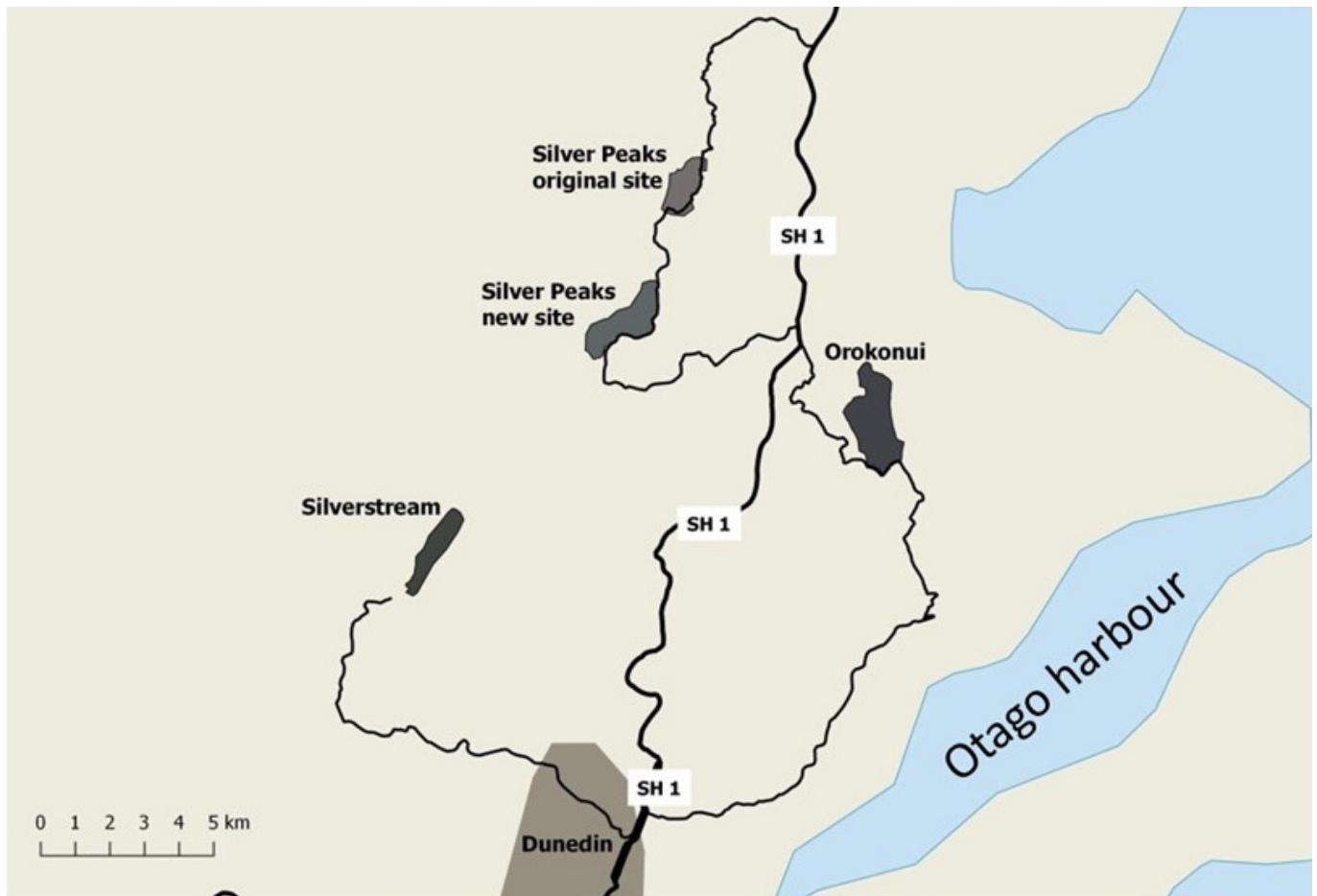
Figure 1. Location of study sites in relation to Dunedin City: Silver Peaks (exotic Douglas fir forest), Silverstream (native kānuka forest); and Orokonui eco-sanctuary (native regenerating kānuka-broadleaved forest). The blue areas are ocean.

degree of endemism, and nesting habit. Robins were reported as only one of two species that showed a significant positive correlation between the populations' response to mammal control and the intensity of the control. Byrom et al. (2016) reviewed outcomes following aerial possum control and found improved bird nesting success after aerial 1080 control in seven of eleven studies of mostly cup-nesting but also cavity-nesting species, as well as one species, the North Island brown kiwi (*Apteryx mantelli*), which nests in burrows. Five of six studies of multiple native bird species that monitored adult bird abundances reported increases in abundance; however, no effects of predator control were found for kākā (*Nestor meridionalis*) and kererū (*Hemiphaga novaeseelandiae*) (Powlesland et al. 2003). Most of these studies measured biodiversity outcomes over a relatively short time period following aerial control (1–3 years); there has been little focus on medium and long-outcomes (Byrom et al. 2016). Schadewinkel et al. (2014) monitored the Dunedin SI robin sub-populations and predator abundance for one year prior to, and one year following aerial 1080 control at Silver Peaks in 2011. We combine the data from this study with an additional five years of data to examine outcomes over a typical cycle of control for possums. We also compare nesting success between incubation and nestling stages, and compare overall nestling success with values obtained from a predator-free environment, the Orokonui Ecosanctuary.

Methods

Here we present data collected during seven robin breeding seasons in Silver Peaks (the site of a 1080 control operation; i.e. the treated site) and Silverstream (the non-treated site) during 2010/11 to 2016/17, and three seasons at the Orokonui Ecosanctuary (a site where predators were completely absent). The Silver Peaks study site was situated in a 5100 ha area that received aerially broadcast cereal pre-feed (spread at a rate of 1.5 kg ha^{-1} on 9th August 2011) followed by 1080 poison pellets (applied at 2 kg ha^{-1}) on 5th September 2011 (RS5 cereal pellets with a cinnamon lure (Animal Control Products, Whanganui, NZ; Schadewinkel et al. 2014).

Silver Peaks comprises a block of exotic Douglas fir (*Pseudotsuga menziesii*) plantation (100 ha) with small areas of Monterey pine (*Pinus radiata*), owned and harvested by City Forests Ltd; see Schadewinkel et al. (2014) for a more detailed description of site characteristics. In the winter of 2015, City Forests Ltd began harvesting the study site and consequently we shifted the study site to an adjacent block of younger mature Douglas fir that was not due to be logged until after our study was completed.

The untreated or control site, Silverstream, is located in kānuka (*Kunzia ericoides*)-dominated native forest (100 ha). In response to low nest survival in the 2011/2012 season at this site (8% from six pairs) and concern this small population

could vanish, Forest & Bird NZ deployed 50 Goodnature[TM] rat traps in December 2012. These traps remained in place until the end of the study, although they were not operational (i.e. not provided with fresh lure or gas cylinders) in the 2014/2015 season: in the final two seasons they were serviced by the University of Otago, and baited for stoats in every tenth trap, because stoat home ranges are larger than those of rats. The Silver Peaks and Silverstream sites are c .10 km apart with no known movement of robins between them.

Orokonui eco-sanctuary is a 307 ha area of mostly regenerating native kānuka-broadleafed forest enclosed by a predator-proof fence. Monitoring of SI robins began in 2010 subsequent to the translocation of 25 robins from Silver Peaks and Flagstaff into the eco-sanctuary. Another translocation of 20 juveniles occurred in 2011 from Silver Peaks. Monitoring in Orokonui ceased after the 2013/14 breeding season as the population was well-established and growing rapidly.

SI robins are a monogamous, insectivorous, ground-foraging species (c. 35g) that occupies mature native and exotic forest with a closed canopy and open understorey (Higgins & Peter 2002). The breeding season extends from August until late December, during which time the female incubates two to three eggs (rarely four) in each of up to three successful clutches (Flack 1979). SI robins are poor dispersers and occupy the same territories from one season to the next (Higgins & Peter 2002). The vulnerability of breeding robins to introduced predators makes them a good indicator species for the effectiveness of predator control operations (Brown 1997, Powlesland et al. 1999). Territory locations in the three sites of this study were well known during all breeding seasons, and the majority of territorial birds were banded with unique colour bands for individual identification. All new robin pairs were found and banded at the new Silver Peaks site in 2015 and were monitored for the remainder of the project.

Monitoring predator abundance

We monitored the relative abundances of possums and rodents (rats and house mice *Mus musculus*) at both sites 34 days before the poison drop (on 1st August 2011), 11 days afterwards and then 80 days afterwards (during the middle of the robins' nesting period in November) following the methods described in Schadewinkel et al. (2014). Both sites were subsequently monitored in February 2012, and then quarterly for the following five breeding seasons, until December 2016.

Predators were detected from tooth impressions on chewtrack cards (CTC; Sweetapple & Nugent 2011; Connovation, Manukau, NZ) that contained peanut-butter-based Connovation FeraFeed 213 Special Blend paste as bait in the flutes. Indices derived from chew track cards have been correlated with other established indices of pest detection, such as tracking tunnels and trap-catch indices, and the proportion of cards bitten by each of the species (the CTCI) was assumed to be positively related to the abundance of that species (Sweetapple & Nugent 2011). Because chewcards are sensitive to the presence of rodents, they are useful for monitoring low-density rat populations, which are expected after aerial 1080 operations (Sweetapple & Nugent 2011).

At each study site we nailed between 69 and 71 chewcards to tree trunks 30 cm above the ground, at 50-m intervals along a continuous tracking line connecting known robin territories. Cards at Silverstream were placed 1–5 m off the track along public walking trails. The same locations were used in all monitoring sessions. Due to the change in the treatment site we set up a new CTC line in Silver Peaks from June 2015 for the

remainder of the study. We retrieved cards after nine to 11 days and examined them for bite marks using hand lenses, comparing tooth marks to colour reference photographs provided by P. Sweetapple (Manaaki Whenua Landcare Research). Since virtually all detections occur within the first seven days of deployment, we did not correct the CTCIs for the number of days that cards were out (Nugent et al. 2011).

We calculated point estimates of rat and stoat abundance and 95% confidence intervals for CTCIs using the adjusted Wald method, which is most appropriate when dealing with proportions from small sample sizes (Sauro & Lewis 2005; Lewis & Sauro 2006), but gave point estimates slightly greater than 0 when actual tracking rates were 0. We inferred significant changes in mammal abundance when 95% CIs did not overlap. This simple method allowed us to detect large changes in pest abundance after the poison drop.

Robin nest success

We monitored nest success of the first two clutches of each territorial pair at both sites during the breeding season both before the 1080 operation (2010/11) and during six breeding seasons following the drop (2011/12 to 2016/17) following the methodology of Schadewinkel et al. (2014). In Orokonui, nest success was monitored during the 2011/12, 2012/13 and 2013/14 breeding seasons. We checked each territorial pair once a week by walking through robin territories. After males were offered mealworms, they would consistently call the female mate off the nest and we were then able to follow her back to the nest. If a female previously known to be incubating at a nest of known location did not return to her nest 20 minutes after being fed by the male, the nest was recorded as failed for that visit. If the male and female did not bring worms to a nest previously known to contain chicks during a 10 minute period, and were not feeding fledglings, the nest was recorded as failed. Robins continue to take food back to their young in the presence of an observer, but we observed the birds from a distance so as not to disturb them or interfere with the feeding of young. Nests were considered to be successful if at least one fledgling was produced from the nest.

We used the definitions for nesting success outlined in Schadewinkel et al. (2014); i.e. as the percentage of nests that survived from initiation to completion (one or more fledglings produced), and the same analytical approach. To correct for the bias introduced by finding nests at different stages of the nesting cycle, we analysed the data using a nest survival model developed by Rotella (2013) using 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). DSR is based on the Mayfield estimate, which accounts for the number of days each nest is exposed and the period of time from when the nest was initiated to when it was first detected, during which time it was not being monitored (Armstrong et al. 2002; Rotella 2013). We divided the robin nesting period into incubation (18 days) and nestling (21 days) periods (Powlesland 1997; Schadewinkel et al. 2014). Analysis was carried out using the RMark package (Laake et al. 2012; Laake 2013): the model type was specified as 'Nest', which fits a nest survival model using a logit-link function.

Following the methodology of Schadewinkel et al. (2014), we fitted seven models that we regarded as ecologically relevant, using the explanatory variables site, year and stage (incubation or nestling). The first four models were (1) Site; (2) Year; (3 & 4) two interactive models. In (3) Site*Year, nesting

success was predicted to rise over several seasons at Silver Peaks subsequent to the 1080 drop due to lower numbers of predators, but remain constant across years at Silverstream. In (4) Site*Stage+Time, we predicted nest survival would be lowest during the nestling phase when the nest is most likely to attract predators. Model (5) was an additive model, Stage+Time, which predicts that DSR is both influenced by the stage of the nest and varies over time as the breeding season progresses. We also fitted (6) a constant (Intercept-only) DSR model, and (7) a time-trend model (Time), in which DSR increased or decreased as the breeding season progressed (Rotella 2013).

We ranked the model set using the sample-size-corrected Akaike Information Criterion (AICc; Burnham & Anderson 2002) and Akaike weights; the model with the highest weight was considered to be best supported by the available data. We evaluated support for each model by assessing the change in AICc from the best model (ΔAICc; Burnham & Anderson 2002). Models with a change in AICc 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).

Nesting data from all breeding seasons were combined to estimate nesting success at the incubation and nestling stages at both sites. Differences in nesting and incubation stage success were considered important if their confidence intervals did not span zero.

Results

Changes in rodent and possum abundance

Before the 2011 1080 control operation, CTCIs for rat and possums were much lower at Silver Peaks (17.8% and 23.3%, respectively) than at Silverstream (55.6% and 45.8%; Fig. 2 & b). The mouse CTCI was similar at both sites (43.8% Silver Peaks and 43.1% Silverstream, Fig. 2). CTCIs for all three species significantly decreased after the 1080 drop at Silver Peaks to low values (rat 1.4%, possum 1.4% and mice 8.3% Fig. 2) but did not change significantly at Silverstream, the untreated site. Estimated control efficacy was thus 92% for rats, 94% for possums, and 81% for mice.

Rat CTCIs were different between sites and peaked in abundance at different times. Rat CTCI in Silver Peaks had almost recovered by May 2012 (15.1%), and within a year abundance was twice that seen before the drop (39.7% Sept 2012; Fig. 2). Abundance subsequently remained mostly high before dropping during the final two years. The rat CTCI in the untreated Silverstream site was high through to December 2012, and relatively low thereafter, probably due to the deployment of the Goodnature traps.

Possums at Silver Peaks were slower to recover after the 1080 control operation, but had surpassed pre-control levels by Dec 2012 (31%), 15 months after the control operation (Fig. 2). The CTCI increased steadily over time and highest numbers were seen in June 2016 (94.6%). The untreated Silverstream site had high possum CTCIs (> 70%) from Dec 2012, higher than at Silver Peaks in every quarter (excluding June 2016).

Mouse CTCI recovered the fastest in Silver Peaks: numbers increased substantially to almost double pre-drop levels by February 2012 (78.1%) just five months after the control operation (Fig. 2), then fluctuated but remained relatively low from March 2012. Mouse CTCI numbers fluctuated over

time at the untreated Silverstream site but were relatively low until 2016.

Nesting success

Daily nest survival rates and estimated nest survival are presented for all three sites in Table 1. There was no support for the Site*Year model (Table 2), indicating no significant change in nest success at either site over the course of the study. Estimated SI robin nest survival in Silver Peaks was 66.7% before the 1080 control operation and 54.1% afterwards (Table 1; Fig. 3). Nest survival at Silver Peaks decreased over time from 2010/11 to 2013/14; there was a slight increase in 2014/15, but numbers decreased significantly in 2015/16 to 12.5%. In the following breeding season nest survival recovered to 45%, similar to the value for the 2014/15 season.

Silverstream experienced much lower nest survival in 2010/11 (13.3%); this decreased to 2.7% in 2011/12 (Table 1; Fig. 3). Subsequently nest survival increased gradually each season and was at the highest recorded level of 22.7% in 2016/17, but remained much lower than the nest success at Silver Peaks, except for the 2015/15 breeding season (21.4% nest success at Silverstream compared to 12.5% at Silver Peaks). Estimated nest survival of robins in the predator-free Orokonui eco-sanctuary ranged between 74% and 77% over the three years monitoring took place (Table 1).

The model comparisons show strong support for the Site*Stage+Time model (Akaike weight 0.86; Table 2), indicating breeding stage influences nest survival at these sites. DSR for the nestling stage was significantly lower than during the incubation stage at Silverstream (RMARK est. logit diff. incubation-nestling = −0.576, 95% confidence interval: −0.576 to −0.576); Silver Peaks showed the same significant pattern (RMARK est. logit diff. incubation-nestling= −0.250, 95% confidence interval: −0.268 to −0.534; Table 3). There was some support for the site model (Akaike weight 0.097), suggesting that nest success was significantly higher at Silver Peaks than at Silverstream (Table 3; Fig. 4).

Discussion

Predator abundance

The 2011 1080 aerial control operation in Silver Peaks reduced rat and possum numbers to almost undetectable levels, as reported in Schadewinkel et al. (2014) and in other studies (Eason et al. 1993; Innes et al. 1995; Powlesland et al. 1999, 2003; Coleman et al. 2007). However, the reduction of mouse numbers to low levels is a novel result; only small declines have previously been reported following aerial 1080 control (Miller & Miller 1995; Nugent et al. 2011) and it has been suggested that mice actively avoid 1080 pellets (Fisher et al. 2009). Food supply for mice at Silver Peaks may have been low before the drop, which increased their uptake of bait (Schadewinkel et al. 2014).

Mice recovered their numbers fastest and by five months post-control had doubled their pre-control levels. Other studies have also reported rapid increases in mice within 2–6 months following aerial control (Innes et al. 1995; Miller & Miller 1995; Murphy et al. 1999). Breeding in female mice is dependent on food availability, but in the right conditions mice can produce litters of up to six young every 20–30 days (Ruscoe & Murphy 2005). Numbers remained high for a year and then declined and fluctuated during the remainder
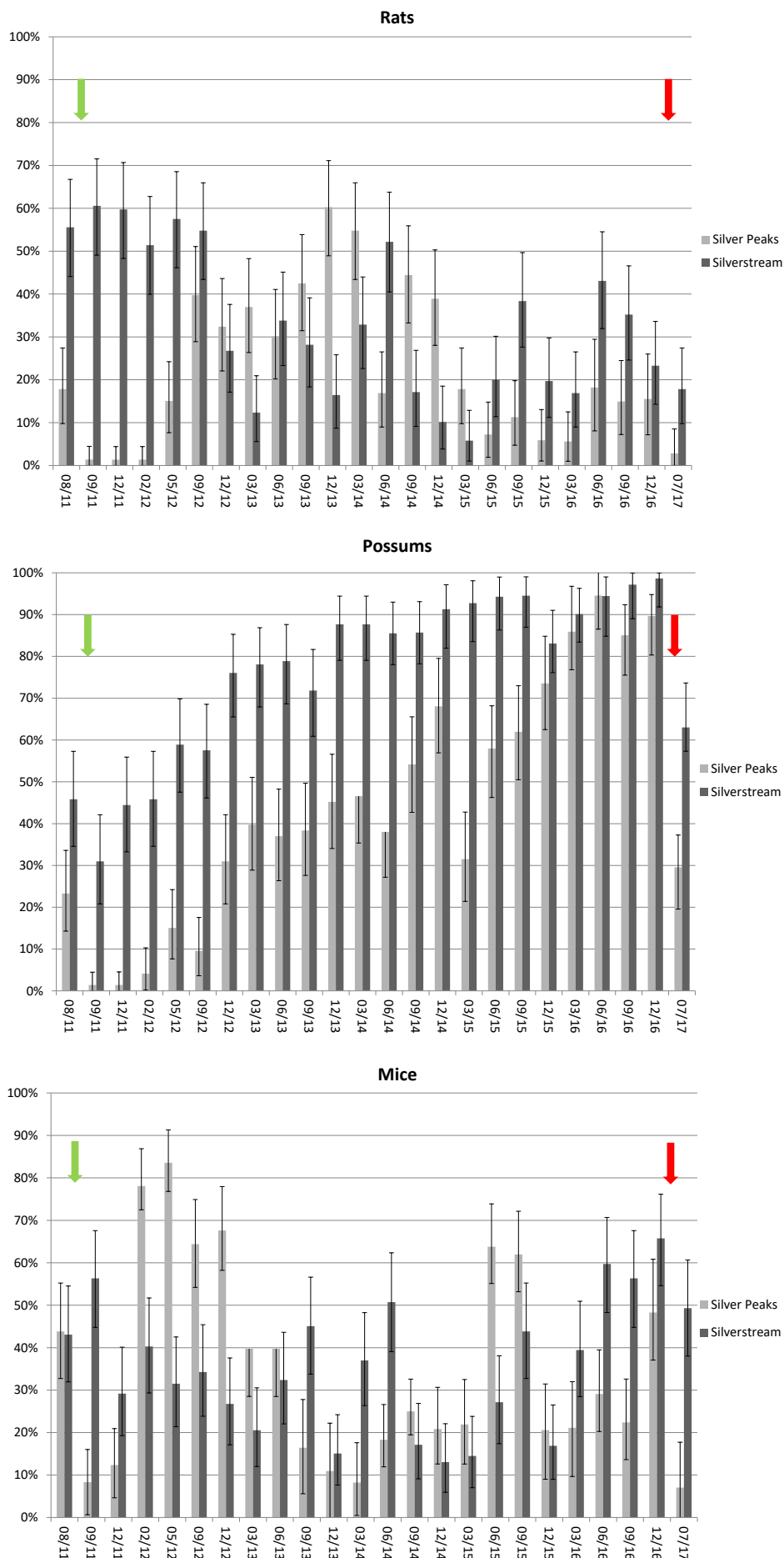
Figure 2. Chew-Tracking-Card Indices (CTCI) for rats, possums and mice activity from August 2011 (pre-1080 drop) till July 2017. Point estimates and confidence intervals are based on adjusted Wald estimates. Green arrows indicate when the first aerial 1080 drop was carried out. Red arrows indicate the second aerial 1080 drop.

Table 1. Summary of nesting success data based on the first two clutches of the season at three study sites. Predator populations were depressed following the 1080 drop between 2010/11 and 2011/12. Daily and nest survival rate estimates were derived from the interaction model (Site * Year)

Site/Year	Pairs Monitored[#]	Nests monitored	Nests successful	Daily survival rate (DSR)	Est. nest survival*
Silver Peaks					
2010/11	10	16	12 (75%)	98.97%	66.68%
2011/12	12	21	14 (67%)	98.44%	54.12%
2012/13	12	20	12 (60%)	98.35%	52.34%
2013/14	16	24	10 (42%)	97.42%	36.02%
2014/15	17	30	14 (47%)	97.88%	43.29%
2015/16	12	19	2 (11%)	94.80%	12.47%
2016/17	9	15	8 (53%)	98.01%	45.72%
Silverstream					
2010/11	10	16	2 (13%)	94.97%	13.34%
2011/12	6	12	1 (8%)	91.15%	2.69%
2012/13	6	11	0 (0%)	92.51%	4.79%
2013/14	10	18	3 (17%)	94.35%	10.36%
2014/15	15	29	4 (14%)	95.26%	15.05%
2015/16	13	24	6 (25%)	96.12%	21.38%
2016/17	11	21	6 (29%)	96.27%	22.71%
Orokonui					
2011/12	10	19	14 (74%)	98.87%	64.28%
2012/13	15	29	24 (83%)	99.33%	77.01%
2013/14	41	70	56 (80%)	99.16%	72.12%

[#]Includes only pairs with nests observed at incubation and/or nestling stage
*Based on DSR value from interaction model (Site * Year) raised to the power of 39 (days for an entire nesting period)

Table 2. Factors explaining variation in daily nest survival rates (DSR) of SI robins' nests in Silver Peaks and Silverstream over seven breeding seasons. "Stage" refers to incubation and nestling stages; "Time" refers to whether the daily survival rate increased or decreased as the breeding season progressed, "Site" refers to Silver Peaks and Silverstream and "Year" refers to the breeding season (2010/11, 2011/12, 2012/13, 2013/14, 2014/15, 2015/16 and 2016/17).

Model	K[1]	AICc	Delta AICc	Weight
Site * Stage + Time	5	776.98	0.00	0.859
Site	2	781.35	4.37	0.097
Site * Year	14	782.91	5.93	0.044
Stage * Time	3	806.08	29.10	0.000
Time	2	808.08	31.10	0.000
Constant	1	808.65	31.67	0.000
Year	7	816.10	39.11	0.000

[1]Number of parameters in particular model

Table 3. Observed pairs, nests and apparent nesting success for the first two clutches of SI robins of each season (2010–2016). Daily and nest survival rate estimates were derived from the model (Stage * Site + Time).

Site/Year	Pair-Years monitored	Nests monitored	Nests successful	Daily survival rate (DSR)	Est. nest survival*
Silver Peaks	88	147	72 (48.98%)		
Incubation				98.42%	53.69%
Nestling				98.03 %	45.94%
Silverstream	71	131	22 (16.79%)		
Incubation				96.93%	29.64%
Nestling				94.86%	12.78%

* Based on DSR value from interaction model (Site * Year) raised to the power of 39 (duration of an entire nesting period).
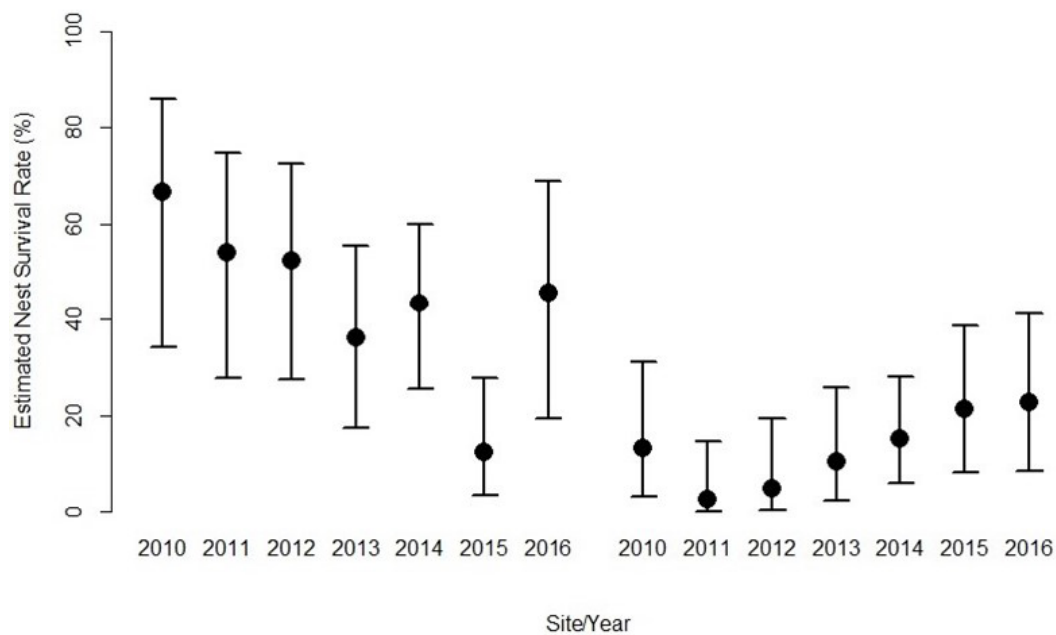
Figure 3. Estimated nest survival rate for monitored nests by site and year as estimated from our interaction model Site ×* Year (see Table 4) with point-estimate. ± 95% CI. Values for Silver Peaks (site of the 1080 operation) are on the left, and Silverstream (no predator control) are on the right.
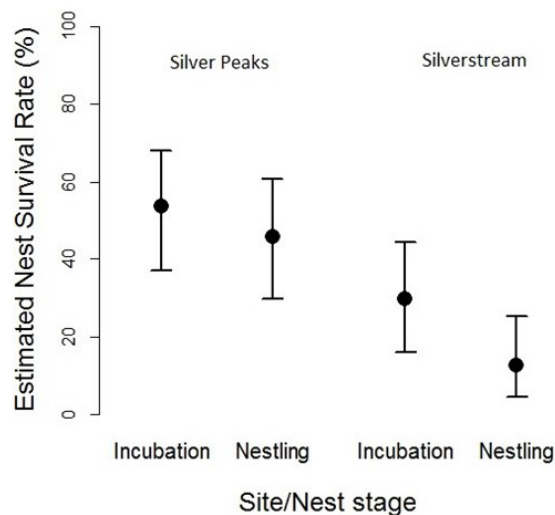

Figure 4. Estimated (± 95% CI) nest survival rate at incubation and nestling stages for Silver Peaks (site of 1080 operation) and Silverstream (no predator control) as estimated from our Stage + Time model (see Table 3).

of the study. The reduction of rats to almost undetectable levels removes competition and predation pressure on mice (Tompkins & Veltman 2006; Ruscoe et al. 2011; Goldwater et al. 2012), contributing to increases in mice numbers. High rat abundances might also affect the detectability of species such as mice, as rats can remove large chunks of the chew track cards (Sweetapple & Nugent 2011).

Rat relative abundance took 12 months to recover following the 1080 control operation at Silver Peaks. Ship rats are less fecund than mice, but can produce an average of 10.9 young per female per year, with a lifetime productivity of 16 young per female (Daniel 1972; Innes et al. 2001). Once recovered, numbers initially remained high, exceeding pre-drop levels and in some years peaking during the robin breeding season, but then decreased to relatively low numbers during the final

two breeding seasons. While rats prey on mice, they are not dependent on them (Tompkins & Veltman 2006). Rats and mice breed in summer and autumn, resulting in peaks in abundance over autumn and winter (Innes et al. 2001), although this trend was not seen at either site in this study. Daniel (1978) suggested that rat abundances are influenced by autumn food availability. More food over winter can also lengthen the breeding season and increase density (Innes et al. 2001). However, seasonal variation in rat abundance is thought to be small compared to year-to-year variation (Innes et al. 2001), especially in beech (Nothofagaceae) forests where large amounts of seedfall following masting of beech cause irruptions of mice and rats (King 1983; Fitzgerald et al. 1996; Choquenot & Ruscoe 2000; Ruscoe et al. 2011). The change of study sites in Silver Peaks from an old block to a younger forest stand may have contributed to the low rat CTCI in the final two breeding seasons, since ship rat abundances can be much lower in young exotic forest stands compared to older stands (King 1996; Innes et al. 2001). As rats and possums compete for fruit and seeds (Sweetapple & Nugent 2007; Ruscoe et al. 2011), a reduction in possums may result in more food for rats. Nonetheless, no detectable increase in rat abundance was associated with more than five years of possum removals on the Otago Peninsula (Wilson 2017).

Possum relative abundance was slowest to recover and remained low for 15 months after the 1080 control operation. Possums produce one or two young per year, depending on food availability (Kerle 1984). Abundance increased gradually and reached high levels in 2016. In contrast, Innes et al. (1999) found possums took 3–4 years to recover from an aerial 1080 operation, and Cowan et al. (1997) found that a possum population reduced by 95% took five years to recover to half pre-control levels. The more rapid recovery we observed was likely driven by immigration from surrounding untreated areas, as population regrowth *in situ* from surviving individuals would take much longer (Veltman & Pinder 2001; Brown & Urlich 2005). Fluctuations in rat abundance and low rat abundances during the final two years of the study may have reduced competition for food sources, allowing possum populations to reach high levels.

At Silverstream, where no aerial 1080 control took place, rat abundance was significantly higher than at Silver Peaks before the control operation and remained moderately high during the first year of the study, decreasing in December 2012 and remaining low except for a couple of peaks in June. The difference in forest type between the treatment (exotic Douglas fir) and non-treatment (native kānuka) sites may have influenced rat numbers initially at least. Rats have been shown to be more abundant, larger and heavier in native than in exotic forest, although these studies compared native podocarp/broadleaf forest to *Pinus radiata* plantation with stands of eucalyptus and Douglas fir (King et al. 1996; Innes et al. 2001). Seasonal peaks in rat abundances are not as evident in kānuka scrub and forest as they are in mature forest of other native species (Ogden & Gilbert 2009). The introduction of Goodnature[TM] rat traps to Silverstream in December 2012 to prevent the extirpation of this SI robin population most likely contributed to the reduction in rat numbers at this site.

The fluctuations in mouse numbers observed at Silverstream more-or-less mirrored those at Silver Peaks, suggesting food availability was the dominant influence driving mouse population size (King 1983; Fitzgerald et al. 1996; Choquenot & Ruscoe 2000; Ruscoe et al. 2004). Possum numbers were higher at Silverstream than at Silver Peaks and gradually increased over time, with extremely high relative abundance since December 2012.

Nesting success

SI robin nesting success did not increase in the breeding season following the aerial 1080 operation at the Silver Peaks site, despite rat, possum and mice numbers being reduced to low levels. This absence of any increase might be because nest success was already reasonably high prior to the 1080 operation due to relatively low predator numbers at that time (Schadewinkel et al. 2014). Indeed, nest survival at Silver Peaks before the 1080 operation was as high as that recorded in the predator-free Orokonui Ecosanctuary in 2011/12. It is not unusual for rodent numbers to fluctuate, influenced by predation, competition, forest type and food availability (Daniel 1978; King 1983; Fitzgerald et al. 1996; King et al. 1996; Choquenot & Ruscoe 2000; Innes et al. 2001; Tompkins & Veltman 2006; Ogden & Gilbert 2009).

Nest success did not increase over the years following the 1080 operation in Silver Peaks, suggesting that the aerial drop had no immediate or long-term positive effect on nest success of SI robins at this site on this occasion. The rapid recovery of rats, possum and mice to levels exceeding pre-drop levels likely contributed to this absence of effect. Another potential factor that could have prevented an increase in nest success is that the reduction of rodents to low levels in Silver Peaks may have caused stoats to switch to eating robins. Stoats mostly prey on rats and mice (Tompkins & Veltman 2006); however, the removal of rodents through pest control operations can cause stoats to switch to preying on birds (Murphy & Bradfield 1992; Murphy et al. 1998). Stoats are known predators of robin nests (Jones 2016), and we recorded one instance of stoat predation at the Silver Peak site, although most of the nests at this site were too high for camera monitoring. Stoats can be reduced by secondary poisoning from aerial 1080 operations (Alterio et al. 1999; Gillies & Pierce 1999; Murphy et al. 1999), although Powlesland et al. (2003) found no change in mustelid abundances after a control operation and attributed this to their low densities prior to control and rapid reinvasion. The relative abundance of stoats at both our sites was unknown

as they are not attracted to chew track card bait.

At the non-treatment Silverstream site, nest success was lower than at Silver Peaks in all but one year. Both nest site placement and predator abundance could have been responsible for the lower nesting success. Nests are situated much higher above the ground at Silver Peaks, where the predominant tree species is Douglas fir. Martin et al. (2000) found nests that are more accessible to terrestrial predators are more susceptible to predation, and it is possible that the nests at Silverstream, which were lower above the ground in native kānuka, were more vulnerable. Native forest can also support higher numbers of rats than exotic forest (Innes et al. 2001), and rat abundance was higher at Silverstream than at Silver Peaks during the first season (2011/12), and in the final two seasons (2015/16 and 2016/17). Nest survival at the Silverstream site improved from the 2013/14 season onwards, possibly in response to lower rat numbers following the deployment of Goodnature[TM] rat traps in December 2012 to avert the extirpation of this population of robins, which had experienced 8% nest survival (n = 6 nests) in 2011/12 and 0% (n = 6 nests) in 2012/13. However, the relationship between rat numbers and nest success is not clear, as despite rat trapping in the final two season, rat numbers as indicated by the CTI were not substantially reduced, yet nest success was the highest recorded at this site. It is also possible at this site that the initial reduction in rat numbers could have caused stoats to switch to eating robins as prey. While stoat traps were introduced into Silverstream for the final two breeding seasons, we did not record any stoats as having been caught by the traps, and stoats were recorded preying on robin nests at this site.

At both sites daily survival of nests was significantly lower during the nestling stage than during incubation. Nestlings are more conspicuous to predators due to the noise made by chicks begging (Redondo & Castro 1992; Leech & Leonard 1997), and greater activity as parents move to and from the nest to feed nestlings (Martin et al. 2000). Predation is a common limiting factor for most songbird species (Zanette et al. 2006; Cresswell 2011), including the Dunedin SI robin population (Jones 2016), as indicated by higher nest survival in the predator-free Orokonui Ecosanctuary. Nest cameras deployed in Silverstream during the 2014/15 and 2015/16 breeding seasons found that unsuccessful nests were more often preyed on at the nestling stage, although the difference in estimated nest survival between incubation and nestling stages was not significant (Jones 2016).

Adult survival of Dunedin robins was reasonably high, with an 84% chance adults would survive from one year to the next (Jones 2016). This value is similar to 86% adult survival of rifleman (*Acanthisitta chloris)* on a predator-free island (Leech et al. 2007), 77% for NI robins (*Petroica longipes*) in the absence of rats (Parlato & Armstrong 2011), and between 88% and 90% for NI saddlebacks (*Philesturnus rufusater*) introduced to a rat-free island (Armstrong et al. 2005). Adult survival was not influenced by predator abundance, and no adult robins were recorded from camera nest footage as having been preyed on over the two breeding seasons during which cameras were used, although adults did disappear during and between seasons (MAT Jones, pers. comm.). The remains of an adult female that had been killed by a predator were found at Silver Peaks. There was also no reduction in marked adult robins immediately following the 1080 drop (Schadewinkel et al. 2014). Robins have high site fidelity and are highly territorial, only infrequently moving away from their breeding site (Higgins & Peter 2002; Armstrong 2017). Adults are also

wary towards predators and display anti-predator behaviours in response to predators near the nest (Flack 1976; Maloney & McLean 1995).

While adult survival does not appear to be affected by predator abundance, and nest survival did not increase in the season following the 1080 aerial drop at the Silver Peaks sites, the lower nest survival at the Silverstream site, which was associated with higher relative abundances of possums and rats in most years, suggests that predators are having a negative effect on nest success of Dunedin's SI robins. North Island robin populations are also limited by rat abundance and respond positively to continuous rat control (Armstrong 2017), and in their meta-analysis of multiple studies of a range of New Zealand bird species, Fea et al. (2020) identified robins as one of only two species for which population responses were positively correlated to increasing intensity of mammal control. Predator control might be more effective if predator population recovery times could be reduced by more frequent control; or by reducing predator immigration to the key robin areas by increasing the spatial scale of aerial control around them, and through tactical use of barriers to predator movement (e.g. fencing); or by slowing the rate of *in situ* predator populations recovery through low-level complementary control (e.g. toxin bait stations or remote-reporting trapping networks). The final option of complementary control, could even make aerial 1080 control at intervals longer than the current six years sufficient to prevent decline in Dunedin SI robin populations.

Populations are limited by factors other than predation. For example, food availability affects the reproductive success of North Island robins: higher invertebrate biomass was associated with increased nest success, although it had no effect on the number of fledglings produced from each nest (Boulton et al. 2008). If food supply is low adults spend more time away from the nest foraging for food, increasing the risk of predation (Boulton et al. 2010). Competition for food between predators and forest birds may contribute to species' declines, although in most cases predation is the primary limiting factor and masks any effects of food limitation (Innes et al. 2010). Aerial 1080 control could be effective in depressing possum and rat numbers to levels that would allow robin populations to increase through increased nesting success if periodic control were carried out at a sufficient intensity over several years (Innes et al. 1999), if stoats were also controlled (O'Donnell & Hoare 2012), or if it were combined with additional management actions.

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