

Predation as a primary limiting
factor: A comparison of the effects of
three predator control regimes on
South Island robins (*Petroica
australis*) in Dunedin, NZ.

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ABSTRACT

The identification of factors limiting the recovery of threatened bird species is an area of significant research in New Zealand, where high levels of endemism make protection of threatened species extremely important. Predation by introduced mammals is often assumed to be the most important limiting factor for populations of threatened bird species, and a number of methods have been developed and implemented to deal with predators. Pest-management operations have a long history of success in NZ, but can also have unexpected consequences for non-target species. The three most common mainland pest-management measures are trapping, poisoning, and predator-exclusion fencing. My study used the South Island robin (*Petroica australis*) as a model to investigate the costs and benefits of three predator control operations over a period of six years at three independent sites in Dunedin, NZ: Silverstream, where rodent trapping occurs; Silver Peaks, the site of an aerially dispersed cereal-bait 1080 operation with pre-feed; and Orokonui, a predator-free sanctuary. Chew track cards were used to track changes in relative abundances of ship rats (*Rattus rattus*), mice (*Mus musculus*), and brushtail possums (*Trichosurus vulpecula*) at Silverstream and Silver Peaks over the period of 2011 to 2014. I monitored known robin pairs and single birds at all sites over the 2013/14 and 2014/15 summer breeding periods and combined this with previous monitoring data to track changes in annual adult robin survival and juvenile and adult robin recruitment rates, as well as robin nesting success. I also filmed nest sites at Silverstream to determine nest predators (nests were not filmed at Silver Peaks due to nests being too high to access). Trapping was effective in reducing ship rat relative abundance, although possum relative abundance increased in parallel. Poisoning resulted in significant initial decreases of all monitored species. However, this was short-lived, with abundances of all monitored robin predators exceeding pre-operational numbers within a year, this being an outcome that has been observed in previous studies. Orokonui displayed high values for all robin metrics except adult recruitment. Silver Peaks displayed comparatively low rates of adult survival, high rates of adult recruitment, moderate rates of juvenile recruitment, and low rates of nesting success. Silverstream displayed high rates of adult survival and recruitment, but low rates of juvenile recruitment and nesting success. No significant differences in adult survival were detected between sexes, and no significant differences in nesting success were detected between incubation and nestling stages. Predation, especially by stoats, was found to be the primary limiting factor affecting nesting success, although no other metrics were thought to be significantly affected by predation. Some evidence for masking of competition effects by predation is presented.

The results of this thesis provide key insights into the efficacy of management for South Island robins as well as knowledge of the effects and interactions of predation and competition on a native bird species. This will be useful in future research and management strategies, helping better tailor predator-control regimes to target problem species, enabling rapid recovery of valuable species and preservation of New Zealand's unique fauna and flora.

CONTENTS

| | |
|---|----|
| Acknowledgements..... | 1 |
| Abstract..... | 2 |
| Table guide | 6 |
| General Introduction | 7 |
| Historical context..... | 7 |
| Limiting factors..... | 7 |
| Study species..... | 8 |
| Competition for food..... | 8 |
| Habitat loss and Fragmentation..... | 9 |
| Genetic bottlenecks | 9 |
| Genetic rescue..... | 10 |
| Predation | 10 |
| Predator control operations | 11 |
| Trapping..... | 11 |
| Poisoning | 12 |
| Ecosanctuaries | 14 |
| Study areas..... | 14 |
| Orokonui | 14 |
| Silverstream | 15 |
| Silver Peaks..... | 15 |
| Aims and Predictions | 16 |
| Chapter guide | 17 |
| Chapter Two - Methods | 17 |
| Chapter Three - Results..... | 17 |
| Chapter Four – Discussion and Recommendations | 17 |
| Methods | 18 |
| Study species..... | 18 |
| Robin monitoring..... | 18 |
| Predator indices..... | 19 |
| Nest cameras..... | 19 |
| Mark-recapture data..... | 19 |
| Adult survival..... | 20 |
| Recruitment..... | 20 |

| | |
|--|-----------|
| Nesting success | 20 |
| Statistical analyses..... | 20 |
| Predator indices..... | 20 |
| Adult survival Pradel model | 21 |
| Juvenile recruitment Pradel model..... | 22 |
| Binomial recruitment GLM..... | 23 |
| Nesting success | 24 |
| Results | 25 |
| Predator indices..... | 25 |
| Ship rats | 25 |
| Possums | 26 |
| Mice..... | 27 |
| Nest cameras..... | 28 |
| Adult survival..... | 28 |
| Pradel juvenile recruitment..... | 29 |
| Binomial recruitment..... | 30 |
| Estimated Nest Survival | 32 |
| Nest stage function | 33 |
| Discussion | 35 |
| Predator relative abundance..... | 35 |
| Nest cameras..... | 36 |
| Robin population metrics | 37 |
| Orokonui..... | 37 |
| Silver Peaks..... | 39 |
| Silverstream | 42 |
| Summary results..... | 44 |
| Predator control regimes..... | 44 |
| Conclusions..... | 46 |
| Recommendations | 47 |
| Orokonui | 47 |
| Silver Peaks..... | 47 |
| Silverstream | 48 |
| References..... | 49 |
| Appendices | 53 |

Figure guide

- Figure 1: Chew track card (CTC) indices for ship rats (*Rattus rattus*) for Silverstream and Silver Peaks for the study period of August 2011 to September 2014. Rodent traps installed at Silverstream in December 2012 (blue marker). Aerial 1080 operation carried out at Silver Peaks in September 2011 (red marker)..... 25
- Figure 2: Chew track card (CTC) indices for brushtail possums (*Trichosurus vulpecula*) for Silverstream and Silver Peaks for the study period of August 2011 to September 2014. Rodent traps installed at Silverstream in December 2012 (blue marker). Aerial 1080 operation carried out at Silver Peaks in September 2011 (red marker)..... 26
- Figure 3: Chew track card (CTC) indices for mice (*Mus musculus*) for Silverstream and Silver Peaks for the study period of August 2011 to September 2014. Rodent traps installed in Silverstream in December 2012 (blue marker). Aerial 1080 operation carried out in Silver Peaks in September 2011 (red marker)..... 27
- Figure 4: Estimated adult survival (and associated 95% CIs) for SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2008/09 (2010/11 for Orokonui) to 2014/15 (2013/14 for Orokonui). $n_{\text{Orokonui}}=88$, $n_{\text{SP}}=68$, $n_{\text{SS}}=49$. No estimate was produced for 2008 for any site as survival estimates are calculated as a function of the previous year's presence/absence data, and there was no data for 2007). No estimates were produced for 2009 and 2014 for Orokonui as robins were only translocated in 2010 and not monitored in the 2014/15 breeding season..... 29
- Figure 5: Estimated juvenile recruitment (and associated 95% CIs) for SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2008/09 (2010/11 for Orokonui) to 2014/15 (2013/14 for Orokonui). $n_{\text{Orokonui}}=264$, $n_{\text{SP}}=80$, $n_{\text{SS}}=61$. No estimates were produced for 2008 as survival estimates are calculated as a function of the previous year's presence/absence data (no data for 2007). No estimates were produced for 2009 and 2014 for Orokonui as robins only translocated in 2010 and not monitored for 2014/15 breeding season. 30
- Figure 6: Recruitment estimates (and associated 95% CIs) for three sites (Orokonui, Silver Peaks, and Silverstream) for two age classes (Adults and juveniles) of SI robins. $n_{\text{adult}}=100$, $n_{\text{juvenile}}=234$. $n_{\text{Orokonui}}=135$, $n_{\text{SP}}=110$, $n_{\text{SS}}=89$ 32
- Figure 7: Estimated nest survival (and associated 95% CIs) for SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2009/10 to 2014/15 (2013/14 for Orokonui¹). ¹No estimates for 2014 for Orokonui as robins not monitored for 2014/15 breeding season. 33
- Figure 8: Estimated nest survival (and associated 95% CIs) of both incubation and nestling periods for SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2009/10 to 2014/15 (2013/14 for Orokonui¹). ¹No estimates for 2014 for Orokonui as robins not monitored for 2014/15 breeding season..... 34

TABLE GUIDE

| | |
|---|----|
| Table 1: Model permutations for RMark analysis of adult survival using the Pradel survival model for robin mark-recapture data with predictors 'Area' and 'Sex'..... | 21 |
| Table 2: Model permutations for RMark analysis of juvenile recruitment using the Pradel recruitment and survival model for robin mark-recapture data with predictor 'Area'. | 22 |
| Table 3: Model permutations for generalised linear modelling of juvenile recruitment using logistic regression analysis for robin recruitment data with predictors 'Area' and 'Year'..... | 23 |
| Table 4: RMark models for nesting success analysis of nesting data with variables 'area' and 'year'..... | 24 |
| Table 5: RMark models for nesting success analysis of nesting data with inclusion of nest stage variable. | 24 |
| Table 6: Outcomes observed from nest camera placement on SI robin nests at Silverstream for the 2014/15 breeding season. | 28 |
| Table 7: Model selection criteria for Pradel mark-recapture models of adult survival for SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2008/09 to 2015/15. Models are listed sequentially with model with the lowest AICc first. | 28 |
| Table 8: Model selection criteria for Pradel mark-recapture models of juvenile recruitment for SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2008/09 to 2014/15. Models are listed sequentially with the model with the lowest AICc first..... | 29 |
| Table 9: Analysis of variance and associated effect size (η^2) values for binomial generalised linear model of recruitment from model: Year+Area*Age Class. Factors are listed sequentially in order of addition..... | 30 |
| Table 10: Analysis of variance and associated effect size (η^2) values for binomial generalised linear model of recruitment from model: Area*Age Class. Factors are listed sequentially in order of addition..... | 31 |
| Table 11: Model selection criteria for binomial generalised linear model of recruitment for all permutations of model including the interaction Area*Age Class. Models are listed sequentially with the model with the lowest AICc first..... | 31 |
| Table 12: Coefficient estimates (and associated 95% CIs and p-values) of recruitment of SI robins for the binomial generalised linear model: Area*Age Class. | 31 |
| Table 13: Model selection criteria for nesting success mark-recapture models for SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2010/11 to 2014/15. Models are listed sequentially with the model with the lowest AICc first. | 32 |
| Table 14: Model selection criteria for nest survival mark-recapture models for SI robins during the incubation period at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2010/11 to 2014/15. Models are listed sequentially with the model with the lowest AICc first. | 33 |
| Table 15: Model selection criteria for stage-based nest survival mark-recapture models for SI robins during the nestling period at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2010/11 to 2014/15. Models are listed sequentially with the model with the lowest AICc first..... | 33 |

GENERAL INTRODUCTION

HISTORICAL CONTEXT

Globally, the conservation of threatened and near-threatened species is an area of increasing importance. With declines in species being observed in all major phyla, understanding the causes of these declines and how conservation efforts may remediate them is of significant interest to managers of threatened species. Factors that might limit a species' recovery are often complex in nature and may change drastically over time.

Understanding these factors is therefore no easy task, and often requires intensive research over an extended period. This is especially true in New Zealand, where declines in native and endemic species are ongoing and have been the focus of extensive research and management efforts.

Historically, three phases of decline and extinction of New Zealand avifauna have been described, the last of which is a result of European arrival and associated activities and introductions since the late 18th century (Holdaway, 1989). Europeans introduced a range of pest species including rodents, mustelids and brushtail possums (*Trichosurus vulpecula*) as well as a diverse range of avifauna (Thomson, 1922, Holdaway, 1989). In addition, there was large-scale landscape clearance and recreational hunting (Holdaway, 1989). The impacts were swift and devastating for many species and are still being felt today, as conservationists seek effective management systems for dealing with causes of population declines.

LIMITING FACTORS

Research into the conservation and management of birds in New Zealand has begun to focus on the identification and mitigation of 'limiting factors' (Innes et al., 2010). Limiting factors are those factors that inhibit a population's growth and recovery (Innes et al., 2010). Conservationists have begun to recognise that these factors might appear simple at first but can often have complex interactions, with populations of different species being subject to different limiting factors (Mackintosh and Briskie, 2005, Boulton et al., 2008, Innes et al., 2010). Additionally, this complexity can be seen at multiple scales, where several limiting factors can act on one population (large-scale, e.g. predation as well as deforestation), or where different sources of pressure form a single limiting factor (small-scale, e.g. multiple predator species interactions). Limiting factors can interact in ways that make them difficult to study and which can obscure causal relationships, especially where the effects of one limiting factor is masked or misattributed to another (Mackintosh and Briskie, 2005, Innes et al., 2010).

For New Zealand avifauna, several limiting factors have been identified, with the significance of each depending on the species, the attributes of the specific population (e.g. island vs. mainland populations), and even sex (Boessenkool et al., 2007, Innes et al., 2010). In general, it is thought that the New Zealand avifauna is limited by five main factors: predation, competition, habitat loss, disease and low genetic diversity (Innes et al., 2010). Effects of one factor can often be masked or misinterpreted as the effect of one or a number of other factors, and factors can be highly co-dependent (Mackintosh and Briskie, 2005, Boulton et al., 2008, Innes et al., 2010). The majority of research in New Zealand indicates that the primary factor limiting the recovery of native bird populations is predation by introduced mammals (Boulton et al., 2008, Innes et al., 2010, Starling-

Windhof et al., 2011). However, this is not always the case, and the key to managing these populations lies in identifying the limiting factors specific to the species and the population, and dealing with them on a case-by-case basis, as we are often unable to draw broad conclusions across different populations of species inhabiting different ecological systems (Boulton et al., 2008).

STUDY SPECIES

One species of significant interest in New Zealand is the South Island robin (Toutouwai, *Petroica australis*). The South Island robin comprises a number of sparsely distributed populations inhabiting the South Island of New Zealand (*Petroica australis australis*) as well as the sub-species, the Stewart Island robin (*Petroica australis rakiura*), inhabiting Stewart Island (Laws and Jamieson, 2011). There is also the closely related North Island robin (*Petroica longipes*) and the severely threatened Chatham Island robin (*Petroica traversi*). Ongoing studies have increased understanding of the factors limiting robin populations. The South Island robin (SI robin hereafter) is a small endemic passerine not currently listed as a threatened species under the Department of Conservation (DOC) Threat Classification System List (Hitchmough et al., 2007, Miskelly et al., 2008, Robertson et al., 2013), and is known to persist in areas where predators are common (Boulton et al., 2008). Currently, there exist healthy populations of SI robins dispersed throughout the western and northern regions of the South Island of New Zealand. On the east coast there remain only two small populations: one population near Kaikoura (Kowhai Bush) and one near Dunedin, although these populations are often fragmented and occupy areas of varying habitat quality (Schadewinkel and Jamieson, 2013b, Schadewinkel and Jamieson, 2013a). The SI robin has been translocated to near-pristine offshore Islands (e.g. Motuara Island in the Queen Charlotte Sound), providing researchers with opportunities to study different populations and the factors that influence them (Boessenkool et al., 2007, Jamieson, 2009, Heber et al., 2013). Predator control operations (e.g. poisoning, trapping and construction of predator-proof sanctuaries) have been carried out in or near robin habitats, providing excellent opportunities for experimental manipulations (Brown, 1997). Experimental manipulations have provided key insights into the factors limiting robin populations, and these factors will be discussed presently.

COMPETITION FOR FOOD

One factor potentially limiting SI robin populations is competition for food with introduced pest species, with several studies identifying a link between breeding success and food availability (Mackintosh and Briskie, 2005, Borkin et al., 2007, Boulton et al., 2010). Low availability of invertebrates influences nest-activity behaviour as robins must spend more time foraging (Boulton et al., 2010). Foraging and allofeeding (e.g. male feeding female mate) behaviours are thought to have the potential to increase the vulnerability of nests to predation, with avian species that share a native range with introduced predators displaying selection against increased nest activity (Boulton et al., 2010). Despite known nest behaviour effects, it has been difficult to identify food availability as the primary limiting factor for any population due to the difficulty in measuring effects, as well as the changing nutritional requirements of birds throughout their natural lifetime (Mackintosh and Briskie, 2005, Innes et al., 2010). Low food availability often results in emigration and reduced breeding attempts, and can be masked by predation where adult birds are forced to forage for extended periods of time, increasing vulnerability to predators (Boulton et al., 2010, Innes et al., 2010). Food supplementation has been shown to increase fecundity in the endemic Hihi (*Notiomystis cincta*) (Armstrong and Ewen, 2001, Castro et al., 2003) and there is evidence that SI

robins inhabiting the Chetwode Islands in the Marlborough sounds are affected by low food availability (after controlling for predation and migration), spending more time foraging and less time carrying out other behaviours such as mating and territorial defence (Powlesland, 1981). Food availability has likely played a significant role in historical declines, but in most cases predation pressure is at such intense levels as to prevent food availability from becoming a primary limiting factor (Innes et al., 2010).

HABITAT LOSS AND FRAGMENTATION

Habitat loss might also play a major role in hindering the recovery of robin populations in New Zealand. Land clearance has resulted in a significant reduction in the available habitat for avian species, affecting their distribution and abundance (Holdaway, 1989). Historically, habitat loss is recognised as one of the major drivers of the initial losses in avian biodiversity in New Zealand (Thompson, 1927) and the resultant fragmentation of habitat has played a significant role in the continual decline of extant species. Recent studies into the effects of habitat loss have focused on how fragmentation might limit nesting success (Boulton et al., 2008) and how robins respond when translocated into forest recovering from fragmentation (Armstrong and Ewen, 2002). In general, robins avoid open areas, with all New Zealand species showing extreme reluctance to cross open areas more than 100 metres wide (Richard and Armstrong, 2010). This behaviour can be a double-edged sword for managers, as while it ensures that robins do not attempt to leave near-shore island sanctuaries in early establishment stages, it also limits the connectivity of fragmented populations. A study aimed at assessing the effects of fragmentation on nesting success of a population of North Island robins found no significant relationship between the two (Boulton et al., 2010). It was suggested that there might be an effect where fragmented habitat promoted the success of Corvid predators, as had been observed elsewhere. However, the lack of an overlap in robin and introduced Corvid ranges, with rooks (*Corvus frugilegus*) inhabiting fringe habitat near farms and high country, meant that this effect was not evident in New Zealand (Boulton et al., 2010). Additionally, research into the ability of SI robins to respond to habitat fragmentation in an area of recovering forest has suggested that robin distribution can be affected by robin density and fragment size, but not by fragment connectivity (Armstrong and Ewen, 2002). Habitat type might also play a major role in robin distribution as the vegetation type can affect invertebrate abundance and species composition (Borkin et al., 2007). However, research indicates that robins are often better able to persist in introduced plantations, such as Monterey Pine (*Pinus radiata*), than in native manuka (*Leptospermum scoparium*) and kanuka (*Kunzea ericoides*), suggesting some forms of native forest are less suitable, though this is most likely due to differences in predation pressure than invertebrate abundance (Schadewinkel and Jamieson, 2013b, Schadewinkel and Jamieson, 2013a). Some studies have suggested that invertebrate abundances of introduced plantations is often no different to that of native forests (Pawson et al., 2008, Pawson et al., 2009).

GENETIC BOTTLENECKS

Recent research has begun to focus on genetic factors as major limiting factors for robin populations. Many papers have focused on the effects of inbreeding depression on population growth and vulnerability (Jamieson et al., 2006, Boessenkool et al., 2007, Hale and Briskie, 2007, Jamieson, 2009, Jamieson, 2010, Laws and Jamieson, 2011). In populations of robins where inbreeding occurs regularly and levels of homozygosity are high, hatching success can be significantly lower than in other, more outbred populations (Boessenkool et al., 2007). This is especially

significant in populations where breeding with closely related relatives is more likely to occur (Jamieson et al., 2009), populations that experience serial bottlenecks (Heber and Briskie, 2010), in populations that were founded by a small number of individuals (Heber et al., 2013), or in declining populations. Such conditions are known to occur in translocated populations of robins to island sanctuaries, where the population is founded by a small number of individuals from a single source population (Innes et al., 2010), or in captive-breeding programmes for severely threatened species (e.g. the black robin). The removal of omnivorous competitors/predators precludes predation and often food availability as limiting factors, which can result in rapid growth of population numbers, but the high level of homozygosity can become a significant limiting factor in the future (Innes et al., 2010). Studies carried out with the aim of determining the effects of food availability have often concluded that the disparities in hatching success observed between mainland and island populations of robins are most likely not a product of food availability as a limiting factor, but due to low genetic diversity in island populations (Mackintosh and Briskie, 2005). Additionally, research has suggested that populations subject to serial bottlenecks are rendered more vulnerable to disease, with highly inbred robin populations displaying decreased immunocompetence as compared to larger, more outbred populations (Armstrong and Ewen, 2002).

GENETIC RESCUE

In the case of robins, the primary limiting factors of a population depends on the location of that population. Avian species inhabiting predator-free island sanctuaries or mainland fenced sanctuaries are most often not subject to predation as a primary limiting factor, and, in the case of island populations, are often most inhibited by genetic factors associated with a small founding population, low or absent immigration, and the high level of inbreeding. In these populations, managers have employed a number of approaches to dealing with high levels of homozygosity. Two such techniques are the use of outbred individuals to increase heterozygosity, and the use of inbred donors to delete harmful alleles (Heber et al., 2013), these techniques being often referred to as 'genetic rescue'. Both these techniques rely heavily on pedigree information and inbreeding coefficient determination, and can therefore be used on only those populations that have been well studied. Results from studies of robins introduced to predator-free sanctuaries and offshore islands have been positive, with many of these populations displaying rapid population increases from a small number of founder robins, helping to establish healthy populations (Taylor et al., 2005, McGavin, 2009). The success of these sanctuaries in establishing populations from a small number of founders is often attributed to the ease of pair formation, the low post-release mortality, and the rapid population increases facilitated by low densities associated with the sanctuaries (Taylor et al., 2005).

PREDATION

A number of studies have identified predation by introduced mammals as having a significant impact on populations of birds in NZ (Brown et al., 1998, Boulton et al., 2008, Starling-Windhof et al., 2011), and this is generally held to be true for many avian species (Holdaway, 1989). Predation has been shown to decrease robin hatching success via predation at the egg stage, to decrease fledging success via predation at the nestling stage, to decrease juvenile recruitment via predation at the juvenile stage and to cause a biased operational sex ratio via predation of females on the nest (Armstrong and Ewen, 2002, Boulton et al., 2008, Starling-Windhof et al., 2011). Predation can also have secondary effects on populations e.g. reduced range in populations reliant upon juvenile survival for dispersal (Boulton et al., 2008). Additionally, these effects are more influential in robin

species because they possess life history traits that make them more vulnerable to predation effects (e.g. female-only incubation, ground-feeding) (Starling-Windhof et al., 2011). However, a number of recent studies have stressed the need for caution when interpreting the proximate predation effect as it might be acting in conjunction with other ultimate limiting factors (Boulton et al., 2008, Boulton et al., 2010, Innes et al., 2010). Situations have been identified where predation can act simultaneously with food availability, with identical demographic outcomes. Consequently, identification of the primary limiting factor is difficult to achieve (Innes et al., 2010). In cases where predation pressure is experimentally manipulated via predator control or eradication in New Zealand, the removal of the predator (most often the omnivorous ship rat *Rattus rattus* and brushtail possum *Trichosurus vulpecula*) is often likely to increase robin numbers through a reduction in both the direct predation of birds as well as a reduction in competition for food (Innes et al., 2010), resulting in increases in survival rates and recruitment. Alternatively, the control of a top predator (such as the stoat *Mustela erminea*) might act to increase numbers of other, secondary predators (such as the ship rat), such efforts often dependent upon the timing of the control procedure (Innes et al., 2010).

In response to predation-related declines of native species, conservation managers look for methods of controlling introduced predators with the aim of alleviating the pressure exerted upon threatened populations. In order to do this, the importance of identifying the limiting factors specific not just to the species but also to the specific population, as well as any sex-based biases is crucial. If conservationists fail to identify the primary limiting factor and whether successional limiting factors may come into play once one has been eliminated, subsequent conservation plans may fail as the true limiting factor has not been addressed (Innes et al., 2010). In essence, predator control methods aimed at population recovery attempt to eliminate or strongly reduce the influence of predation as a limiting factor. In New Zealand, a range of methods have been developed and implemented, the most common of these examples being trapping and poisoning. These methods, though often successful, can have unexpected consequences for ecosystems, especially where species interactions are unclear. This makes it important to consider each method carefully before application and to continue to monitor the progress of any pest-management operation after its initial implementation.

PREDATOR CONTROL OPERATIONS

In populations of robins where predators (often omnivores) coexist, predation is of great significance (Innes et al., 2010), and any control procedure must aim to reduce the influence of predation, most often by eliminating problematic species with the use of trapping, poisoning, exclusion programmes, or a combination of these techniques. Predator-proof sanctuaries not only provide a stronghold for reintroduction of threatened species, but can also provide researchers with an excellent control to test the effects of introduced predators on threatened species.

TRAPPING

Trapping involves the control of predators via lethal traps (Alterio et al., 1999). This method most often targets rodents, possums and mustelids, with traps being placed on or near the ground where these species are most likely to encounter them. A number of different traps have been developed for the eradication of rats and mustelids, with the next generation of traps being designed to self-reset. This means that traps can remain set and effective for extended periods of time without the

need for clearing or resetting/re-baiting (Campbell et al., 2015). These traps operate with a baited lure and a lethal gas-fired piston designed to pierce the brain of the target species (Blackie et al., 2014, Campbell et al., 2015). Research into these methods of trapping is ongoing, and might provide a cost-effective alternative to small-scale aerial toxicant eradication (Blackie et al., 2014). Current methods of trapping, such as single-catch mechanical traps, can prove effective in maintaining numbers of predators at allowable levels that will not impede the recovery of some populations (Innes et al., 2010). However, trapping can also be ineffectual where predators have become aware of traps, where predators are too abundant, or where servicing the trap lines becomes too costly (Bomford and O'Brien, 1995). Additionally, control operations targeting individual species, as trapping often does, can have consequential effects whereby the limitation of primary predators such as cats (*Felis catus*) and mustelids can result in mesopredator release of species such as rats and mice (Tompkins and Veltman, 2006, Ritchie and Johnson, 2009). However, research has suggested that competitor release, rather than mesopredator release, is a more significant result of control operations, and that control of primary predators might not affect mesopredators to a significant degree (Ruscoe et al., 2011). In these instances, poisoning operations are often favoured over trapping.

POISONING

Poisoning provides a cost-effective way of eliminating high proportions of introduced predators from areas where they have been inhibiting species' recovery (Veltman and Westbrooke, 2011). In New Zealand, the most commonly used poison for predator control is sodium fluoroacetate (Eason et al., 2011). Referred to as 1080, sodium fluoroacetate is a broad-spectrum poison that targets and inhibits the Krebs cycle of organisms that ingest it (Eason et al., 2011). It has been the poison of choice in a large number of successful operations carried out by the Department of Conservation of New Zealand and TBFreenZ (now OSPRI) aimed at aiding the recovery of species threatened by introduced predators (Veltman and Westbrooke, 2011). Despite this, some lobby groups remain unconvinced of the safety of 1080 with regard to the potential for contamination of soil and waterways, the humaneness of the poison in killing target species, and the potential for poisoning of non-target species. These doubts are often unfounded, as modern poisoning operations are meticulous in design and implementation, taking care to eliminate or limit cause for these concerns (Eason et al., 2011).

Research into the toxicology of sodium fluoroacetate at levels found in standard 1080 doses indicates that, under favourable conditions, 1080 can be rapidly defluorinated in 1-2 weeks, with defluorination time increasing as conditions become less favourable (i.e. in extreme cold and drought) (Eason et al., 2011). Therefore, the potential for contamination may depend on localised conditions. However, extensive monitoring of waterways after 1080-based poisoning operations indicates that the majority of associated waterways contain no traceable quantities of 1080 after exposure and in those that do, display levels that are not close enough to known LD₅₀ levels to be problematic (Eason et al., 2011). Additionally, a number of plant and microbial species present in New Zealand were identified as effective decontaminators of sodium fluoroacetate (Eason et al., 2011). This suggests the potential for contamination of waterways by 1080 is minimal, although water monitoring procedures are enforced in order to prevent human contamination (Eason et al., 2011).

The humaneness of 1080 is a subject of much debate. Many recognise its effectiveness as a predator control poison, with the majority of target species being killed humanely and within hours of exposure to the toxicant (Eason et al., 2011). However, the symptoms of 1080 vary amongst species, with some predators (such as dogs) displaying prolonged and seemingly traumatic deaths (Eason et al., 2011). In contrast, the relatively higher tolerance of 1080 by lizards and birds, as well as low levels of contamination of invertebrates, are advantageous as effects on non-target species are of extreme importance (Eason et al., 2011). Before any poisoning programme is initiated, the risk of non-target species mortality and contamination must be rigorously assessed (Eason et al., 2011). The death or contamination of non-target species, often those species that the operation is trying to safeguard, is of significant concern. The prevalence of this kind of mortality is dependent upon the species (Eason et al., 2011). Non-target mortality is particularly significant where pre-feeding operations, designed to increase bait acceptance of target species, are used prior to poison drops, as this can encourage bait acceptance by non-target species (Eason et al., 2011, Schadewinkel and Jamieson, 2013b, Schadewinkel and Jamieson, 2013a). Poison operations must therefore balance the effectiveness of the chosen poison in controlling pest species with the risks associated with it for non-target species (including humans) and contamination, the speed of death (in most target species), and the ease of application (as compared to poisons such as brodifacoum (Brown, 1997)). These are all factors in favour of 1080's use (Eason et al., 2011), although it is stressed that intensive monitoring should be carried out after any operation to ensure its effectiveness and to allow managers to respond to any abnormal effects (Eason et al., 2011).

Numerous 1080 operations have been carried out in areas known to be inhabited by robin populations (Schadewinkel and Jamieson, 2013b, Schadewinkel and Jamieson, 2013a) and research has indicated that robin mortality due to poison consumption varies with the application procedure and the population of interest. In one instance, in a population of North Island robins exposed to a 1080 operation, a 50% mortality rate was recorded (Powlesland et al., 1999, Eason et al., 2011). While this percentage seems alarming at first, ongoing monitoring revealed that the remaining robins responded well, with numbers recovering rapidly and the population reaching greater numbers than were observed prior to the operation after only one year (Powlesland et al., 1999, Eason et al., 2011). In contrast, monitoring of populations of SI robins inhabiting an area subject to aerial application of 1080 revealed zero mortalities associated with poisoning (Schadewinkel and Jamieson, 2013b, Schadewinkel and Jamieson, 2013a), highlighting the variation that occurs between populations and the importance of application procedures. Robins, are naturally inquisitive birds, and are known to respond to human presence, especially those birds associated with ongoing research and the use of hand-distributed mealworms (Eason et al., 2011, Veltman and Westbrooke, 2011). This association with humans and hand-distributed food can be problematic when 1080 is hand-scattered, as it can increase the likelihood of consumption of poisons by robins (Eason et al., 2011), although this practice is very rare. This is often one of the major reasons for favouring aerial application over other distribution methods (as well as the ease of application), as birds are less likely to treat aerial baits as potential food sources (Eason et al., 2011, Veltman and Westbrooke, 2011).

The potential for undesirable effects of 1080 operations is acknowledged, with users continuing to develop better techniques, baits and practices to mitigate unwanted effects while achieving an effective reduction in predators (Eason et al., 2011). Developments in bait effectiveness, such as the use of less chaffy cereal baits, better application techniques, and the use of deterrents are ongoing

and ensure side effects of poisoning operations are minimised (Eason et al., 2011). In addition to this, strict monitoring of application areas ensures any undesirable responses can be identified and controlled before significant damage is caused (Eason et al., 2011). It is important that the development of better poison practices continues, and that studies associated with the use of 1080 are conducted to ensure any ongoing effects are identified (Eason et al., 2011).

ECOSANCTUARIES

Ecosanctuaries are a relatively recent development in the fight against introduced pest species. Many ecosanctuaries have proven valuable in restoring and promoting the health of populations of a number of native species (Burns et al., 2012). The eradication and exclusion of pest species from an area, with maintenance of a pest-free or pest-reduced status allows successful reintroductions and recoveries of many threatened species. Ecosanctuaries also provide an excellent opportunity to study native species' interactions in the absence of introduced pests, providing insight into a pest-free scenario and giving conservationists a target to aim for in any pest control operation. However, ecosanctuaries have been criticised as being highly costly to establish and run, with considerable maintenance costs (Scofield et al., 2011). Critics of ecosanctuaries point to the low species diversity that can occur in sanctuaries, the potential for sanctuaries to limit evolutionary potential, and the high cost-benefit ratio of ecosanctuary goals (Scofield et al., 2011). Often sanctuaries will require a number of translocations be carried out with ample time to allow settlement and establishing of a standing population of a desired species (Schadewinkel, 2013) which can be costly, this applies to both newly established as well as long-standing sanctuaries. However, evidence from research suggests that the absence of the influence of introduced pests, not just as predators but also as competitors, allows rapid growth of populations of protected species which could not be achieved outside of fenced areas (Burns et al., 2012, Schadewinkel, 2013). Additionally, ecosanctuaries provide potential for spill-over into surrounding habitats adjacent to the sanctuary (Russell et al., 2015). Spill-over allows the spread to and colonisation of areas outside the sanctuary, helping to promote the establishment of populations outside the protected area (Glen et al., 2013). This is beneficial to both the growth of the population as well as promoting the restoration of natural community assemblages.

STUDY AREAS

This research follows previous research conducted by the Department of Zoology at the University of Otago in three areas in close proximity to the city of Dunedin; Orokonui, Silverstream and Silver Peaks. Previous studies have monitored the effectiveness of three different predator control programmes implemented at these sites and their effects on the local populations of SI robins (Schadewinkel, 2013, Schadewinkel and Jamieson, 2013b, Schadewinkel and Jamieson, 2013a).

OROKONU

Orokonui Ecosanctuary is a 307ha predator-proof ecosanctuary with a mixture of eucalypt (*Eucalyptus regnans*) and regenerating native kanuka forest north of Dunedin. Established in 2006 with the construction of an 8.7km predator-proof fence, the sanctuary was declared pest-free in 2007 after eradication of goats (*Capra hircus*) and possums via ground-based teams combined with an aerial drop of brodifacoum. The sanctuary is the site of ongoing monitoring and preventative trapping, with populations of many threatened native and endemic birds, fish and reptiles. Robin monitoring began here in 2010 after the translocation of 25 adult robins from Silver Peaks and

Flagstaff near Dunedin city, with a follow-up translocation of 20 juveniles from Silver Peaks to reinforce numbers the next year. There has been successful breeding in the years since, with high nesting success and very few adults leaving the area (Schadewinkel et al., 2014).

SILVERSTREAM

The Silverstream study site is a 120ha kanuka-dominated area north-west of Dunedin. The study area supports a populations of ship rats, mice (*Mus musculus*), brushtail possums, stoats (*Mustela erminea*) and populations of native and introduced birds, including SI robins, and is surrounded by larger plots of unstudied habitat of similar quality (Schadewinkel and Jamieson, 2013b). Monitoring began at Silverstream in 2007, with trapping of rodents being carried out since December 2012 using 50 Goodnature™ A24 traps. Traps were deployed 50m apart in 3 separate lines in order to cover the maximum area of known robin territories. Traps were active between August and February (the breeding season of robins) but remained deployed year round, with gas replacement once per season and bait checks at least twice per season. Rodent detection declined significantly since the deployment of the traps (Schadewinkel and Jamieson, 2013a). However, possum detection has increased and nestling mortality of robins remains high, with the last-studied season (2012/13) indicating no successful nests (Schadewinkel and Jamieson, 2013a). It is thought that high levels of predation pressure and nests being close to the ground, and thus relatively accessible to even the least arboreal predator, is significantly hindering this population. It is now thought that possums are the primary predators of nests at this site, or that they may have occupied this position since the decline in detection rates of rodents in the area (Schadewinkel and Jamieson, 2013a). However, the nature of predator interactions is highly complex, and this requires further investigation.

SILVER PEAKS

The Silver Peaks study site is a 100ha Douglas fir (*Pseudotsuga menziesii*) and Monterey pine plantation north of Dunedin. The study site supports populations of ship rats, mice and possums as well as native birds, including SI robins (Schadewinkel and Jamieson, 2013a). Like Silverstream, the study area is surrounded by larger plots of similar but unstudied habitat. Monitoring began at Silver Peaks in 2009, with a cereal-based 1080 poisoning operation with pre-feed being carried out in September of 2011 over the whole of the plantation (5100ha total, including my study site) targeting brushtail possums (on a seven-year rotation). Initial mortality of possums (as well as rodents) was high, but this did not translate to increases in robin nesting success (Schadewinkel and Jamieson, 2013a). Additionally, detection rates of predators have increased to levels greater than those before the operation, while estimated nest survival is decreasing (Schadewinkel and Jamieson, 2013a). This is thought to be due to the ability of these populations of invasive predators to recover, with ship rats thought to be the primary predator at nests at this site (Schadewinkel and Jamieson, 2013a). However, estimated nest survival at this site is not significantly different from that in Orokonui Ecosanctuary and remains higher than that of Silverstream. This is most likely due to the construction of nests higher in trees (due to the tall nature of Douglas firs), making them inaccessible to the majority of predators (Schadewinkel and Jamieson, 2013a).

AIMS AND PREDICTIONS

This research builds on previous research by Robert Schadewinkel (2013) and Graham Parker (2013), with the continuation of nest survival analysis and predator relative abundance monitoring, and the addition of adult survival analysis, juvenile and adult recruitment analysis, and the determination of the species responsible for nest predation.

This research aims to investigate the long-term effectiveness of three different methods of predator control for the three sub-populations of SI robins in the Dunedin area, and whether predation is acting as a limiting factor on these sub-populations. This aim will be achieved by tracking changes in predator detection rates at each site over a two year period and by monitoring subsequent robin survival, recruitment and nesting success, these being known from previous studies to be affected by predation. This will then be combined with data from previous research to assess the effectiveness of the predator-control operations over a seven year period (2008/09 to 2014/15). By assessing the efficacy of these predator control methods, I will determine whether predation is acting as a limiting factor to populations of robins in these areas. Additionally, this study aims to identify the primary nest predators at Silverstream in order to assess whether current predictions from predator monitoring are reliable, with the goal of increasing the effectiveness of the operation.

If predation is acting as a limiting factor to populations of SI robins in the Dunedin area I predict the following:

- 1) In Orokonui, I expect to observe the highest rates of all population metrics as predation has been almost completely eliminated.
- 2) Relatively lower rates of robin nesting success, adult robin survival, and juvenile and adult robin recruitment at Silver Peaks due to increased predation rates as a result of increased predator relative abundance following post-1080 eruptions of all three monitored species in the year following the operation.
- 3) Relatively lower rates of robin nesting success, adult robin survival, and juvenile and adult robin recruitment at Silverstream as a response to increased predation rates by possums following increases in possum relative abundance. Possums will occupy the role as primary predator of robins following the control of rats at Silverstream.
- 4) Lower adult survival of female robins in Silver Peaks and Silverstream due to the female-only nesting strategy of robins.
- 5) Higher rates of nest survival during the incubation stage as compared to the nestling stage at Silver Peaks and Silverstream due to increased conspicuousness of nests to predators as a result of increased nest activity and begging behaviours associated with nestlings.

CHAPTER GUIDE

CHAPTER TWO - METHODS

This chapter will introduce the methods of the study including an overview of the breeding behaviour of the SI robin, the survey methods for monitoring SI robins, and the methods for estimating predator relative abundance. Chapter Two will also explain the data used for each analytical method and which statistical methods were applied to these data (including explanations of model structure).

CHAPTER THREE - RESULTS

Chapter Three will present the results of the statistical analyses of robin data, the relative abundance estimates for predators, and the results of the nest camera placement experiment. Information presented will include point estimates, uncertainty values and model selection criteria for all analyses, as well as all relevant graphs and tables associated with each statistical method.

CHAPTER FOUR – DISCUSSION AND RECOMMENDATIONS

Chapter Four is a discussion of the results in relation to each study site and each year of the study. Chapter Four will also include an explanation of the results in the context of the method of predator control implemented at each site, concluding with recommendations for future monitoring and management efforts.

METHODS

STUDY SPECIES

The SI robin breeding season extends from early September to late February, with clutches of one to three eggs (Robertson and Heather, 2005). Eggs hatch after about 18 days with chicks fledging after a further 21 days (Robertson and Heather, 2005). SI robins are territorial and highly monogamous (Schadewinkel and Jamieson, 2013a), allowing the establishment of known territory maps and pair combinations. Predation of eggs, chicks and adults is known to occur (Schadewinkel et al., 2014). The naturally inquisitive nature of robins facilitates monitoring. Meal worms (*Tenebrio molitor*) can be used to facilitate the location of nest sites and fledglings, because males feed incubating females and both parents feed nestlings/fledglings. Although not listed as threatened under the Department of Conservation's Threat Classification System (Hitchmough et al., 2007, Miskelly et al., 2008, Robertson et al., 2013) due to healthy populations on the west coast of the South Island, there remain only two isolated and threatened populations on the east coast: Kowhai Bush in Kaikoura, and the Dunedin populations, which include Silverstream, Silver Peaks and Orokonui (Schadewinkel and Jamieson, 2013a).

ROBIN MONITORING

Robins at these sites have been monitored for a number of years and are marked with coloured and metal tarsus bands which allow individual identification through unique combinations, and allowing repeated monitoring of each separate site. Data in this study were collected over a period of two years from September 2013 to February 2015, and was combined with previous monitoring efforts from 2008 to 2012 ((Parker, 2013, Schadewinkel, 2013)). Initial monitoring occurred in early October to determine pair combinations and record re-sightings, as well as territory size and position. All unbanded birds were caught using box or clap traps and banded and sexed. Territory positions and corresponding bird IDs were recorded using Garmin GPSMAP[®] 62s units (Garmin Ltd.), enabling the construction of territory maps. Once pair combinations and territory maps were established, the focus of monitoring was to determine nesting success of identified pairs and the status of single birds. Through the use of mealworms, I was able to determine the breeding status of birds at four distinct stages: courtship feeding, incubation, nestling, and fledgling. At the courtship feeding stage, the male collects meal worms and feeds the female, with both birds present at all times. While this indicates the presence of a pair, no nest has yet been built. The incubation stage is indicated by the male collecting mealworms and feeding the female by calling her, with only the male present initially. This indicates the female is incubating, and by following her post-feeding I was able to locate the nest. At the nestling stage, both the female and male are feeding nestlings in the nest, and so both collect mealworms, returning frequently to the nest. This allows the location of nests that were not located at the incubation stage, and the age of nestlings can be estimated by the stage of pin feather development of the nestlings' wings. At the fledgling stage, both the male and female (and at later stages, the fledglings) collect mealworms, with adults feeding fledglings. These stages can occur simultaneously if nests have failed or re-nesting has occurred, and some transitional stages such as nest building can also occur. Nest site characteristics were recorded and linked to the corresponding parent pair number, including GPS coordinates, dominant vegetation type, nest height, nest tree species, gradient and aspect, and nest stage. Nests were marked with flagging tape to enable repeated monitoring. Nests were checked three times a week, with a maximum of two

nests being monitored per pair for the breeding season as per previous monitoring (Schadewinkel and Jamieson, 2013a). Nests were considered successful when at least one nestling successfully fledged.

PREDATOR INDICES

Predator relative abundance data were obtained using Connovation Ltd. chew tracking cards (hereafter referred to as CTCs) with FeraFeed™ 213 peanut-based special blend bait. Detection of rats, mice and possums is facilitated through bite marks left in the card, and rat and possum indices for CTCs are known to be correlated to other predator indices such as tracking tunnels (Sweetapple and Nugent, 2011). At each site, 71 cards were placed along a total of 3.5km of transect with stations at 50m intervals, and one card placed per station. Transects were selected to coincide with known robin territories. Cards were folded in half and nailed to selected trees approximately 30cm vertically on the tree trunk as per manufacturer recommendations, and were collected after 10 days (previous research indicates bite mark saturation occurs within 7 days (Sweetapple and Nugent, 2011)). Species responsible for bite marks were identified in the field for each card as they were collected, with cards being re-checked in the lab. CTC operations were carried out once every three months starting in March 2013, with subsequent cards being deployed within 10m of the previous location. CTC bite mark data for the period of March 2013 to September 2014 were combined with previous CTC data from August 2011 to December 2012 (Schadewinkel, 2013) in order to track the activity of predators at Silver Peaks and Silverstream over the entire period of the study. The percentage of CTCs indicating bite marks is known to be positively correlated to the relative abundance of the respective species (Sweetapple and Nugent, 2011).

NEST CAMERAS

A small-scale study was run parallel to monitoring, with automatic cameras placed at nests to identify the primary nest predators at Silverstream. Cameras were not placed at Silver Peaks as nests were too high to access. Four Little Acorn™ trail cams, three regular and one high-definition, were placed on those nests that were easiest to access in order to enable simple maintenance of deployed cameras. Where possible, cameras were placed on neighbouring trees rather than on the nest tree itself. This was to reduce the likelihood of cameras alerting predators to nest locations. If a camera was placed on the nest tree, it was done in a manner that would reduce the conspicuousness of the camera. Where this was not possible, cameras were secured to wooden poles (most often fallen kanuka branches) and erected adjacent to the nest. Cameras were checked weekly for battery life, available memory, and to address any unforeseen problems (such as cameras becoming water-logged) prior to predation or fledging. Cameras were retrieved once the nest was empty, whether due to predation, abandonment or fledging. Video data were analysed manually using the VLC media player program.

MARK-RECAPTURE DATA

I collated mark-recapture data of nesting success and adult survival from the 2014/15 season with data for breeding seasons from 2009/10 to 2013/14 at Silver Peaks and Silverstream, and from 2010/11 to 2012/13 at Orokonui. Data were obtained as annual capture history records of individually banded robins, separated by area cohort, age class at banding, current age (including 'deceased') and sex. All kin data and breeding success data were recorded for all robins, with breeding success data being separated by breeding stage (nest building, eggs, chicks and fledglings).

ADULT SURVIVAL

I estimated adult survival using capture history records for SI robins, unsexed birds were excluded from the data as I wished to assess the presence of sex-biased mortality. Survival here is comprised of any instance of recapture or failure to recapture of any bird known to have been seen in the previous year. Capture history data for survival includes birds that originated from within the area (annual survival and mortality) as well as birds that arrived from adjacent areas (immigration) or that moved to adjacent areas (emigration). I combined capture histories with information on sex and area of origin for each individual bird for analysis of potential sex-based and area-based differences in adult survival across the study period.

RECRUITMENT

For recruitment, I produced two separate data sets, one for RMark analysis of juvenile recruitment and one for binomial generalised linear modelling of adult and juvenile recruitment. These two methods differ in that RMark analysis tracks annual juvenile recruitment as a proportion of the total population (allowing recruitment rates that exceed 100%) while binomial modelling gives an absolute recruitment estimate (i.e. what percentage of juveniles were recruited annually out of the total produced). Recruitment here is defined as any instance of a bird settling in a given area, and is comprised of both recruitment of juveniles from within the area (settlement) and recruitment of adults or juveniles from adjacent areas (immigration).

For RMark analysis, I collated capture history records of juvenile birds with area of origin data to allow analysis of area-based differences in juvenile recruitment across the study period. All birds discovered as adults or that entered an area as adults were excluded from this analysis.

For generalised linear modelling, I combined data for year of origin, area of origin and age class at recruitment with the binary recruitment variable (recruited or not recruited) to allow year-based, area-based and age-based binomial modelling of potential differences in recruitment across the study period.

NESTING SUCCESS

I recorded the day each nest was first found, the day it was last present, the day it was last checked, the fate of the nest (successful/unsuccessful), the age at date of discovery of the nest, and the site, year, pair number, clutch number and nest stage. Date-based variables (first found, last present, last checked and age at discovery) were standardised relative to the first day of the monitoring season. In the case of pairs having more than one nest, each nest was recorded individually with up to two nests being recorded.

STATISTICAL ANALYSES

PREDATOR INDICES

I calculated point estimates and associated confidence intervals for CTC bite mark data using Wald's adjusted method as per previous monitoring (Schadewinkel, 2013). Predator relative abundance point estimates and associated confidence intervals were calculated for each site and for each predator (ship rat, mouse and possum) for all CTC operations over the entire period of the study (equation 1).

I analysed nest camera data to calculate the proportional incidence of predation by different predators (stoats, ship rats and possums) at the different stages (incubation and chicks) of nests. This was calculated by taking the number of predation events attributable to any given predator and dividing over the total number of recorded nest failures. Data from all cameras found to reveal nothing where the nest had failed were also included.

$$\theta = \frac{(\hat{\theta} - \theta_0)^2}{se(\hat{\theta})}$$

Equation 1: Basis of Wald's adjusted method approximated to normal data whereby θ represents the estimate of the true abundance value, $\hat{\theta}$ represents a maximum likelihood estimate of the abundance parameter, θ_0 represents the null value to be compared to the MLE, and $se(\hat{\theta})$ represents the standard error of the estimate.

ADULT SURVIVAL PRADEL MODEL

I analysed adult survival data using the Rmark package (Laake et al., 2012). I produced five models, with an intercept-only model including only year (year was included in all RMark models presented) as a null model. I produced separate models including area of origin and sex, with an additive model for the inclusion of both area of origin and sex. I also produced a full interaction model with predictors for area of origin and sex, and the interaction term of area of origin and sex (Table 1). Akaike's corrected information criterion (AICc) was used for model selection, with those models with a difference in AICc of less than two being considered equal (Johnson and Omland, 2004).

Table 1: Model permutations for RMark analysis of adult survival using the Pradel survival model for robin mark-recapture data with predictors 'Area' and 'Sex'.

| Model | Parameters |
|----------------|-----------------------------------|
| Null | Intercept |
| Main effects 1 | Intercept + Sex |
| Main effects 2 | Intercept + Area |
| Additive | Intercept + Sex + Area |
| Interaction | Intercept + Sex + Area + Sex*Area |

The model gives the probability of adult survival based on the percentage of adults seen in the previous year that were resighted in the year of interest (eq. 2) and identifies differences based on area and sex.

$$\begin{aligned}
N_{i+1} &= N_i \varphi_i + B_i \\
\lambda_i &= N_{i+1} / N_i \\
\lambda_i &= \frac{B_i}{N_i} + \varphi_i \\
\lambda_i &= f_i + \varphi_i \\
\varphi_i &= f_i - \lambda_i
\end{aligned}$$

Equation 2: Basis of the adult survival model whereby λ_i is the annual population growth rate derived from the ratio of successive population sizes, N_{i+1} is the number of birds that survived in year $i+1$, N_i is the number from the previous year, φ_i is the probability of survival, B_i is the number of individuals entering the population between i and $i+1$ and f_i is the per capita rate of additions (Cooch and White, 2015).

JUVENILE RECRUITMENT PRADEL MODEL

I carried out mark-recapture analysis of juvenile recruitment using the RMark package (Laake et al., 2012). I produced two models, with an intercept-only model including only year as a null model. This was compared to a main effects model including area of origin (Table 2). Akaike's corrected information criterion (AICc) was used for model selection, with those models with a difference in AICc of less than two being considered equal (Johnson and Omland, 2004).

The model gives juvenile recruitment as a proportion of a given year's total population, rather than as a proportion of that year's juveniles (equation 3). This allows tracking of the rate of change of juvenile recruitment in the population and identifies differences based on area. The per capita rate of additions (f_i) is often mislabelled as recruitment where population demographics such as age class are not considered (Cooch and White, 2015). In the case of juvenile recruitment, however, because the dataset was limited to those birds that entered as juveniles, this was avoided.

Table 2: Model permutations for RMark analysis of juvenile recruitment using the Pradel recruitment and survival model for robin mark-recapture data with predictor 'Area'.

| Model | Parameters |
|--------------|------------------|
| Null | Intercept |
| Main effects | Intercept + Area |

$$\begin{aligned}
N_{i+1} &= N_i \varphi_i + B_i \\
\lambda_i &= N_{i+1} / N_i \\
\lambda_i &= \frac{B_i}{N_i} + \varphi_i \\
\lambda_i &= f_i + \varphi_i \\
f_i &= \lambda_i - \varphi_i
\end{aligned}$$

Equation 3: Basis of the juvenile recruitment model whereby λ_i is the ratio of successive population sizes, N_{i+1} is the number of birds that survived in year $i+1$, N_i is the number from the previous year, ϕ_i is the probability of survival, B_i is the number of individuals entering the population between i and $i+1$ and f_i is the per capita rate of additions (Cooch and White, 2015).

BINOMIAL RECRUITMENT GLM

Binomial generalised linear modelling of recruitment was carried out using the lme4 package (Bates et al., 2012). Five models were produced, with an intercept-only model as a null model. Models were also produced for area of origin, year, and age class at recruitment, and a full interaction model was produced with predictors for area of origin and age class, as well as the interaction term of area of origin with age class (Table 3). Akaike's corrected information criterion (AICc) was used for model selection, with those models with a difference in AICc of less than two being considered equal (Johnson and Omland, 2004).

Model outputs give the probability of a given bird being recruited into the population based on differences in area, age class and year. Juvenile recruitment here is given as a proportion of the total juveniles in any given year at any given site, not as a proportion of the population (as in RMark analysis). This allows tracking of the absolute rate of recruitment, rather than the rate of change of recruitment in the population. Adult recruitment is comprised of both those birds that entered the study area from adjacent unstudied areas, and those birds that were missed as juveniles the previous year but remained in the study area.

Table 3: Model permutations for generalised linear modelling of juvenile recruitment using logistic regression analysis for robin recruitment data with predictors 'Area' and 'Year'.

| Model | Parameters |
|----------------|---|
| Null | Intercept |
| Main effects 1 | Intercept + Area |
| Main effects 2 | Intercept + Year |
| Main effects 3 | Intercept + Age class |
| Interaction | Intercept + Age class + Area + Age class*Area |

$$\text{logit}\pi_{ij} = \pi + \text{area}_i + \text{year}_j + \text{area} : \text{year}_{ij}$$

Equation 4: Basis of juvenile recruitment logistic regression model where π_{ij} is the probability of recruitment of an individual as predicted by 'area', 'year' and the interaction term 'area:year'. Whereby i has 2/3 levels (Silver Peaks, Silverstream and Orokonui¹) and j has 6/7 levels (2008/09 or 2009/10¹ to 2014/15).

¹This represents the model including data from Orokonui

NESTING SUCCESS

Nesting success analysis was carried out using the RMark package (Laake et al., 2012). Three models were produced, with models for year and area of origin, and an interaction model of year with area of origin (Table 4). Three additional models were then produced from the original models for inclusion of the nest stage variable (Table 5). Akaike's corrected information criterion (AICc) was used for model selection, with those models with a difference in AICc of less than two being considered equal (Johnson and Omland, 2004).

Table 4: RMark models for nesting success analysis of nesting data with variables 'area' and 'year'.

| Model | Parameters |
|----------------|-------------------------------------|
| Main effects 1 | Intercept + Year |
| Main effects 2 | Intercept + Area |
| Interaction | Intercept + Area + Year + Area*Year |

Table 5: RMark models for nesting success analysis of nesting data with inclusion of nest stage variable.

| Model | Parameters |
|--------------|---|
| Year | Intercept + Year + Nesting stage |
| Area | Intercept + Area + Nesting stage |
| Area by Year | Intercept + Area + Year + Area*Year + Nesting stage |

Model outputs give daily survival rate (DSR) values (and associated 95% CIs) for each variable of interest which were then transformed to allow calculation of estimated nest survival. Transformation for the Site by Year model was carried out by taking the DSR to the power of 39. This gives the estimated nest survival over the 39 day period of an SI robin nest (Schadewinkel et al., 2014).

RESULTS

PREDATOR INDICES

SHIP RATS

Chew track card (CTC) results indicated higher rat activity in Silverstream between August 2011 and September 2012 as compared to Silver Peaks, with a sharp decline from intermediate levels in December 2012. During this period, rat relative abundance increased at Silver Peaks, and eventually exceeded those of Silverstream in December 2012 (Fig. 1). After December 2012, rat relative abundance values remained low but fluctuated greatly in Silverstream. Rat relative abundance in Silver Peaks increased after May 2012 but fluctuated at intermediate levels (Fig. 1).

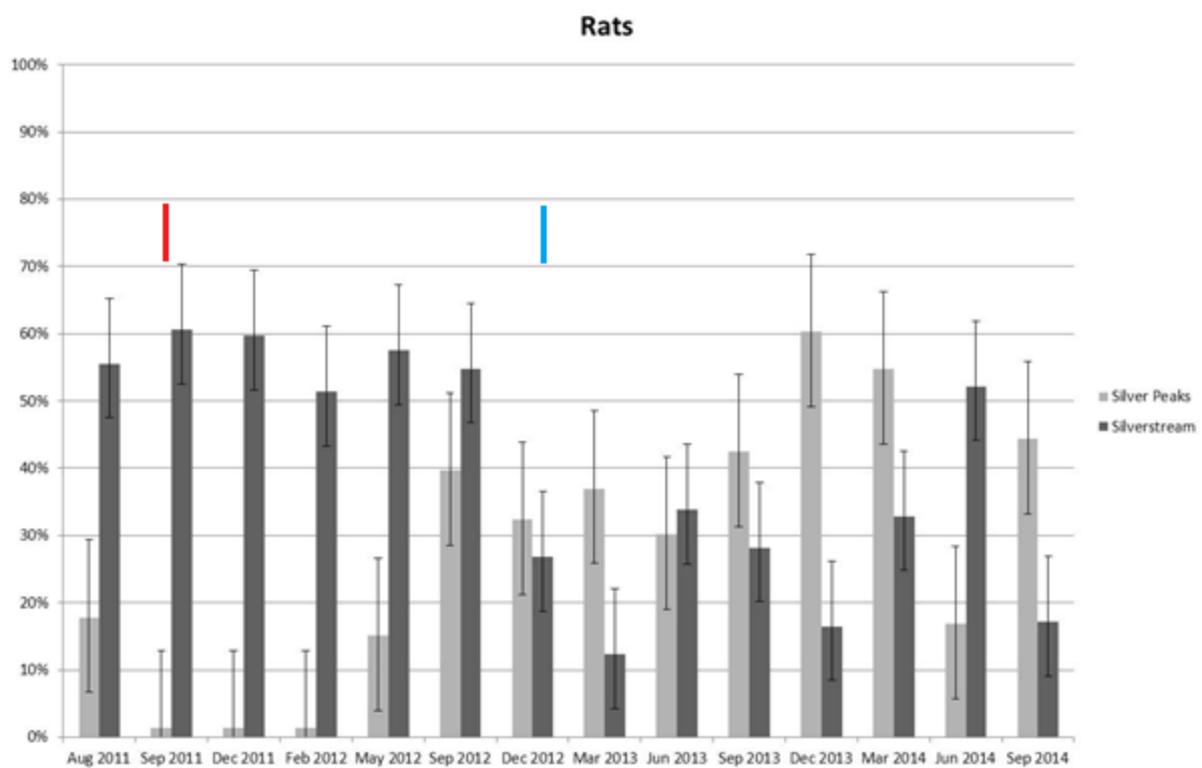


Figure 1: Chew track card (CTC) indices for ship rats (*Rattus rattus*) for Silverstream and Silver Peaks for the study period of August 2011 to September 2014. Rodent traps installed at Silverstream in December 2012 (blue marker). Aerial 1080 operation carried out at Silver Peaks in September 2011 (red marker).

POSSUMS

CTC results indicated an initial high relative abundance of possums in Silverstream, followed by a gradual increase after December of 2012 (Fig. 2). Silver Peaks displayed lower possum relative abundance as compared to Silverstream, with this estimate increasing gradually after December of 2012, but remaining lower than possum relative abundance in Silverstream (Fig. 2).

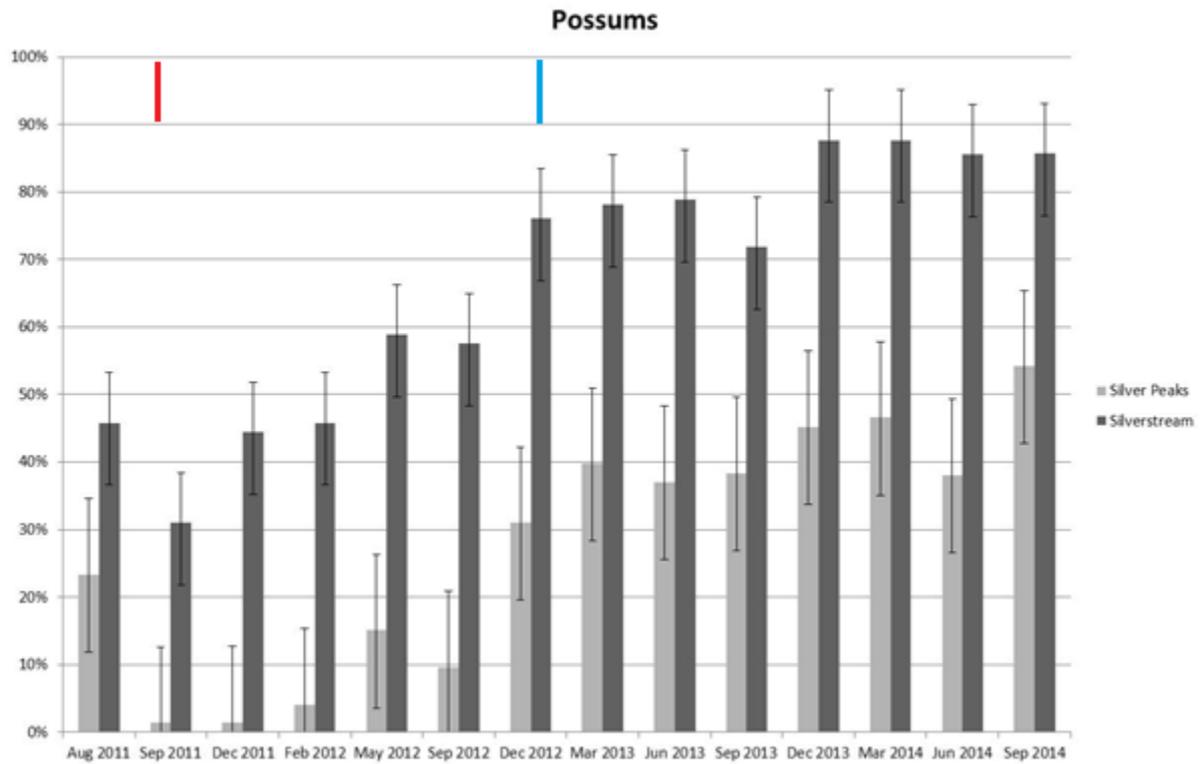


Figure 2: Chew track card (CTC) indices for brushtail possums (*Trichosurus vulpecula*) for Silverstream and Silver Peaks for the study period of August 2011 to September 2014. Rodent traps installed at Silverstream in December 2012 (blue marker). Aerial 1080 operation carried out at Silver Peaks in September 2011 (red marker).

MICE

CTC results indicated intermediate relative abundance of mice in Silverstream, with estimates fluctuating between low and intermediate levels across the entire period of August 2011 to September 2014 (Fig. 3). In Silver Peaks, relative abundance of mice was initially low but increased sharply in February of 2012, with gradual declines and moderate fluctuations thereafter (Fig. 3).

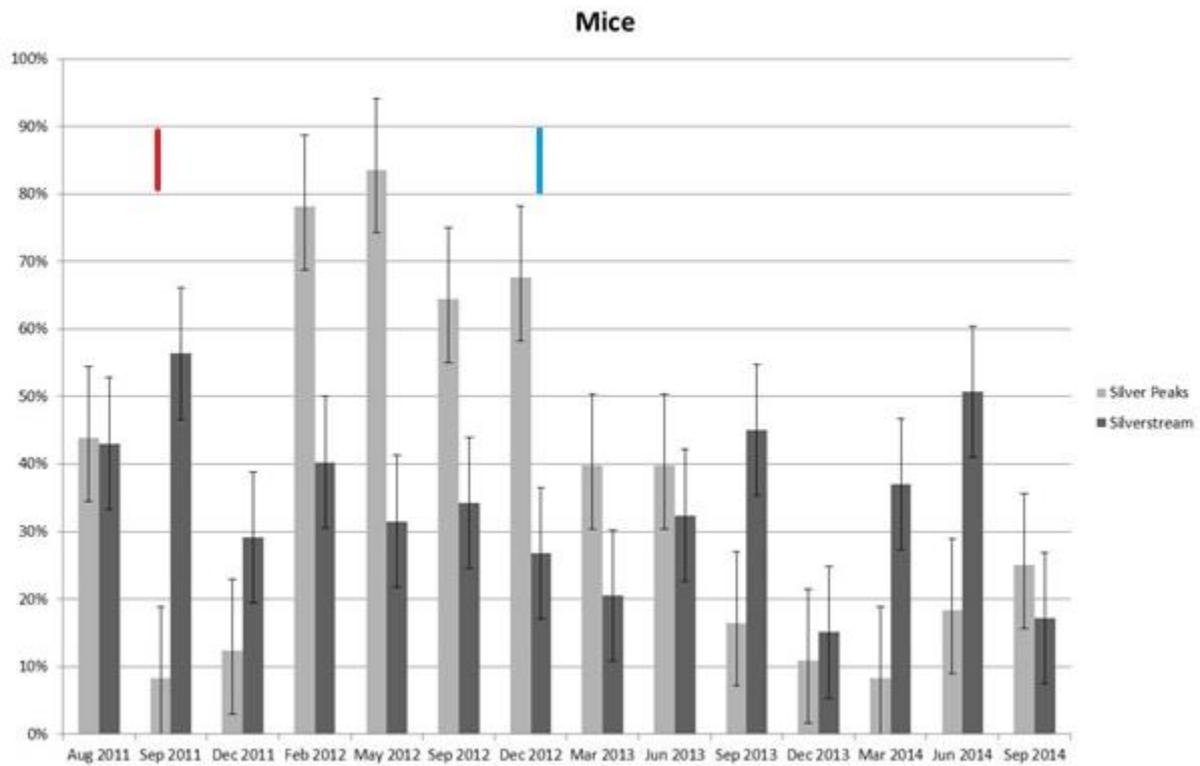


Figure 3: Chew track card (CTC) indices for mice (*Mus musculus*) for Silverstream and Silver Peaks for the study period of August 2011 to September 2014. Rodent traps installed in Silverstream in December 2012 (blue marker). Aerial 1080 operation carried out in Silver Peaks in September 2011 (red marker).

NEST CAMERAS

Cameras at Silverstream revealed a high prevalence of stoat predation of nests at both incubation and nestling stages, and possum predation at the incubation stage. Ship rats had a comparatively low impact on nests at both incubation and nestling stages (Table 6).

Table 6: Outcomes observed from nest camera placement on SI robin nests at Silverstream for the 2014/15 breeding season.

| Predator | Stage | Frequency | Percent of total (%) |
|-----------------|--------|-----------|----------------------|
| Stoat | | | |
| | Eggs | 2 | 13 |
| | Chicks | 5 | 33 |
| Ship rat | | | |
| | Eggs | 1 | 7 |
| | Chicks | 0 | - |
| Possum | | | |
| | Eggs | 2 | 13 |
| | Chicks | 0 | - |
| Unknown | | | |
| | Eggs | 0 | - |
| | Chicks | 4 | 27 |
| Other | | | |
| Successful | | 1 | 7 |
| Total | | 15 | |

ADULT SURVIVAL

Pradel analysis of adult survival revealed the best model for estimating adult survival of SI robins contained the factors Year and Area. Sex was not included in the best-fit model (Table 7).

Table 7: Model selection criteria for Pradel mark-recapture models of adult survival for SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2008/09 to 2015/15. Models are listed sequentially with model with the lowest AICc first.

| Model | Factors | AICc | Δ AICc | Deviance |
|--------------|------------------------|----------|---------------|----------|
| Area | Year+Area | 1427.104 | 0 | 390.116 |
| Full effects | Year+Area+Sex | 1430.466 | 3.362 | 479.626 |
| Interaction | Year+Area+Sex+Area:Sex | 1437.687 | 10.583 | 478.447 |
| Null | Year | 1468.844 | 41.740 | 239.713 |
| Sex | Year+Sex | 1470.653 | 43.549 | 297.288 |

There was an increase in adult survival of SI robins at all three sites during the study period between the breeding seasons of 2008/09 to 2014/15. Robins at Orokonui displayed the highest adult survival, with variability in annual estimates decreasing over time (Fig. 4, Appendix 1). Silver Peaks robins had the lowest adult survival, with variability decreasing towards 2012, then increasing again. Robins at Silverstream showed a similar trend in variability to those at Silver Peaks, but a higher overall adult survival for all years (Fig. 4, Appendix 1).

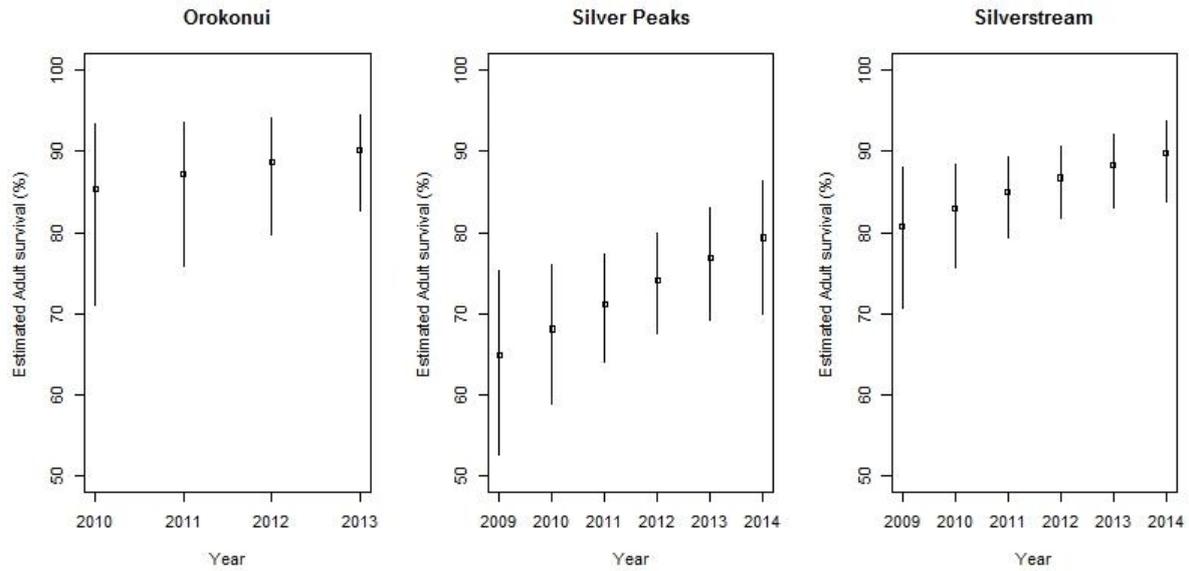


Figure 4: Estimated adult survival (and associated 95% CIs) for SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2008/09 (2010/11 for Orokonui) to 2014/15 (2013/14 for Orokonui). $n_{\text{Orokonui}}=88$, $n_{\text{SP}}=68$, $n_{\text{SS}}=49$. No estimate was produced for 2008 for any site as survival estimates are calculated as a function of the previous year's presence/absence data, and there was no data for 2007). No estimates were produced for 2009 and 2014 for Orokonui as robins were only translocated in 2010 and not monitored in the 2014/15 breeding season.

PRADEL JUVENILE RECRUITMENT

Pradel juvenile recruitment analysis, which represents juvenile recruitment as a proportion of the given year's total population rather than a proportion of that year's juveniles, revealed the best model for estimating juvenile recruitment of SI robins contained the factors Year and Area (Table 8).

Table 8: Model selection criteria for Pradel mark-recapture models of juvenile recruitment for SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2008/09 to 2014/15. Models are listed sequentially with the model with the lowest AICc first.

| Model | Factors | AICc | ΔAICc | Deviance |
|-------|-----------|----------|---------------------|----------|
| Area | Year+Area | 1708.219 | 0.000 | 225.275 |
| Null | Year | 1874.009 | 165.790 | 132.622 |

There was a decreasing trend in juvenile recruitment of SI robins at all three sites for the study period of 2008/09 to 2014/15. Orokonui robins displayed the highest proportional juvenile recruitment, with variability in the estimate decreasing in subsequent years. Silverstream robins displayed the lowest proportional juvenile recruitment, with variability remaining mostly constant. Silver Peaks robins displayed a similar trend in variability to that of Silverstream, but higher overall proportional juvenile recruitment for all years (Fig. 5, Appendix 2). However, confidence intervals indicate no significant differences in juvenile recruitment over the study period between populations at Silverstream and Silver Peaks (Fig. 5, Appendix 2). Confidence intervals for Orokonui indicate juvenile robin recruitment to be significantly higher than at the other two sites for every year of the study period (Fig. 5, Appendix 2).

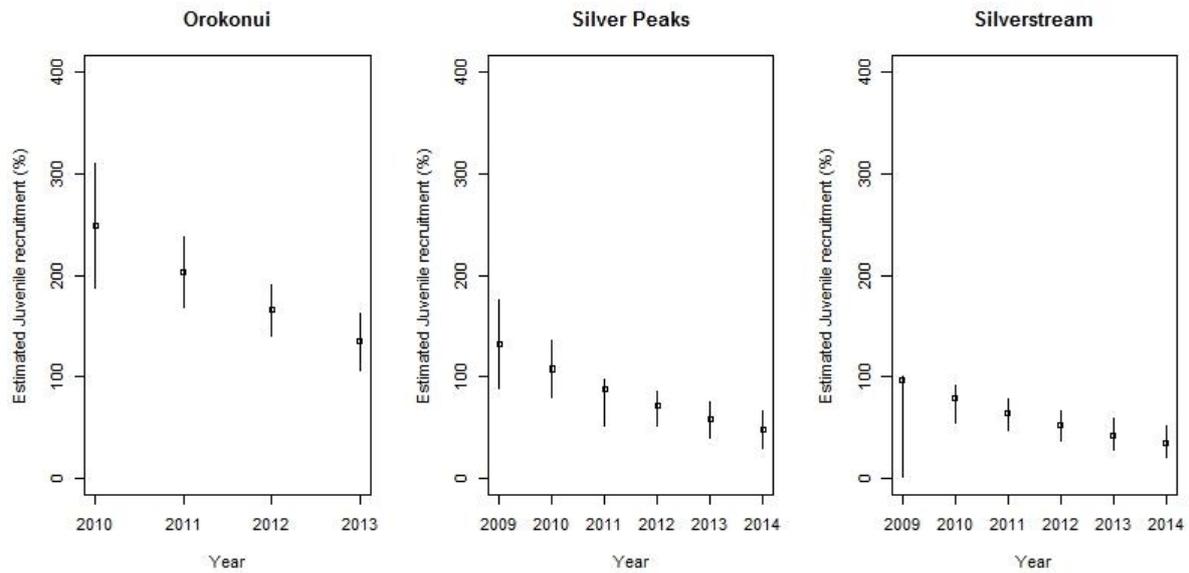


Figure 5: Estimated juvenile recruitment (and associated 95% CIs) for SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2008/09 (2010/11 for Orokonui) to 2014/15 (2013/14 for Orokonui). $n_{\text{Orokonui}}=264$, $n_{\text{SP}}=80$, $n_{\text{SS}}=61$. No estimates were produced for 2008 as survival estimates are calculated as a function of the previous year's presence/absence data (no data for 2007). No estimates were produced for 2009 and 2014 for Orokonui as robins only translocated in 2010 and not monitored for 2014/15 breeding season.

BINOMIAL RECRUITMENT

Binomial GLM recruitment analysis revealed Area and Age class as significant predictors of recruitment (Table 10). Year was not a significant predictor of recruitment and was not included in the best-fit model (Table 9).

Table 9: Analysis of variance and associated effect size (η^2) values for binomial generalised linear model of recruitment from model: Year+Area*Age Class. Factors are listed sequentially in order of addition.

| Factor | df | Deviance | Residual deviance | Residual df | p-value | η^2 |
|----------------|----|----------|-------------------|-------------|---------------|----------|
| NULL | - | - | 459.56 | 333 | - | - |
| Year | 5 | 7.387 | 452.17 | 328 | 0.193 | 0.385 |
| Area | 2 | 4.805 | 447.36 | 326 | 0.090 | 0.472 |
| Age Class | 1 | 36.153 | 411.21 | 325 | $1.824e^{-9}$ | 0.812 |
| Area:Age Class | 2 | 8.389 | 402.82 | 323 | 0.015 | - |

Initially, Area displayed marginal significance, but after removal of Year due to low significance ($p=0.193$, see Table 9), significance of Area was increased (Table 10).

Table 10: Analysis of variance and associated effect size (η^2) values for binomial generalised linear model of recruitment from model: Area*Age Class. Factors are listed sequentially in order of addition.

| Factor | df | Deviance | Residual deviance | Residual df | p-value | η^2 |
|---------------|----|----------|-------------------|-------------|----------------------|----------|
| NULL | - | - | 459.56 | 333 | - | - |
| Area | 2 | 7.364 | 452.19 | 331 | 0.025 | 0.637 |
| Age Class | 1 | 32.077 | 420.11 | 330 | 1.482e ⁻⁸ | 0.727 |
| Area:Age Clas | 2 | 12.043 | 408.07 | 328 | 0.002 | - |

Model selection indicated the interaction model containing the main effects Area and Age Class and the interaction effect Area:Age Class to be the best model for predicting recruitment (Table 11).

Table 11: Model selection criteria for binomial generalised linear model of recruitment for all permutations of model including the interaction Area*Age Class. Models are listed sequentially with the model with the lowest AICc first.

| Model | Factors | AICc | Δ AICc |
|--------------|---|-------|---------------|
| Interaction | Intercept+Age Class+Area+Age Class:Area | 420.3 | 0.00 |
| Full effects | Intercept+Age Class+Area | 428.2 | 7.91 |
| Age Class | Intercept+Age Class | 445.3 | 24.99 |
| Area | Intercept+Area | 458.3 | 37.94 |
| Null | Intercept | 461.6 | 41.24 |

SI robins at Silverstream displayed the highest adult recruitment and those at Orokonui displayed the lowest (Fig. 6, Appendix 3). However, confidence intervals overlap indicating no significant difference in adult recruitment between the three sites (Fig. 6, Table 12, Appendix 3). Robins at Orokonui displays the highest juvenile recruitment with Silverstream displaying the lowest, a significant difference in juvenile recruitment is evident between Silverstream and Orokonui. Robins at Silver Peaks displayed no significant difference compared to the other two sites (Fig. 6, Table 12, Appendix 3). Robins at Silverstream also displayed a significantly lower juvenile recruitment as compared to adult recruitment (Fig. 6, Table 12, Appendix 3).

Table 12: Coefficient estimates (and associated 95% CIs and p-values) of recruitment of SI robins for the binomial generalised linear model: Area*Age Class.

| Factor | Estimate(β) | Standard error | CI | p-value |
|-----------------------|---------------------|----------------|-------------------------|---------|
| (Intercept) | 0.3365 | 0.5855 | -0.806< β <1.554 | 0.5655 |
| Silver Peaks | 0.5798 | 0.6655 | -0.772< β <1.884 | 0.3836 |
| Silverstream | 0.9853 | 0.7079 | -0.434< β <2.387 | 0.1640 |
| Juvenile | 0.2360 | 0.6151 | -1.032< β <1.436 | 0.7012 |
| Silver Peaks:Juvenile | -1.6544 | 0.7404 | -3.106< β <-0.168 | 0.0255 |
| Silverstream:Juvenile | -2.8735 | 0.8075 | -4.474< β <-1.275 | 0.0004 |

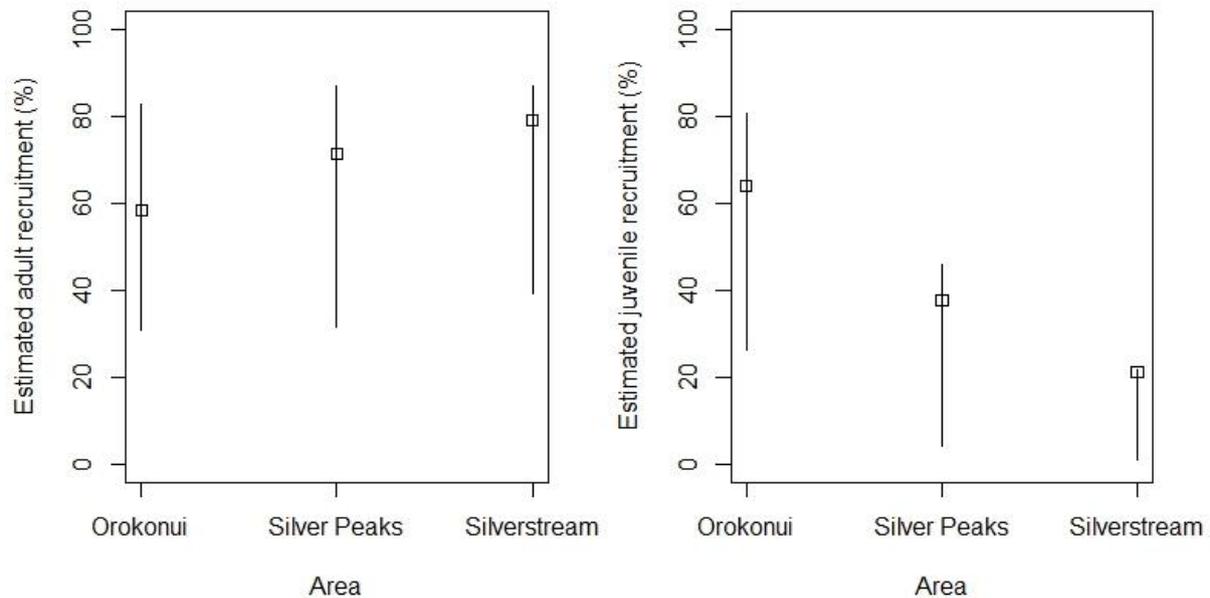


Figure 6: Recruitment estimates (and associated 95% CIs) for three sites (Orokonui, Silver Peaks, and Silverstream) for two age classes (Adults and juveniles) of SI robins. $n_{\text{adult}}=100$, $n_{\text{juvenile}}=234$. $n_{\text{Orokonui}}=135$, $n_{\text{SP}}=110$, $n_{\text{SS}}=89$.

ESTIMATED NEST SURVIVAL

The best model for predicting daily survival rate (DSR) and subsequent estimated nest survival (ENS) contained only the factor Area (Table 13).

Table 13: Model selection criteria for nesting success mark-recapture models for SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2010/11 to 2014/15. Models are listed sequentially with the model with the lowest AICc first.

| Model | Factors | AIC | ΔAICc |
|-------------|-----------|---------|---------------------|
| Area | Area | 746.214 | 0.000 |
| Interaction | Area*Year | 761.431 | 15.217 |
| Year | Year | 832.967 | 86.753 |
| Time | Time | 839.366 | 93.152 |
| Null | Intercept | 840.209 | 93.995 |

Silver Peaks displayed a decreasing trend in estimated nest survival (Fig. 7, Appendix 4) although confidence intervals indicated no significant differences between years (Fig. 7). Silverstream ENS values were very low and showed an initial decrease followed by a gradual increase in ENS (Fig. 7, Appendix 4), although confidence intervals indicate no significant differences between years (Fig. 7). ENS was high at Orokonui, rising to nearly 80% in 2011 and stabilising at around 72% (Fig. 7, Appendix 4). However, confidence intervals indicate no significant differences between years (Fig. 7).

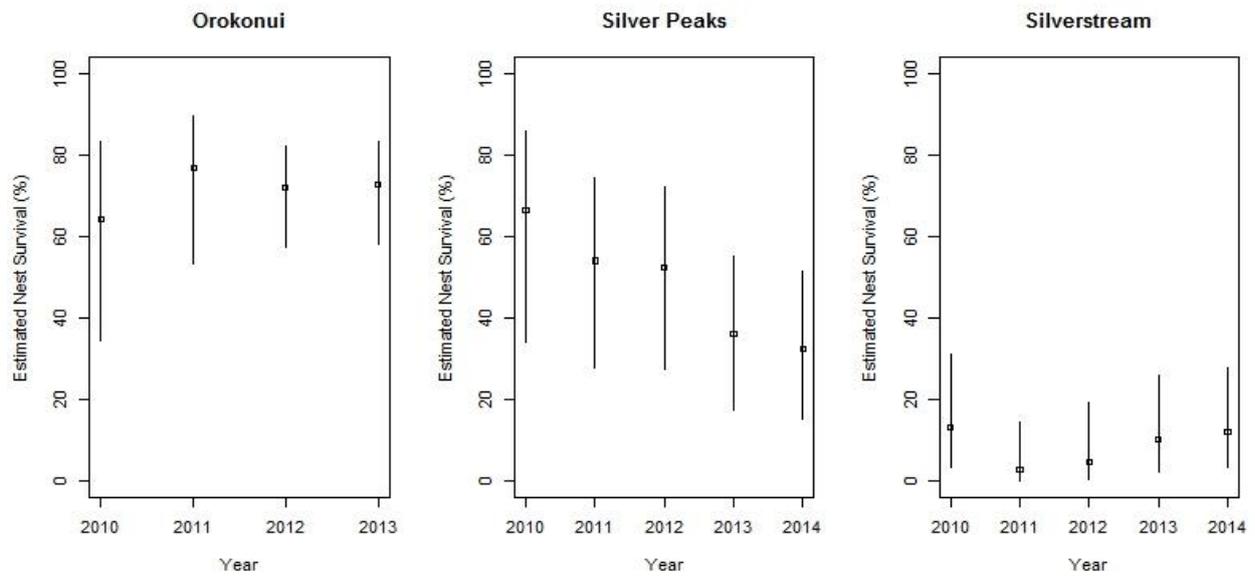


Figure 7: Estimated nest survival (and associated 95% CIs) for SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2009/10 to 2014/15 (2013/14 for Orokonui¹).
¹No estimates for 2014 for Orokonui as robins not monitored for 2014/15 breeding season.

NEST STAGE FUNCTION

When taking into account nest stage (incubation or nestling) both the best models for predicting DSR and subsequent ENS included the factor Area (Table 14 and Table 15).

Table 14: Model selection criteria for nest survival mark-recapture models for SI robins during the incubation period at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2010/11 to 2014/15. Models are listed sequentially with the model with the lowest AICc first.

| Model | Factors | AIC | $\Delta AICc$ |
|-------------|-----------------------------|---------|---------------|
| Interaction | Area*Stage(Incubation)+Year | 744.925 | 0.000 |
| Null | Stage(Incubation)+Year | 840.126 | 95.201 |

Table 15: Model selection criteria for stage-based nest survival mark-recapture models for SI robins during the nestling period at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2010/11 to 2014/15. Models are listed sequentially with the model with the lowest AICc first.

| Model ^a | Factors ^b | AIC ^c | $\Delta AICc^d$ |
|--------------------|---------------------------|------------------|-----------------|
| Interaction | Area*Stage(Nestling)+Year | 745.688 | 0.000 |
| Null | Stage(Nestling)+Year | 840.449 | 94.761 |

SI robins at Orokonui had the highest nest survival rate during both incubation and nestling stages. Robins at Silverstream had the lowest for both stages, with confidence intervals for Silverstream indicating significantly lower incubation ENS than Orokonui and significantly lower nestling ENS than

at both Orokonui and Silver Peaks. Confidence intervals for Silver Peaks indicate no significant difference from Orokonui (Fig. 8, Appendix 5).

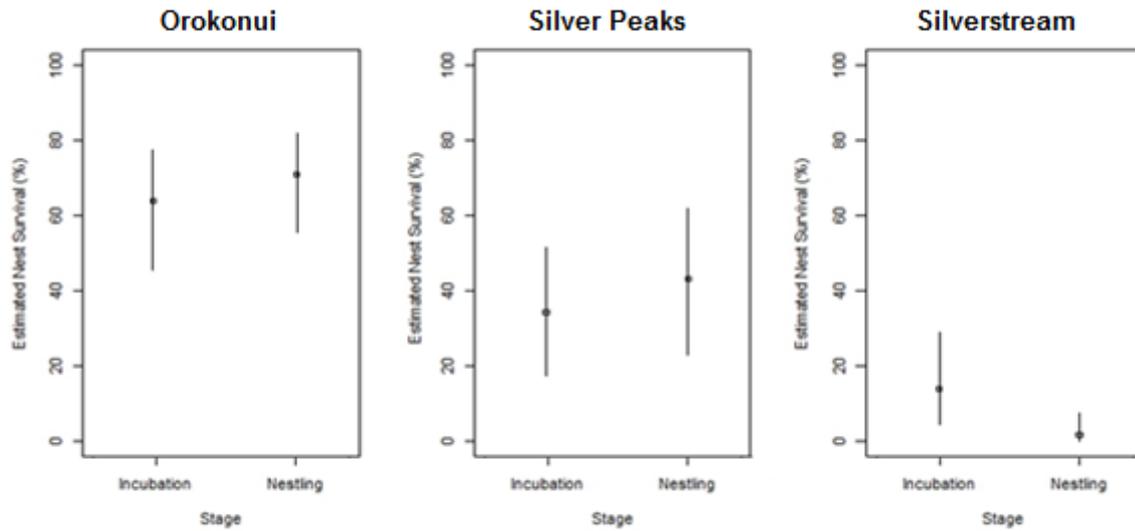


Figure 8: Estimated nest survival (and associated 95% CIs) of both incubation and nestling periods for SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2009/10 to 2014/15 (2013/14 for Orokonui¹). ¹No estimates for 2014 for Orokonui as robins not monitored for 2014/15 breeding season.

DISCUSSION

PREDATOR RELATIVE ABUNDANCE

Monitoring at Silver Peaks in 2011 before and after the aerial 1080 operation revealed ~80% reductions in all three monitored pest species (possums, ship rats and mice), with numbers remaining low in the follow-up survey seven months later (Schadewinkel et al., 2014). Since then, increases in relative abundance of possums and rats were observed, with mouse numbers fluctuating but remaining low since March 2013. This is characteristic of rebounding predator populations, whereby numbers are greatly reduced by eradication but can rapidly increase to or exceed pre-operational levels (Nugent et al., 2011, Ruscoe et al., 2011).

Mice were the quickest to recover after the operation, with monitoring in February of 2012 indicating numbers almost twice what was observed prior to the 1080 operation. Numbers increased rapidly as the pressure of predation and competition by rats was relieved (Ruscoe et al., 2011). Decreases observed after February 2012 may be confounded by increases in rat numbers, as rat bite marks on CTCs have been known to greatly diminish the ability to detect mouse bite marks due to destruction of the card by rats (Sweetapple and Nugent, 2011). However, it is likely that a true decrease occurred as increases in rat numbers would translate to increases in competition and predation rates of mice by rats, resulting in decreases in mouse numbers. Initial reductions in mice were considered novel, as most 1080 operations have very little effect on this species (Nugent et al., 2011, Bridgman et al., 2013). This could have been a product of pre-feeding combined with possible low food availability, resulting in higher rates of bait-acceptance (Schadewinkel et al., 2014).

Rats showed a somewhat slower recovery time, with relative abundance remaining moderately low for a year, but since then increasing to twice their pre-operational number, and remaining high, often peaking during the robin breeding season. Rats are known to recover rapidly from poisoning operations (Nugent et al., 2011), and the high abundance of mice after the 1080 operation most likely played a role in their recovery at Silver Peaks, as rats are known to predate mice (Tompkins and Veltman, 2006). High mice relative abundance would provide rats with a source of food and this, coupled with reduced inter and intra-species competition due to initial low rat and possum densities, would facilitate their rapid recovery (Nugent et al., 2011).

Possums were the slowest to recover after the operation, with numbers remaining low until December of 2012. They have since recovered to numbers greater than pre-operational estimates, but have only very recently reached twice the pre-operational number. This is likely due to reduced food supply as a result of the rapid recovery of competitive rodents (Nugent et al., 2011).

Monitoring at Silverstream indicated varying levels of predator numbers, with possums displaying the highest relative abundance (Schadewinkel et al., 2014). Pre-trap monitoring indicated high relative abundance of rats and possums and fluctuating abundances of mice. Since the installation of the Goodnature™ A24 traps, relative abundance of possums has increased concurrently with declines in relative abundance of rats. Mouse relative abundance has fluctuated but remained low since May 2012.

Rats showed a marked decrease in December of 2012 (the month of the trap installation), with numbers remaining low until March 2014. Rats have since only reached pre-trap numbers once, with

this peak occurring outside of the robin breeding season. Trapping is considered an effective way of managing pest species where those species do not exceed critical abundances or where the species are not innately 'trap-shy' (Innes et al., 2010)). In the case of rats at Silverstream, trapping seems to have been effective in controlling their population. The decreases in rats observed at Silverstream might also be due in part to the presence of predatory stoats. Stoats have been observed on many occasions in Silverstream (pers. obs.) and are known to depredate rodents in large numbers (Murphy et al., 1998). This might contribute to decreases in rat abundances, although the stoats' role in the decrease of rats at Silverstream has not yet been quantified.

Possums at Silverstream maintained a high and stable relative abundance until December 2012, when relative abundance increased to almost twice the original number. This coincided with the installation of the rodent traps and is most likely a case of competitive release or of ship rats obscuring possum bite-marks on CTCs. Observed reductions in rats would relieve competition pressure on possums, allowing them to rapidly increase. Competitor release has been suggested as a common outcome of some operations, often playing a more significant role than mesopredator release (Ruscoe et al., 2011). As discussed, whether this competitive release is a result of trapping or of stoat activity remains uncertain. Alternatively, ship rats are known to obscure possum bite marks on CTCs when saturation of the card occurs. This might be responsible for the increase in possum relative abundance after control of ship rats. This seems more likely, as possums breed only once per year, with this occurring in Autumn.

Mice numbers at Silverstream remained low, with moderate fluctuations throughout the study period. Mice showed initial declines after the installation of the rodent traps in December 2012, but these declines were short-lived and probably not due to trap-catch (as the traps are designed for rats and stoats). This is most likely a product of decreases in rat numbers observed over this period, as rats are known to depredate mice (Tompkins and Veltman, 2006). Declines in rats would relieve predation pressure on mice, and could counteract the initial declines in mice numbers observed after trap installation. Fluctuations in mice numbers may also be attributable to stoat activity, as stoats are known to depredate mice as well as rats (Tompkins and Veltman, 2006), and this might explain the absence of a mouse irruption following the rat control.

NEST CAMERAS

Video monitoring of nest sites at Silverstream revealed stoats to be the primary predator of robin nests, responsible for the failure of 14 nests. This contradicts what was expected; that possums would be observed as the primary nest predator due to decreases in rat numbers after the installation of rodent traps. While possums did match stoats in egg predation, possums did not depredate nestlings. Ship rats depredated only one nest at the incubation stage and mice were not observed to have predated any nests. Stoats depredated a total of 46% of all observed nests; 33% chicks and 13% eggs, during a time when rat relative abundance was the lowest it had been since June 2013, indicating increased stoat activity in the Silverstream catchment.

Stoats are known to depredate rats, and research suggests that in areas or periods with rat population booms, stoats will preferentially prey upon rats over birds (Murphy et al., 1998). Rat population spikes in March and June of 2014 may have acted to increase stoat activity in the Silverstream area just prior to the breeding season. Reduced rat numbers before the start of the season due to trapping might have led to the high incidence of stoat predation of nests during the

breeding season when rat numbers were low. Whether this is a recurring pattern is uncertain as nest predator data were not collected in previous seasons.

Four nests had unknown fates and were all at the nestling stage, but predators were determined from nest remains (Brown et al., 1998). Three were attributed to stoats and one to rats. Camera failure due to battery life or weather effects was the most common reason for unknown fates of nests.

ROBIN POPULATION METRICS

OROKONU

Adult survival at Orokonui was high throughout the duration of the study, with no significant differences observed between any of the years or between sexes, although uncertainty around the estimate was reduced across the study period. Proportional juvenile recruitment was initially very high, and subsequently decreased significantly across the duration of the study, but remained above 100%. No differences in absolute recruitment rates were detected between adult and juvenile robins, neither recruitment metric differed between years of the study at Orokonui. Robin nesting success at Orokonui was high throughout the study, with no differences in estimated nest survival detected between any years of the study. No differences in nest survival were detected between the incubation and nestling stages at Orokonui.

In Orokonui, where predators are almost absent (with the exception of rare incursions of mammals and the occasional avian predator), predation of adult and juvenile birds as well as nests is likely to be low or absent. Predator abundances are known to be highly negatively correlated with adult survival, juvenile survival, and nesting success in robins (Armstrong et al., 2014) (Armstrong and Ewen, 2002, Schadewinkel et al., 2014) and this could explain the relatively high and stable levels of these population metrics observed within the sanctuary after predator removal, suggesting predation is a limiting factor for this SI robins.

Studies of robin responses to simulated predators indicate that adult robins display marked wariness toward identified predators (alarm calling, especially when feeding fledglings, and avoidance of identified predator) (Maloney and McLean, 1995), suggesting adult robins might be less vulnerable to predation than previously thought, although predation rates of adults in this study remain unclear. Alternatively, the absence of introduced mammals in their role as competitors with robins may be more significant with respect to adult survival in the Orokonui population. Many introduced predators such as ship rats and possums act as competitors with birds (Innes et al., 2010), consuming a large variety and volume of invertebrates, the principal component of robins' diet. This can result in reduced adult survival, with robins in areas of low invertebrate biomass being forced out of territories in search of more favourable conditions (Boulton et al., 2008). The absence of introduced competitors in Orokonui ecosanctuary might increase the availability of invertebrates and negate any potential resource competition. This seems likely, as survival rates are high, suggesting adults birds are not dispersing but are instead remaining in the sanctuary. Adult robins are known to display high levels of site fidelity, with established birds rarely abandoning their territories (Higgins and Peter, 2002). This, coupled with the absence of competitors, would explain Orokonui's high adult survival rate.

Due to the nesting strategy of robins (female incubation), I predicted that females would be more vulnerable to predation during a nest predation event. In Orokonui, no sex-based bias in adult survival was observed. This is most likely due to the absence of mammalian predators within the sanctuary combined with wariness of robins to predators. If robins have been reported to show marked wariness of mammalian predators (Maloney and McLean, 1995), predation of adult females at the incubation stage may also be negligible in areas with predators. Robins have been shown to lose their wariness of mammalian predators after translocation to predator-free areas (Jamieson and Ludwig, 2012). This might mean female robins at Orokonui could be more vulnerable to predation on the nest than their Silver Peaks and Silverstream equivalents, but this would only be a problem if incursions of predators were to occur.

High levels of site fidelity combined with the low connectivity of Orokonui to other remnant populations (the closest being Silver Peaks) might also be affecting recruitment rates. For adult recruitment, where failure to capture and band robins at the juvenile stage occurs, juvenile robins that do not disperse after the breeding season are often captured the next year and recorded as an instance of adult recruitment. This would act to bias the estimate of adult recruitment high and juvenile recruitment low. The rate of occurrence of this is not known, but is thought to be lower at Orokonui than at the other two sites due to the ease of access of Orokonui and the well-mapped territories therein, making robin capture and banding simpler. This might also be facilitated by the absence of invasive mammalian competitors reducing competition for invertebrate biomass and facilitating a greater number of robin territories (Boulton et al., 2008).

Declines in proportional juvenile recruitment, representing the annual recruitment of juveniles as a proportion of the total population in the sanctuary, indicate that the small and initially highly productive population was able to support settlement of many juveniles in the sanctuary but that, as adult robin numbers increased, the number of juvenile birds being recruited has diminished. Due to the high site fidelity of adult robins (Higgins and Peter, 2002), forced dispersal (i.e. a robin being forced out of an area by another established robin) most often occurs in juvenile birds. Where competition pressure is high we would expect to see reduced juvenile recruitment as juveniles are forced out of adult territories. At Orokonui, where introduced mammalian competitors are excluded, competition pressure was initially reduced but has increased since the establishment of adult robins. This explains why I observed declines in proportional juvenile recruitment despite high estimates of absolute juvenile recruitment; the absolute rate of recruitment remained stable and high (i.e. the same proportion of juveniles produced annually were settling in the sanctuary) but the number of established adults was increasing, meaning the proportional rate of recruitment decreased.

The stable and high estimates of nesting success at Orokonui provide evidence to suggest that predation is a limiting factor for robin populations in Dunedin. Invasive mammalian predators such as rats, possums and stoats are well known for their predation of both eggs and chicks of a number of native bird species (King and Moody, 1982, O'Donnell, 1996, O'Donnell et al., 1996). These predation events decrease nesting success and in some cases (though not found in this study) can result in male-biased operational sex ratios, where adult females are depredated while incubating the nest. However, in Orokonui, where mammalian predators have been eradicated, this does not occur and estimates of nesting success are high, contributing to healthy population growth. In addition, the absence of competition may contribute to the high estimated nest survival at Orokonui. Competition for invertebrate resources has been shown to affect nesting behaviour in

robins, with robins in areas of low invertebrate biomass being forced to spend more time foraging (Boulton et al., 2008) and having a greater prevalence of nest failure (Luck, 2002). Invasive predators are thought to also act as competitors because they consume large quantities of invertebrates (Innes et al., 2010). At Orokonui, competition pressure with mammals is negligible, meaning adult birds do not have to spend more time foraging, consequently reducing the risk of nest failure. This is supported by a number of studies that show that areas with higher invertebrate biomass are often the most productive (Newton, 1998, Crawford et al., 2006, Boulton et al., 2008). However, invertebrate biomass has not yet been measured at Orokonui, so the effect of reductions in competition with introduced mammals at this site remains speculative. I might expect to observe an eventual decrease in nesting success when populations of insectivorous birds reach carrying capacity, with intra and interspecific competition peaking. However, it is more likely that this will result in higher rates of juvenile exclusion, as adult birds show high levels of site fidelity (Higgins and Peter, 2002) and would exclude juveniles from territories.

Where robins coexist with mammalian predators, begging behaviours of nestlings can increase the risk of nest predation (Boulton et al., 2008) by increasing the conspicuousness of the nest to predators (Boulton et al., 2008). In areas where this occurs, failure rates of nests are expected to be greater at the nestling stage than at the incubation stage. The absence of any difference in nest survival between the incubation and nestling stages is most likely due to the absence of mammalian predators within the sanctuary. Begging behaviours are also thought to be more pronounced in areas with lower invertebrate biomass (Leech and Leonard, 1997). If predators are also acting as competitors by consuming invertebrates (Boulton et al., 2008) then chicks could be depredated more often in areas where predators are abundant and acting simultaneously as competitors.

SILVER PEAKS

Robins at Silver Peaks displayed an increasing trend in adult survival rate across the duration of the study, though this increase was not significant for any year or for either sex. Proportional juvenile recruitment was initially high, but decreased to below 100% across the duration of the study. Adult recruitment at Silver Peaks was high, with a slightly lower rate of absolute juvenile recruitment, although this difference was not significant. Estimated nest survival for robins at Silver Peaks was initially high, decreasing over the duration of the study. No significant differences in ENS were detected between any years of the study or between the incubation and nestling stages at Silver Peaks.

Increases in adult survival occurred at Silver Peaks despite concurrent increases in predator relative abundances, suggesting predation rates of adults by the three monitored predators are low, possibly due to wariness of adult robins to predator species (Maloney and McLean, 1995). I observed that robins at Silver Peaks become alarmed when aware of nearby stoats and rats. This wariness might act to reduce the vulnerability of adult birds to predation by these species (Maloney and McLean, 1995). This wariness might also explain why no differences in sex-based adult survival were detected, despite expected increased vulnerability of nesting females to predation (Maloney and McLean, 1995). During the incubation period, females will remain on the nest, being fed by foraging males. At this stage females remain highly wary of activity near the nest (pers. obs.) and footage from nest cameras placed in Silverstream indicate a swift reaction of incubating females to invading nest predators. It is most likely that the majority of birds that are thought not to have survived from the previous year have instead dispersed to surrounding forest patches, as there remains a good

level of connectivity between adjacent native and exotic stands within Silver Peaks (Parker, 2013) and birds thought previously to be deceased have been observed in adjacent areas during extended surveys.

High rates of adult recruitment coincided with increases in predator relative abundances, suggesting predator numbers did not play a significant role in the high and stable rate of adult recruitment. It is likely that high adult recruitment was a product of high rates of adult survival coupled with moderate rates of immigration of adult birds from adjacent plots, facilitated by high levels of connectivity between adjacent native remnant and conifer plots (Parker, 2013). Research into connectivity of conifer plantations indicates high levels of connectivity are often maintained between conifer stands and adjacent native remnant stands (Carnus et al., 2006). Adult robins recruited into the population are most likely juvenile birds from the previous breeding season from adjacent unstudied plots or juveniles that failed to be captured and banded prior to dispersal, as adult robins with established territories are not likely to move far (Higgins and Peter, 2002).

Observed declines in proportional juvenile recruitment at Silver Peaks coincided with increases in abundances of monitored predators, suggesting predation may have played a factor in this decline. However, proportional juvenile recruitment represents the annual recruitment of juveniles as a proportion of the total population. In reality, this measurement of juvenile recruitment does not indicate that raw annual recruitment of juveniles is declining, only that the population of adults is increasing. Initially high proportional juvenile recruitment rates may be indicative of a small founder population being moderately to highly productive and, as the population of adults increases, fewer juveniles are able to settle (e.g. Orokonui). Additionally, absolute juvenile recruitment was low but showed no differences between any years. This suggests that predation by the three monitored species may have played only a minor role and that a different predator, possibly stoats, might be responsible for the low recruitment, or that juveniles are highly dispersive. Stoats were encountered at Silver Peaks on several occasions (pers. obs.) although no attempt was made to monitor their abundance, this being a difficult task (Gleeson et al., 2010). Stoats are considered to be one of the most destructive introduced predators affecting New Zealand's avifauna today. They are highly efficient and mobile killers, and are known to prey upon a wide range of bird species. Stoat predation could be responsible for the low observed absolute juvenile recruitment rate if stoats were preying on juveniles. Predator monitoring indicated increases in rats over the duration of the study, and increases in rat abundances have been correlated with increases in stoat abundances (Murphy et al., 1998). However, research into the causal relationship between increasing populations of stoats in response to increases in rat populations have shown that when rat populations boom, stoats will feed mainly on rats and the incidence of bird catch is often low (Murphy et al., 1998). This, coupled with the strong predator avoidance displayed by robins (Maloney and McLean, 1995) suggests that stoats may not be responsible for the low juvenile recruitment. Alternatively, low observed absolute juvenile recruitment may be a product of dispersal by juvenile birds out of the study area. Juvenile birds at Silver Peaks can disperse before capture and banding occurs; these dispersal events can obscure the true absolute juvenile recruitment rate and could inflate the adult recruitment rate.

The consistent trend in decline in estimated nest survival at Silver Peaks across the duration of the study from an initially high rate coincided with an initial reduction in predator relative abundances followed by marked increases in relative abundance for all three predator species within a year of

the 1080 operation. I would expect that, after the 1080 operation, estimated nest survival would increase due to the alleviation of predation pressure. However, ENS continued to decline, despite reduced predator numbers. This may suggest that the primary limiting predator, at least immediately following the eradication, was not one of the monitored species; i.e. stoats. It was previously thought that robin nests at Silver Peaks might be afforded some level of protection from stoat predation due to the increased height of the nesting trees, those being mature Douglas fir. However, stoats exhibit arboreal behaviours, and are able to find and access species nesting high in trees (Dilks et al., 2003). This is likely to have played some role in the continual decline in ENS observed at Silver Peaks. Monitored species rebounding rapidly after the operation to abundances greater than pre-operational estimates could also result in decreasing ENS through increased predation rates, with the lowest observed ENS values coinciding with the greatest rat and possum CTC values. However, decreases in ENS observed between years were not significant, suggesting the effect of predation may not be as pronounced as previously thought, or that more nesting data is required to detect an effect.

An alternative explanation for decreasing ENS is competition. Increases in adult survival combined with recruitment of adult and juvenile birds would increase competition for food, and together with increases in monitored predators, would increase competition pressure for invertebrate resources (Innes et al., 2010). Invertebrate biomass has been shown to be linked with nesting success in the closely related North Island robin (*Petroica longipes*), influencing foraging behaviour and ultimately nesting success (Boulton et al., 2008). Robins are known to have low reproductive success where invertebrate biomass is low, and this might play a part in Silver Peaks. However, no attempt has been made to assess invertebrate biomass at Silver Peaks and evidence from other studies suggests invertebrate biomass might be preserved in a number of conifer plantations (Pawson et al., 2008, Pawson et al., 2009), so this relationship remains uncertain. It is difficult to discern whether competition is the primary cause for failure of nests where nests are depredated (Boulton et al., 2008): nests might fail due to starvation, when adult birds fail to provide sufficient invertebrates for their offspring (Luck, 2002), but some nests are hard to monitor when inaccessible (such as in Silver Peaks). Begging behaviour increases when chicks are hungry, making the nest more conspicuous to potential predators (Leech and Leonard, 1997). Coupled with the increased time spent foraging and not tending the nest (Zanette et al., 2000), these behaviours can increase risk of predation. I would expect that begging behaviours might also contribute to an increased failure rate at the nestling stage compared to the incubation stage due to begging behaviours and feeding nestlings increasing the conspicuousness of the nest to predators (Boulton et al., 2008), but nests at Silver Peaks were equally likely to fail at the nestling and incubation stages. The reason for this remains uncertain, but may be that conspicuousness only plays a significant role in areas or species where nesting occurs closer to the ground. This has been observed in birds, with ground-nesting species displaying decreased nesting success as compared to tree-nesting species (Haskell, 2002). Food availability is also thought to affect territory abandonment rates of robins, with increased emigration in areas with low food availability (Boulton et al., 2008). However, adult survival remained relatively high in Silver Peaks, suggesting adult robins are remaining in their territories and that competition for food may not be adversely affecting robins at this site.

SILVERSTREAM

Adult robins at Silverstream had high and increasing rates of survival across the duration of the study. Although differences in adult survival were not significant between any year of the study and no differences in adult survival were detected between sexes, uncertainty around the estimates was reduced. Proportional juvenile recruitment rates were low at Silverstream, decreasing significantly across the duration of the study. Adult recruitment at Silverstream was significantly higher than absolute juvenile recruitment, but no differences were detected between years of the study. Estimated nest survival was low and decreased across the duration of the study, although no significant differences were detected between years of the study or between nestling and incubation stages.

Increases in adult survival corresponded with increases in possum relative abundances and fluctuations in rodent relative abundances. This suggests predator removal did not play a significant part in this increase, and that predation rates of adults at Silverstream by the three monitored predators are inherently low due to wariness of adult robins to predator species (Maloney and McLean, 1995). This is consistent with observed responses of adult robins at Silverstream; becoming alarmed when aware of nearby stoats (pers. obs.) and avoiding predation (Maloney and McLean, 1995). This also applies to incubating females, with video data indicating a strong avoidance response of adult females to nest predators as well as observers. This would explain the absence of any sex-based bias in adult survival rates at Silverstream. It is likely that only a small number of birds that were thought not to have survived from the previous year had been predated. More probable is that, of the small number that were not resighted, the majority instead moved to adjacent areas. Much like Silver Peaks, Silverstream is a moderately ubiquitous forest and there remains a good level of connectivity between the Silverstream site and adjacent unstudied forest habitats. This would seem to contradict the high levels of site fidelity observed in robins (Higgins and Peter, 2002), but a relocation to suitable territory within Silverstream would not require the same level of displacement as in Silver Peaks.

It is likely that, much like in Silver Peaks, high adult recruitment is a product of high rates of adult survival and moderate rates of adult dispersal, facilitated by high levels of connectivity within the Silverstream catchment. No differences in adult recruitment were observed between any years of the study, suggesting adult recruitment has remained relatively constant throughout the study period despite fluctuations in predator abundances, suggesting that, like adult survival, adult recruitment at Silverstream might not be significantly influenced by predator abundances. High rates of adult recruitment may also be a result of failure to capture juvenile birds within the study area combined with rapid rates of juvenile dispersal. Robins at Silverstream were banded only after fledging to reduce the risk of nest abandonment and predation, and any juveniles that were not located and banded before dispersal could re-enter the population the next year as adults, suggesting that adult recruitment events might primarily be young birds seeking to establish a territory. This seems to contradict earlier conclusions for adult survival however, the rate of adult migration, despite being the primary cause for failure to resight adult birds, is relatively low, as evidenced by the high adult survival rate.

The low and declining proportional juvenile recruitment rate at Silverstream coincided with increases in abundances of possums and fluctuations in rodent abundances, suggesting predation may have influenced juvenile recruitment via nesting success. Proportional juvenile recruitment

does not indicate raw annual recruitment of juveniles is declining, but might indicate that the population of adults is increasing and that juveniles are being excluded by adult birds. At Silverstream, proportional juvenile recruitment never exceeded 100%. This might be cause for alarm as Silverstream is a relatively small population with only a small number of adults, suggesting that very few juveniles are being recruited into the population. This is supported by the low rate of absolute juvenile recruitment. The absence of any differences in absolute juvenile recruitment between years indicates predation by possums or rodents did not influence recruitment, as changes in these predators' abundances did not affect recruitment. Alternatively, a different predator (e.g. stoats) may be responsible for the low recruitment, or it may be that juveniles are leaving the study area immediately. Stoats were observed in the Silverstream catchment on several occasions both haphazardly and on nest cameras (pers. obs.). Predation of juvenile robins by stoats could have played a role in the low absolute juvenile recruitment rate at Silverstream. Increases in rat abundances have been correlated with increases in stoat abundances (Murphy et al., 1998). However, predator monitoring indicated fluctuations in rats over the duration of the study, with the abundances declining notably and remaining low after the installation of goodnature rat traps. I might expect that declines in rodent abundances would have resulted in higher rates of predation of juvenile robins by stoats, but this was not reflected in the results, as no differences in juvenile recruitment were observed between any years of the study. This, coupled with the strong predator avoidance displayed in robins (Maloney and McLean, 1995) suggests that stoats may not be responsible for the low juvenile recruitment. Juvenile birds at Silverstream are most likely being excluded from the study area by established adult birds before capture and banding can occur, and are reappearing the following year as adult recruitment events, obscuring the true absolute juvenile recruitment rate and biasing the adult recruitment rate high.

Nesting success of robins at Silverstream was lower than elsewhere and although it did increase over the past three seasons, was variable and never exceeded 14%. Possums might have replaced ship rats as the primary nest predator at Silverstream after rat trapping began. Possums have been observed to depredate nests of a number of bird species, including robins (Brown, 1997, Innes et al., 2012). Footage from nest cameras placed at Silverstream indicated possum nest predation (eggs) occurred twice, suggesting possums are most likely not responsible for the majority of nest predation events and that stoats, the predator identified as responsible for the majority of recorded nest predation events, are a more likely culprit.

Footage from nest cameras indicated stoats to be the primary predators of robin nests at both the incubation and nestling stages, and this might be the result of increased nest predation by stoats in response to declines in rat abundances. Depletion of rodents as the stoats' primary food source due to trapping might have resulted in increased predation of robin nests. Stoats have been shown to exhibit dietary-switches in areas of successful operations to control rodent abundances, preying mostly on birds (Murphy et al., 1998). It should be noted that the high observed predation rate of robin nests by stoats does not necessarily indicate either an increase in stoat abundance or an increase in the baseline predation rate, as data came from only one breeding season. Additionally, evidence suggests that the control of rats acts to reduce stoat abundance (Tompkins and Veltman, 2006). This suggests that stoat numbers may have been maintained or reduced in the Silverstream catchment over the study period, and that high rates of predation of nests by stoats and subsequent low ENS rates might instead be a result of prey-switching due to low rodent numbers. No significant difference in survival rate between the nestling and incubation stages was detected at Silverstream,

suggesting increased conspicuousness of the nest as a result of nestling begging and increased nest activity did not increase predation rates.

SUMMARY RESULTS

The predator fluctuations at Silver Peaks and Silverstream and the absence of those same predators from Orokonui have been influential in determining population metrics at these sites. Orokonui displayed consistently higher measures of all metrics except adult recruitment as compared to the other two sites, indicating a highly stable and highly productive population with relatively little dispersal. The robin population at Silver Peaks displayed intermediate levels of almost all metrics except adult survival, with this measure being lower than at the other two sites. This indicates this population is moderately productive and moderately stable, with comparatively high rates of dispersal. The robin population at Silverstream displayed the lowest values for almost all metrics excluding adult survival and adult recruitment, suggesting a poorly productive and likely unstable population. These low estimates of population metrics combined with the relatively high dispersal rate of robins at this site suggest the population might be threatened.

Although adult survival and juvenile and adult recruitment seem to be less affected by predator abundance than expected, nesting success was highly influenced by predation. Increased predator activity greatly reduced the nesting success of robins and, with nesting success playing a significant role in the stability and growth of a population, this can greatly affect the health and viability of these populations. This provides evidence to suggest that predation is a significant limiting factor for robin populations in Dunedin, but the potential influence of competition on robin population metrics in this study makes it difficult to discern whether predation is the primary limiting factor. Therefore, it remains important to carefully consider the methods of pest control at each site when comparing the populations.

PREDATOR CONTROL REGIMES

Ecosanctuaries such as Orokonui are a relatively recent development in the fight against introduced pest species. Many ecosanctuaries have proven valuable in restoring and promoting the health of populations of a number of native species (Burns et al., 2012). The objective of ecosanctuaries is the eradication and exclusion (or reduction) of pest species from an area. This allows increases in populations of a number of different native species, helping aid their path to recovery. Ecosanctuaries also provide an excellent opportunity to study native species interactions in the absence of introduced pests. To introduce desired species to the sanctuaries, translocations must first be carried out with ample time to allow settlement and establishment (Schadewinkel, 2013). Orokonui received two translocations of robins (as well as a number of other species), with the second translocation designed to boost initially low settlement rates (Schadewinkel, 2013). Evidence from this study suggests that the absence of the influence of introduced pests, not just as predators but possibly also as competitors, has allowed a five-fold growth of this population. The success of robins at Orokonui in this study was also attributed in part to the low permeability of the ecosanctuary, with very few birds entering or leaving the sanctuary, a feature that may prove inhibitory to the goal of robin work at the sanctuary. A key aspect of ecosanctuaries is their potential for spill-over into surrounding habitats (Russell et al., 2015), allowing the spread to and colonisation of areas outside the sanctuary.

Poisoning provides a cost-effective and relatively humane method of predator control. Poisoning is often favoured over trapping as it can be aerially dispersed over large areas, providing a much less labour-intensive method of deployment (Eason et al., 2011). Poisoning operations are highly effective in reducing pest target species abundances, and have the advantage in that they can target more than one pest species (Eason et al., 2011). These advantages can be instrumental in the control of highly abundant and rapidly reproducing pests such as rats and possums, playing a key role in the protection of desired species. However, poisoning can result in greater than pre-operational abundances of target species. Due to the relatively indiscriminate nature of poisons, there exists a potential for non-target species mortality through consumption of baits or through secondary poisoning (Eason et al., 2011). The prevalence of this is conditional upon the operation, and many steps have been taken to reduce the occurrence of non-target mortality (such as use of cereal baits with cinnamon deterrents/lures). Evidence suggests that even where initial mortalities of desirable non-target species occur, reduced predator abundances result in increases in desired species (Eason et al., 2011). Silver Peaks was the site of an aerial 1080 operation using cereal baits and pre-feed to target possums. It is notable that the 1080 operation at Silver Peaks incurred a zero mortality rate of monitored robins while also greatly reducing pest numbers prior to the robin breeding season (Schadewinkel and Jamieson, 2013b, Schadewinkel and Jamieson, 2013a), although no increases in nesting success or juvenile recruitment were observed. Poisoning has also been criticised for its failure to control pest species' abundances in the long-term. Often, following poisoning operations and subsequent reduction in pest species, abundances can increase to levels greater than pre-operational estimates (Nugent et al., 2011, Ruscoe et al., 2011). This can have consequences for the species the operation was intended to protect. Evidence from Silver Peaks suggests this might have occurred, with pest species rebounding to numbers greater than the pre-drop estimates and resulting in decreases in nesting survival. Developments in predator-specific poisons such as PAPP may lead the way in the phasing out of 1080 and anticoagulant poisons. With both aerial and ground-based dispersion as viable options and the high target specificity of the new generation of poisons, future operations may provide many of the benefits of current poison practices without the costs (Russell et al., 2015).

Trapping is considered an acceptable way of controlling pest species as it provides a relatively humane kill method and can be maintained long-term to ensure pest species abundances remain within a manageable level (Bomford and O'Brien, 1995). However, trapping operations often target a single species, which can result in either mesopredator or competitor release (Nugent et al., 2011, Ruscoe et al., 2011). This can be disastrous, whereby the intent of controlling an invasive and destructive pest can result in massive increases in the abundance of another destructive pest that had been previously controlled by the target species. This can have marked effects on any species that the operation was implemented to protect, and it is highly recommended that trapping operations be carefully planned so as to maximise the conservation goals while minimising the potential for negatively impacting desired species. The trapping line of 50 GoodnatureTM self-loading A24 rodent and stoat traps at the Silverstream site (loaded with rodent bait only) was successful in reducing rat relative abundance but these declines coincided with significant increases in possum relative abundance. However, possums were determined not to be the primary predator of nests at Silverstream, with stoats presenting the biggest threat to nesting success of robins at this site. Whether stoats were also affected by the rat trapping remains uncertain. Trapping can be cost-ineffective where the area that requires pest control is too large to allow suitable maintenance of

trap lines (Bomford and O'Brien, 1995). Current advances in trapping might enable circumvention of this problem through the use of auto-loading traps, such as those produced by Goodnature, with each trap having the potential to kill a much greater number of pests. This, in combination with developments of new poisons such as PAPP, may enable efficient and effective management of target species (Russell et al., 2015). This may be more desirable than aerial application of broad-spectrum anticoagulant poisons such as 1080 as it can provide increased specificity towards target species, reducing the occurrence of non-target mortality.

CONCLUSIONS

High values for all population metrics for robins at Orokonui indicate this site to be effective for the recovery of robins, most likely due to the ability of predator-proof fencing in reducing the competition with introduced pest species for invertebrates as well as eliminating nest predation of nestlings and eggs. This result agrees with a number of other studies indicating robins' high potential for rapid recovery after exclusion of mammalian predators (Taylor et al., 2005, Schadewinkel, 2013). High metrics at Orokonui provide evidence of the potential for the predation factor to mask the limiting effects of any competition factor, as removing the predation factor did not result in adult survival or recruitment rates that differed from those of Silverstream. Despite high population metrics, little colonisation of surrounding areas has occurred, with robins opting to remain confined within the sanctuary. Research into robin territory establishment and territory size inside the sanctuary would be useful in informing efforts to encourage the colonisation by robins of areas outside the sanctuary.

High and increasing predator abundance at Silver Peaks suggests that the 1080 operation designed to reduce predator numbers at this site was ineffectual in the long-term, with predator abundance exceeding pre-drop numbers. Increases in predator abundances post-operation have been known to occur, but it is worth noting that this does not necessarily pose an immediate threat to protected species. Predatory activity, possibly that of stoats, has probably played a key role in the decline in nesting success at this site. High connectivity and a potential competition effect likely played a role in the low adult survival and recruitment rates and high juvenile recruitment rates at this site. This might also represent evidence for the predation limiting factor to mask the effects of the competition factor where the two act simultaneously. Research into combining poison operations with ongoing control regimes might prove useful in providing long-term solutions to pest abundance issues.

Decreases in rat abundance at Silverstream indicate the trapping operation was successful in limiting rodent numbers, but not to key thresholds. Moderate connectivity and high invertebrate abundance is likely to have played a role in the high and stable rates of adult survival and recruitment and juvenile recruitment. Low rates of nesting success were attributed to high rates of stoat predation, this being a potential example of prey-switching as a result of rodent control. Silverstream provides strong evidence for predation as the primary limiting factor. Research into parallel multiple-species control at this site may prove useful in aiding robin and other species' recovery and may help to unravel uncertainties in relation to the complex nature of predation as a limiting factor.

RECOMMENDATIONS

OROKONU

Due to the large and relatively stable nature of the robin population at Orokonui, further intensive monitoring does not seem necessary. Post-season surveys will be a more practical measure of determining population numbers without the need for repeated monitoring. In light of the large, stable and productive nature of this population, it might be useful to consider potential reinforcement translocations of birds from Orokonui to the Silverstream area. This would help bolster robin numbers and ensure that the population is not lost, while also providing some potentially new genetic material to the small population.

I therefore recommend that:

- 1) Repeated monitoring of established pairs and single birds over the breeding season remains on hold (since 2014).
- 2) Walk-through surveys of robin population numbers should be carried out annually to determine whether numbers have remained high.
- 3) Surveys of adjacent areas surrounding the ecosanctuary should be carried out to determine whether establishment has occurred outside the sanctuary.
- 4) Translocations to Silverstream should be considered if the predator numbers at the site are adequately controlled.

SILVER PEAKS

Further monitoring is recommended at the Silver Peaks site in order to continue to investigate the ongoing effects on robins of increases in predator relative abundances after the 1080 operation, and provide information on the population prior to any subsequent poisoning operations. A potentially useful task would be to run similar nest camera placement investigations to Silverstream. This would help to identify the primary nest predator at the Silver Peaks site, enabling a more well-targeted control procedure. Adding stoats to the list of pest species monitored may also prove useful in targeting the primary pest species, although this could be hard to accomplish. Harvest of the study site is underway, and further monitoring of the current banded population might prove difficult. Due to the high connectivity of the Silver Peaks site, it is likely that robins are dispersed throughout the Douglas fir stands. In light of this, further monitoring at Silver Peaks would likely have to be carried out in adjacent, un-harvested areas.

I therefore recommend that:

- 1) Field surveys of adjacent areas within Silver Peaks should be carried out to identify new robin territories and pairs in order to establish a new Silver Peaks population and to check for displaced robins from previous years.
- 2) Monitoring of any new pairs and single birds should continue in order to further assess the effect of increases in predator relative abundance in preparation for the next predator control operation.

- 3) Establishment and implementation of a new CTC line should occur to enable monitoring of predator abundance in the new Silver Peaks area.
- 4) Placement of nest cameras at known nest sites should be carried out to determine the primary nest predator at the Silver Peaks site.
- 5) Stoats should be added to the monitoring regime in order to track changes in their abundance.

SILVERSTREAM

Further monitoring is recommended at the Silverstream site in order to investigate the ongoing effectiveness of the Goodnature™ A24 traps in controlling rodent numbers and enabling robin population recovery and growth. Due to the revelation that stoats are the primary nest predator at Silverstream, monitoring and control of their numbers might prove useful in aiding robin population recovery. Control of the Forest & Bird A24 traps has been transferred to me, which will enable immediate implementation of any recommendations made herein. Camera placement at traps might also provide useful insights into the behaviour of target species towards traps, enabling better targeting of those species.

I therefore recommend that:

- 1) Continual monitoring of established pairs and single birds should be carried out at Silverstream to assess the ongoing effectiveness of the trapping operation.
- 2) The trapping regime should be altered to include stoat lures (not just rat lures) in order to reduce nest predation by stoats.
- 3) Placement of cameras at trap locations should occur to assess target species behaviour towards traps in order to provide better insight into improving trapping systems.
- 4) Ongoing placement of nest cameras should occur in order to monitor any changes in nest predation incidence by identified predators in response to the altered trapping regime.
- 5) Stoats should be added to the monitoring regime in order to track changes in their abundances.

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APPENDICES

Appendix one: Estimated adult survival (with upper and lower confidence limits) of SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2010/11 to 2014/15.

| Site/Year | Estimated Adult Survival (%) | Lower CI (%) | Upper CI (%) |
|---------------------|------------------------------|--------------|--------------|
| Silver Peaks | | | |
| 2009 | 64.8 | 52.6 | 75.4 |
| 2010 | 68.1 | 59.0 | 76.0 |
| 2011 | 71.2 | 64.1 | 77.4 |
| 2012 | 74.1 | 67.5 | 79.8 |
| 2013 | 76.9 | 69.2 | 83.1 |
| 2014 | 79.4 | 69.9 | 86.4 |
| Silverstream | | | |
| 2009 | 80.8 | 70.7 | 88.0 |
| 2010 | 83.0 | 75.6 | 88.5 |
| 2011 | 84.9 | 79.3 | 89.3 |
| 2012 | 86.8 | 81.7 | 90.6 |
| 2013 | 88.4 | 83.0 | 92.2 |
| 2014 | 89.8 | 83.7 | 93.8 |
| Orokonui | | | |
| 2010 | 85.4 | 71.0 | 93.3 |
| 2011 | 87.2 | 75.8 | 93.6 |
| 2012 | 88.7 | 79.7 | 94.1 |
| 2013 | 90.1 | 82.6 | 94.6 |

Appendix two: Estimated proportional juvenile recruitment (with upper and lower confidence limits) of SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2010/11 to 2014/15.

| Site/Year | Estimated Juvenile recruitment (%) | Lower CI (%) | Upper CI (%) |
|---------------------|------------------------------------|--------------|--------------|
| Silver Peaks | | | |
| 2009 | 132.2 | 89.0 | 175.4 |
| 2010 | 107.8 | 80.0 | 135.7 |
| 2011 | 88.0 | 52.3 | 98.0 |
| 2012 | 71.7 | 51.1 | 86.0 |
| 2013 | 58.5 | 39.8 | 75.1 |
| 2014 | 47.7 | 29.8 | 66.3 |
| Silverstream | | | |
| 2009 | 96.5 | 0.9 | 99.9 |
| 2010 | 78.7 | 54.6 | 91.9 |
| 2011 | 64.2 | 47.4 | 78.0 |
| 2012 | 52.3 | 37.0 | 67.2 |
| 2013 | 42.7 | 27.8 | 59.0 |
| 2014 | 34.8 | 20.6 | 52.4 |
| Orokonui | | | |
| 2010 | 249.0 | 188.3 | 309.8 |
| 2011 | 203.1 | 168.7 | 237.5 |
| 2012 | 165.7 | 140.3 | 191.0 |
| 2013 | 135.1 | 107.0 | 163.3 |

Appendix three: Estimated adult and juvenile absolute recruitment (with upper and lower confidence limits) of SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2010/11 to 2014/15.

| Site/Age Class | Estimated recruitment (%) | Lower CI (%) | Upper CI (%) |
|---------------------|---------------------------|--------------|--------------|
| Silver Peaks | | | |
| Adult | 71.4 | 31.6 | 86.8 |
| Juvenile | 37.7 | 4.3 | 45.8 |
| Silverstream | | | |
| Adult | 78.9 | 39.3 | 86.8 |
| Juvenile | 21.2 | 1.1 | 21.8 |
| Orokonui | | | |
| Adult | 58.3 | 30.9 | 82.6 |
| Juvenile | 63.9 | 26.3 | 80.8 |

Appendix four: Observed pairs, nests and apparent nesting success of SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2010/11 to 2014/15. Daily survival rate estimated from the interaction model: Area*Year (see Table 11).

| Site/Year ^a | Pairs Monitored ^b | Nests Monitored ^c | Nests successful ^d | Daily Survival rate (DSR) ^e | Est. nesting success ^f |
|------------------------|------------------------------|------------------------------|-------------------------------|--|-----------------------------------|
| Silver Peaks | | | | | |
| 2010/2011 | 10 | 16 | 12 (75%) | 98.97% | 66.68% |
| 2011/2012 | 12 | 21 | 14 (67%) | 98.44% | 54.12% |
| 2012/2013 | 12 | 20 | 12 (60%) | 98.35% | 52.34% |
| 2013/2014 | 16 | 24 | 10 (42%) | 97.42% | 36.02% |
| 2014/2015 | 17 | 30 | 14 (47%) | 97.17% | 32.61% |
| Silverstream | | | | | |
| 2010/2011 | 10 | 16 | 2 (13%) | 94.97% | 13.34% |
| 2011/2012 | 6 | 12 | 1 (8%) | 91.15% | 2.69% |
| 2012/2013 | 6 | 11 | 0 (0%) | 92.51% | 4.79% |
| 2013/2014 | 10 | 18 | 3 (17%) | 94.35% | 10.36% |
| 2014/2015 | 15 | 29 | 4 (14%) | 94.74% | 12.17% |
| Orokonui | | | | | |
| 2010/2011 | 10 | 18 | 13 (72%) | 98.87% | 64.28% |
| 2011/2012 | 15 | 29 | 24 (83%) | 99.33% | 77.01% |
| 2012/2013 | 41 | 70 | 56 (80%) | 99.17% | 72.12% |
| 2013/2014 | 41 | 68 | 54 (79%) | 99.19% | 72.95% |

^aSite and corresponding year of study. Year represents breeding season between Aug and March.

^bNumber of pairs confirmed to have nesting activity and monitored for corresponding site and year.

^cNumber of nest monitored for corresponding site and year (with a maximum of 2 per pair).

^dNumber of nests to successfully produce at least one fledgling for corresponding site and year.

^eDaily survival rate (DSR) of nests over entire breeding season for corresponding site and year.

^fEstimated nest survival (ENS) of nests based on DSR value raised to the power of 39 (days for completion of nesting cycle).

Appendix 5: Observed pairs, nests and apparent nesting success of SI robins at three sites (Orokonui, Silver Peaks, and Silverstream) for the study period of 2010/11 to 2014/15. Daily survival rate estimated from the interaction model: Area*Year (see Table 11).

| Site/Stage ^a | Pairs Monitored ^b | Nests Monitored ^c | Nests successful ^d | Daily Survival rate (DSR) ^e | Est. nest success ^f |
|-------------------------|------------------------------|------------------------------|-------------------------------|--|--------------------------------|
| Silver Peaks | 67 | 111 | 62 (55.86%) | | |
| Incubation | | | | 97.29% | 34.19% |
| Nestling | | | | 97.87% | 43.16% |
| Silverstream | 47 | 86 | 10 (11.63%) | | |
| Incubation | | | | 95.06% | 13.87% |
| Nestling | | | | 90.23% | 1.81% |
| Orokonui | 107 | 185 | 147 (79.46%) | | |
| Incubation | | | | 98.85% | 63.80% |
| Nestling | | | | 99.13% | 71.06% |

^aSite and nest stage. Data corresponds to all annual breeding season data between Aug and March for entire study period.

^bNumber of pairs confirmed to have nesting activity and monitored for corresponding site and year.

^cNumber of nest monitored for corresponding site and year (with a maximum of 2 per pair).

^dNumber of nests to successfully produce at least one fledgling for corresponding site and year.

^eDaily survival rate (DSR) of nests over entire breeding season for corresponding site and year.

^fEstimated nest survival (ENS) of nests based on DSR value raised to the power of 39 (days for completion of nesting cycle).