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Direct and indirect effects of 1080 on South Island robin populations in the Marlborough Sounds

A thesis presented in partial fulfilment of the requirements for the degree of

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Mara Alexandra Nydegger Bell

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Thesis Abstract

Pest control efforts are vital for the long-term survival of biodiversity native to New Zealand. Improvements to control methods, techniques and application methods are constantly made. In this thesis, I assessed the efficiency of aerial 1080 applications, in regards to survival rates and nest success of adult South Island robins (*Petroica australis*). Applications used a 1 kg sowing rate of 6 g cereal pellets containing 0.15% of sodium fluoroacetate per ha.

SI robins were monitored in Tennyson Inlet, Marlborough Sounds, from 2012/13 – 2016/17. Nest cameras were used through two aerial 1080 applications and a beech mast. Nest outcomes were successfully captured in 83% of 210 nests monitored, identifying the cause and time of failure or the number of fledglings produced. Nests were most commonly preyed on by rats (*Rattus* spp). Rats were also the sole predator to kill brooding or incubating females. We found that adult survival and nest success were both negatively affected by increases in rat tracking rates. Aerially applied 1080 pest control increased nest success by 43% in 2013 and by 47% in 2014.

Our study did not show any direct negative effects of aerial 1080 pest control for SI robins under current application methods and sowing rates. However, the benefits from the pest control efforts were short lived. Two years after the beech-mast-1080 application, rats were causing proportionately more nest failures than during the beech mast season. To achieve long-term population benefits for SI robins, future pest control operations may need to use higher 1080 sowing rates or multiple applications after a beech mast event.

Preface

Origin of research project and thesis data

This research study is part of ongoing 1080 research conducted by Dr Graeme Elliott as part of the Department of Conservation Research and Innovations projects. The data is owned by the Department of Conservation and was collected by volunteers and Department of Conservation staff, including myself.

Project history and context for thesis data

In 2009, around Lake Paringa, South Westland, a research project was established monitoring the effectiveness of aerial 1080 pest control in response to beech masts. The project was the first of a series of projects setup around the South Island taking a more holistic approach to New Zealand pest control monitoring. The project assessed the use of aerial 1080 pest control in response to beech masts, by conducting large scale bird call monitoring, beech seed fall monitoring, intensively monitoring specific species survival and breeding success and monitoring key predator presences.

A sister project in Tennyson Inlet, Marlborough Sounds

A second project ran in Tennyson Inlet from 2010-2017. Three bird species were selected for intensive monitoring rifleman/tītipounamu (*Acanthisitta chloris*), weka (*Gallirallus australis*) and SI robins through two aerial 1080 pest control efforts. Additional data was collected monitoring beech seed fall, rodent, mustelid and possum presence and recording bird calls. The Tennyson Inlet project aimed to provide a greater understanding to aerial 1080 pest control, in a small coastal forest block heavily influenced by beech masts.

Thesis Aim

This thesis aims to assess the efficiency of aerial 1080 pest control on SI robin breeding success and adult survival over five years, 2012/13 - 2016/17, in Tennyson Inlet. By analysing SI robin nest success and adult survival in response to changes in predator pressures, the 2013 beech mast and two aerial 1080 pest control efforts. Additionally, the thesis explores the possible relationship between tracking tunnel and nest camera data and the use of tracking data as a predictor for SI robin nest success and adult survival.

My personal involvement in the data collection and analysis

I have been employed by Dr Graeme Elliott since 2009. This included four summer seasons at Lake Paringa, followed by three years at Tennyson Inlet, where I became project leader, managing staff and volunteers for mustelid, possum, rodent and bird monitoring in 2016/17. I was involved in recording bird calls and predator presence in both projects and led the monitoring of tītipounamu at one of the Lake Paring sites and the SI robin monitoring in Tennyson Inlet. For my thesis I analysed a subset of data from the Tennyson Inlet project, focusing on the nest success and adult survival of SI robin and relevant data on introduced predators collected by myself and my colleagues at Tennyson Inlet.

Photos below show myself overlooking the Tennyson study area and climbing trees to change SI robin nest cameras. *Photos were taken during the 2016/17 season by my volunteers.*

Acknowledgments

I feel a great gratitude for the opportunity given to me by **Graeme Elliott** and his team to write up this research project on behalf of the Department of Conservation. Thank you, Graeme for being a fantastic boss and co-supervisor and an inspirational mentor. Huge thanks to Graemes' team, I have loved working with you; Joris Tinnemanns, Chris Bell, Kirsty Moran, Mitch Bartlett, Tom Allen, Rebecca Davies, Claire Kilner, Marion Rhodes, Bruce Davies, Shane Collins, Ruaridh Davies, Timmothy Dawe, Klayre Cunnew, Tristan Rawlence, Anja McDonald, Robin Blyth, Jenny Long, Christine Hunter, Josh Kemp, Corey Mosen, Sarah Fisher, Ruth Cole, Jason Malham and everyone else that I have accidentally missed off the list!

Completing a Masters has been my dream, and I owe the attainment of that dream to **Doug Armstrong**. Thank you, Doug, for taking me on as a student and being a dedicated supervisor and friend over the years.

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Thank you to Massey University, and the **fantastic librarians** who have helped me many times over the years. A special thank you to **Debbie Leader** with all you help with R. I am also very grateful to have received the **Julie Alley Bursary** assisting me through my first year, and the assistance I received from Massey in regards to health and safety during the field work period in 2017. All **permits** for field work were obtained through the Department of Conservation.

I also want to thank the Tennyson Inlet SI robin population for putting up with all the nest cameras, and banding. I have grown very fond of you all and each spring I fear for the survival of your clutches. I dearly hope that this thesis will help in your species long-term survival!

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Photos used in this thesis were kindly gifted to me by my field volunteers or were taken by myself.

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Chapter One

General Introduction

The 'General Introduction' includes sections on New Zealand, Conservation in New Zealand including pest control and specifically in regards to aerial 1080 treatments and research into aerial 1080 pest control improvements. This information applies equally to all sections in this thesis (**Chapters 3-6**).

An isolated island nation

New Zealand consists of around 600 islands, these make up a total landmass of 268,021 km^2 (Holdaway et al., 2001). New Zealand separated from the supercontinent Gondwanaland ~85mya and drifted away from Antarctica and Australia, becoming geographically isolated ~55 mya (Cooper and Millener, 1993). While the rest of the world's landmass mostly stayed connected, allowing for the dispersal of the newly evolving mammals (Gibbs, 2006, Daugherty et al., 1993). Contrary to this in New Zealand life evolved at its own pace, predominantly free of mammals allowing many Gondwanan species to adapt and survive to the present day.

As a result, New Zealand has one of the highest rates of endemism in the world, with 71% of the 245 once extant bird species being endemic (Brockie, 2007, Holdaway et al., 2001). Other groups of organisms in New Zealand display close to 100% endemism (Losos and Ricklefs, 2009, Lloyd, 2001), with few terrestrial mammals making New Zealand home, other than three species of bats whose ancestors arrived in New Zealand from Australia 50-60 mya (Lloyd, 2001). Instead, birds and reptiles evolved to occupy the niches commonly regarded as those of mammals in other regions of the world (Lee et al., 2010, Losos and Ricklefs, 2009). Many bird species became long-lived, taking a long time to reach sexual maturity and have mammalian features such as a sense of smell and a low reproductive output (Lee et al., 2010, Daugherty et al., 1993).

Many species adapted to the local environment and became irregular breeders, such as kakapo (*Strigops habroptila*) and kaka (*Nestor meridionalis*). Timing their breeding to coincide with mass fruiting events of specific tree species (Powlesland et al., 1992). These irregular mass fruiting events of trees and tussocks are well documented throughout New Zealand (Schauber et al., 2002), and commonly referred to as 'masting'.

A drastic change for New Zealand

Around 1300AD New Zealand's biodiversity started to change drastically with the arrival of humans, and the mammals they brought with them (Innes et al., 2010, Wilmshurst and Higham, 2004, Holdaway, 1989). Initially, around 1200 AD the arrival of carnivores such as dogs and rodents and later pigs and cats, creating the first wave of extinctions (King, 1984). European settlement commenced in the 1800s, leading to a second wave of extinctions, which we are still experiencing today. In total 1,623 new species, including plants and insects have established following introductions by man, now making up a third of all documented species in New Zealand (Vitousek et al., 1997).

Adaptations such as loss of flight, has left New Zealand endemic birds at a disadvantage to the competitive, fast, agile and cunning mammalian hunters (Duncan and Blackburn, 2004). A steep declines and extinctions of native birds, reptiles, molluscs and insects occurred during the 19th and 20th centuries (Tennyson and Martinson, 2006, King, 1984). The native faunal extinctions parallel the Blitzkieg of the "Overkill hypothesis' experienced worldwide as humans spread out of Africa (first wave of extinctions) (Mosimann and Martin, 1975, Martin, 1966). Predator recognition naivety in combination with a low reproductive output meant that the largest species, such as moa (Dinornithiformes), went extinct first, likely due to direct human hunting (Duncan and Blackburn, 2004, Duncan et al., 2002).

Extinctions of smaller New Zealand species followed (second wave of extinctions) (Allentoft et al., 2014, Tennyson and Martinson, 2006, Duncan and Blackburn, 2004, Duncan et al., 2002). The cause of these extinctions can largely be attributed to mammalian predators including rodents, cats, marsupials and mustelids, which have reduced bird diversity by 40-50% (Innes et al., 2010, Tennyson and Martinson, 2006, Holdaway et al., 2001, Holdaway, 1989).

Thirty-one of New Zealand's bird extinctions can be directly attributed to humans, while the remaining extinctions are predominantly due to rodents, cats and dogs, followed by pigs, mustelids and the cascading effects that followed (Tennyson and Martinson, 2006). In New Zealand since human arrival, not a single species went extinct due to habitat loss or climate change alone, demonstrating that the lack of coevolution with mammalian predators has left New Zealand endemics at a disadvantage and we may witness several more extinctions in the years to come.

The impact of introduced mammalian species has been unprecedented in New Zealand's history. Today, only a fraction of New Zealand's native birds are still extant (Tennyson and Martinson, 2006) and a few key predators mustelids, cats (*Felis catus*), possums (*Trichosurus vulpecula*) and rodents are largely to blame for the 77.5% of extant native bird species now in decline, conservation dependent, or threatened with extinction (excluding migratory and vagrant species) (Robertson et al., 2012, Miskelly et al., 2008). Internationally this rates New Zealand as having one of the highest proportions of at-risk species. The war is not over. More native species will continue to go decline unless a solution to controlling or removing introduced predators is found and applied nationwide. Research has clearly identified that introduced predators are the key driver of New Zealand's ongoing declines in species abundance and that the control of these mammals should be the main focus of conservation efforts in New Zealand (Armstrong, 2017, Innes et al., 2015, Elliott et al., 2010, Innes et al., 2010, Atkinson, 2001, Clout, 2001, Holdaway, 1989, King, 1984).

Conservation in Aotearoa, New Zealand

The New Zealand conservation estate is managed by the Department of Conservation (DOC), which was established in 1987 to replace the Forest Service, Department of lands and Survey and Wildlife Services. DOC now manages 8 million hectares of publicly owned land, making up 29% of New Zealand's landmass (Wikipedia, 2017, Craig et al., 2000, Cessford and Dingwall, 1997). DOC estate is predominantly located in the South Island's western and southern areas where land has low economic value (Cieraad et al., 2015, Craig et al., 2000, Cessford and Dingwall, 1997).

DOC is funded on an annual basis; these funds are then split into several sectors including biodiversity which in turn includes pest control. In 2017/18 the pest control conducted by DOC was given the title 'Battle for our Birds' and was allocated a total of $42 million, up from $21 million in the 2016/17 season (*"DOC's Budget 2017 explained"*, 2017).

With annually changing budgets that cover only a fraction of the conservation estate, many areas are left to their own devices (Craig et al., 2000). As a result, introduced mammals multiply, causing catastrophic population declines and local extinctions of vulnerable native species in the majority of the conservation estate.

There is an urgency to develop and implement long-term, cost-effective pest control strategies that have clear conservation gains. These techniques need to be successful in large and small areas of land if we are to bring an end to the current wave of extinctions. Research and ongoing monitoring of pest control efforts is essential to improving pest control methods.

Pests in Aotearoa, New Zealand

So what is a pest?

By definition it is an annoying or destructive species that causes damage to goods, stock or crops. Meaning that a 'pest' causes monetary loss to humans, if not directly then indirectly. In a New Zealand conservation setting, the term 'pest' and 'pest control' is used to refer predominantly to introduced mammalian species while 'predator control' is the term commonly used to refer to the control of introduced mammalian predators.

DOC views all introduced mammalian species as 'pests' if they have an adverse effect on indigenous biota as stated by the Conservation Act. But to the general public these naturalised mammals can also have a value; in sport, culture, companionship and as an income, such as red deer for meat and possums for fur. Under the Wildlife Act 1953 most introduced mammals are in one of two classes, "unprotected" or "noxious", and therefore subject to the Noxious Animals Act 1956 (now the Wild Animal Control Act). This brings the responsibility of pest control to the landowner who has 'ownership' of the 'pests' on their land and likewise gives DOC the responsibility for pest control on 29% of New Zealand landmass and management of over 220 islands of 5 ha or more ("*Offshore Islands*", 2018).

From the 1860s to 1890s the Acclimatisation Societies around New Zealand attempted to introduce some 130 or more bird species. Additional to birds were a huge range on mammals including monkeys (Sir George Grey), African lions (Canterbury Acclimatisation Society), red deer (Otago Acclimatisation Society) and even more were on the 'to be released' list but never got the go ahead such as mountain lions *(Puma concolor).* Over time humans learned both through research and through consequences of releases, such as the release of mustelids in New Zealand and cane toads *(Rhinella marina)* and red foxes *(Vulpes vulpes*) in Australia*,* that certain species if not already established in the wild, need to be restricted or completely banned from ever entering new countries. Today 81%

(N=31) of the introduced and established mammal species in New Zealand are actively managed as pest species (Parkes and Murphy, 2003).

In 2002, the New Zealand government ruled that ferrets *(Mustela putorius furo)* were to be made illegal to sell, distribute or breed in captivity under the Biosecurity Bill (Lee, 2002). Other species like hamsters (Cricetinae) are illegal to be imported, sold or purchased in New Zealand by private citizens, with the only member of the hamster family the Syrian research hamsters LVG (SYR) being present in New Zealand and restricted to lab experiments for medical research.

Governmentally induced restrictions on new introductions and regulation on pests that are already present decreases the threat to vulnerable native species and crops. Accidental introductions still occur, but with one of the strictest quarantine procedures in the world, New Zealand manages to capture most unwanted intruders through its border security services, protecting native wildlife, agriculture, horticulture, fisheries and its human population from diseases, invertebrate and animal pests.

In New Zealand, not everyone sees introduced mammalian predators as pests. There is strong opposition against the control and or eradication of certain species in New Zealand, mainly cats, ungulates and pigs. Even ferrets and rats are well loved pets. Not only can species declared as 'pests' be viewed as well-loved pets such as cats, but the public's approach to pests can be broadly divided into four viewpoints; 1) idealists, 2) traditionalists 3) pragmatists 4) commercial users (King 1996 referenced in (Parkes and Murphy, 2003)). As Bomford and O'Brian (1995) point out, there is a need for public support for the long-term success of eradication and pest control. Without public support, government bodies such as DOC are unable to carryout pest control efforts (Parkes and Murphy, 2003, Bomford and O'Brian, 1995). In a survey, 71-94% of people viewed smaller mammals as pests but not larger ones such as deer (Fraser, 2001). Overall, the general New Zealand public sees mammalian species such as stoats and possums as having a negative impact on New Zealand, but when questioned about control techniques, poison applications were generally less favourable to trapping and least of all new genetic manipulation techniques (Fitzgerald et al., 2002).

This shows that the general public have a weak understanding of the true situation that New Zealand faces. Trapping alone, is simply not a feasible pest control option for most of New Zealand's land area, especially the conservation estate due to its sheer size, remoteness and terrain. It is not feasible to think that eradication or pest control efforts can be conducted right across New Zealand by trapping (Linklater and Steer, 2018). Only selected areas can logistically and financially be made more habitable for indigenous biota with trapping and hunting alone. In an ideal world, a combination of different ethical control techniques could be applied simultaneously or alternatively in all areas to capture the individuals that have immunity, higher poison tolerance or trap shyness capturing individual with traits that would allow them to survive single technique pest control efforts.

Pest control and predator-prey interactions

Pest control is a complicated science where man interferes and manipulates the numbers of one or several species with the aim of suppressing their population. There are no black and white answers and no one cure for all habitats or pest species. Each action has a multitude of effects and a cascading effect on the interactions of a food chain. The control of top predators can lead to meso predator population explosions, while the removal of meso predators may lead to prey switching by top predators. After the removal of rats (meso predator), stoats (top predator) have been found to prey switch from a 74.3% rat diet to 23.4% substituting the remainder of the diet with birds (Murphy and Bradfield, 1992). Prey-switching is often a short-term-phenomena, and the impact could be anything from drastic to minimal.

Control means that the population of the target pest species is decreased, though not removed completely. As a consequence, the surviving individuals often act differently after the pest control effort than before. Additional challenges may develop such as increased immigration or breeding success. It is only over time and a multitude of pest control efforts in different habitats and years that we can understand our exact effects on the ecosystem as a whole. Overall, we know that in New Zealand the main predators of most native species are mustelids, rodents and possums (often called the trio). Pest control efforts such as use of sodium fluoroacetate (1080) can successfully target all three of these predators simultaneously.

Aerial 1080 pest control

Large-scale pest control, in an areas with difficult to access terrain, is dependent on aerial poison applications as the method of control (Morgan et al., 2015). There are few poisons available to control all three key predators (possums, stoats and rats) in a single poison application, two of which are brodifacoum and 1080. The application of 1080 has become the preferred option and the only option sanctioned by the Department of Conservation (DOC) for ongoing pest control, this is due to the readily biodegradable properties of 1080 and the lack of bioaccumulation issues (Wright, 2011).

Improvements to aerial 1080 pest control

Sowing rates, size and composition of baits, helicopter application method versus fixed wing aeroplanes, and timing of operations have been the key areas of focus, over the past ~50 years (Morgan, 1994). Aiming to increase kill rates of target species with minimal damage to waterways, native fauna and flora. In the past baits were composed of carrot chaffing, caused high levels of by-kill in species such as North Island robins (*Petroica longipes*) (Powlesland et al., 1999). Baits used today, are composed of compressed cereal and laced with 0.15% sodium fluoroacetate. They are applied at lower sowing rates of 1-5 kg/ha (Warburton and Cullen, 1995), these two simple improvements have reduced by-kill rates of native species (Westbrooke et al., 2003). Pellet sizes most commonly used today are either 6g or 12g and all 1080 applications are now conducted by helicopter with a pre-feed to train the target species onto the baits and increase the kill rates of rats and possums (Nugent et al., 2011).

Helicopters have an increased accuracy and flexibility for the boundaries of sowing areas (Warburton and Cullen, 1995) and when fitted with spreader buckets, sowing rates can be adjusted. It is now common practice that all applications are conducted with the aid of a navigational support systems which increases kill rates at lower sowing rates by decreasing the gaps between the swaths (Morgan, 1994). There are still short comings where further improvement into poison application is needed, such as a way of measuring the pellet dispersal and taking into account wind and topography to decrease the chances of accidentally leaving strips or pockets of bush without poison baits. Buckets and Helicopters are also not standardised throughout New Zealand, and each helicopter company and pilot have their own preferences.

Moving away from possum focused management to targeting rats

Large-scale aerial 1080 possum control commenced in the 1956 and its ability to suppress possum populations is well documented, these used sowing rates ranging between 1-40 kg/ha (Nugent et al., 2010, Warburton and Cullen, 1995, Hughes, 1994). The effectiveness of these poison applications depends not just on application methods and bait quality, but can also be affected by time of year, bait acceptance and density of the target species (Morgan et al., 2000).

Until recently the timing of operations has largely been focused on possum densities and many conservation estates are still managed with possum number as an indicator. This is not ideal, especially in areas influenced by beech masts. All forest systems contain the trio of mammalian predators, though only rats can breed multiple times a year. Rats therefore produce the greatest population fluctuations in response to environmental factors such as masts, making them a better indicator species to guide pest control efforts.

Recent research has identified aerial 1080 poison applications as an effective multi-pest-control tool (J. Kemp unpublished data 2007 referenced in (Elliott and Kemp, 2016,) Gillies and Pierce, 1999, Murphy et al., 1999). Aerial 1080 can effectively reduce rats and stoats through secondary poisoning (Gillies and Pierce, 1999, Murphy et al., 1999), though this has not been shown to have reliable outcomes between years and sites (Elliott and Kemp, 2016).

Beech-masts are leading to rodent and mustelid population explosions

New Zealand forests systems are strongly affected by annual variation in breeding, flowering and seeding cycles. Many plant species are influenced by weather patterns as the trigger to flower and fruit on mass at irregular intervals (Schauber et al., 2002, McKone et al., 1998). This includes the five New Zealand beech species (*Fuscospora fusca, Fuscospora cliffortioides, Fuscospora solandri, Fuscospora truncata, Lophozonia menziesii*). These fruit every 2-6 years and fruit on mass (mast), every 2-12 years across whole regions. Such fruiting events allow for increased productivity of insects and birds alike. Beech forest ecosystems are believed to be at least in part reliant on these high yield years to maintain native species abundance and diversity. Masting podocarp species such as rimu (*Dacrydium cupressinum*) have a two-year delay to the same weather patterns as beech. Today, mast events temporarily lift the carrying capacity of introduced rodents (Pech et al., 2015, McQueen and Lawrence, 2008). Meaning that mixed beech-rimu forests experience high yields over two successive years lifting both rat and stoat numbers over a four-year period.

With around 4 million ha of native forest containing some beech, and around 2 million ha being pure beech forest, understanding predator-prey relationships and the effectiveness of predominantly aerial 1080 pest control is vital to the long-term management of native flora and fauna. Rodents are the key drivers in mustelid population growths (Pech et al., 2015, King, 1983) and if suppressed immediately after a beech mast may also be able to stop stoat population explosions. The combination of both rodent and mustelid population explosions in successive years is known to cause local extinctions of vulnerable native bird species. It is therefore a key area for more research.

NIWA climate models used to map New Zealand into high medium or low seeding zones are issued annually (Elliott and Kemp, 2016). These models have allowed for forward planning and increased the ability for conservationists to execute timely pest control operations. Without these models, efficient pest control targeting rodents would not be possible, as time is needed to source funding and prepare staff in high risk areas for the treatment operations to give timely relief to native species before extreme predator pressures are experienced (Elliott and Kemp, 2016)

Rodents as the target pest species

Since 2000, the pest control management approach in the Conservation Estate has shifted away from possum numbers, to focus on rodent numbers instead. This change of management practise has been gaining momentum after rodents were linked to local extinctions of vulnerable species such as mohua (*Mohoua ochrocephala*) on Mt Stokes (Macalister, 2002, Macalister, 2001). By focusing on rodents, stoats are managed automatically due to secondary poisoning (Murphy et al., 1999) and possums are managed to low target densities, as long as poison applications occur every 6 years. This shift towards targeting rats as has changed the way aerial 1080 treatment areas have been prioritised on an annual basis and how the operations have been executed in regards to toxic load and timing of application.

The difficulty with targeting rodents is that their population densities fluctuate dramatically between seasons, years and vary depending on forest type and altitude and can recover very quickly through immigration and rapid population growth (Fitzgerald et al., 2004).

Rats reach their reproductive age at 4-5 weeks old and have a gestation period of 21-23 days (19-21 for mice) with 6-12 pups (3-14 for mice). Females can become pregnant while still nursing their previous litters which are weaned around 21 days. Mice can easily have up to 10 litters a year and all rodents have larger litter sizes when food is readily available, also decreasing their age of sexual maturity and time between litters. Rodent densities vary with habitat type and food availability. Densities of ship rats in forested gullies, faces or ridges were consistent and ranged between 6.5-7.8 rats/ha (Brown et al., 1996b). However, higher densities of rats were found in forested areas then in scrubland, this was not the case for mice who were more abundant in scrubland than in forested areas (Brown et al., 1996b).

Competition for food and predation between mice, kiore, ship and Norway rats occurs though their territories do not always overlap. Whereas Norway rats and kiore don't climb trees, ship rats and mice are arboreal. Ship rats tend to be found in native forest in New Zealand whereas Norway rats tend to be found in forest margins and farmland where they overlap with kiore and outcompete them (Innes, 2001). Mice have been found throughout New Zealand, they compete with kiore and are prey to ship and Norway rats (Innes, 2001). Mice hold small territories and 100% kill rates are therefore hard to achieve by largescale aerial pest control or trapping networks. Rats in comparison have the ability to move from low food source areas to high food sources areas making them more versatile at exploiting seasonal fruit and seeding events as well as colonisation after pest control events. Home ranges of ship rats are predominantly stable, but as individuals are removed, neighbours will quickly absorb the vacant territories (Innes and Skipworth, 1983).

Population explosions in response to high food and migratory behaviour are therefore site specific and hard to predict. Rodent population explosions termed as mouse and rat 'plagues' have been observed in beech forests around New Zealand during the autumn/winter and subsequent spring following a beech mast event, from where they spread into farmland and townships (Blackwell et al., 2001, King and Moller, 1997).

What is sodium fluoroacetate (1080)

Poisons kill or negatively affect a species by attacking, dissolving, blocking or changing one or several aspects of the organism's metabolic, nervous, digestive or respiratory processes, or other aspects of its physiology. This is referred to as direct poisoning or primary poisoning. This occurs from direct ingestion or coming into direct contact with the poison. Most poisons are not able to target a single species or individual, instead they affect a whole range of non-target species/individuals through direct ingestion, contact or secondary poisoning. Secondary poisoning refers to a sub-lethal or lethal dose to an individual, caused by the ingestion of one or multiple species/individuals that have themselves ingested lethal or sub-lethal doses of a particular poison. Cumulative secondary poisoning is common for poisons that do not biodegrade, such as the famous cases between the 1940 - 70s killing many American raptors with the accumulation of Dichlorodiphenyltrichloroethane (DDT). Biodegradation of poisons, refers to the breakdown of the compound to organic particles naturally found in the environment.

In New Zealand several different types of pesticides are in regular use, including sodium fluoroacetate (FCH_2CO_2Na), commonly called 1080. Wright (2011) evaluated pesticides commonly used in New Zealand as the Commissionaire to the Environment for the New Zealand Government in 2015. Her team compared the poisons on several grounds: ethical, impact on non-target species and the poisons persistence in the ecosystem, as well as the amount of research conducted. Jan Wright (2011) report concludes that aerially distributed 1080 was the best pest control option available for large-scale rodent, mustelid and possum control in New Zealand. Other smaller studies have called specifically for more research into how humane 1080 is as a pesticide (Sherley, 2007), while others support more ongoing monitoring to keep improving 1080 bait operations (Eason et al., 2011).

Sodium fluoroactetate is naturally present in many plants including native/endemic species of New Zealand, but is more common in plants of Western Australian. Sodium fluoroactetate can be present at very high doses, making plants toxic if consumed. Due to this natural occurrence and long evolutionary history, sodium fluoroacetate can be metabolised and broken down by bacteria into naturally occurring smaller compounds and eventually individual elements. This allows the re-absorption of sodium fluoroacetate into the natural environment as a nutrient, rather than a toxin, such as is the case with manufactured compounds including plastics or other pesticides such as Brodifacoum.

Sodium fluoroacetate was first artificially manufactured in Belgium and later in Germany as an insecticide, then as a rodenticide during WWII. Subsequently, in the 1970s, 1080 became banned in many countries such as the United States of America due to its high toxicity in mammals and lack of an antidote (Calver and King, 1986). Most countries have a large number of native or endemic mammals that are vulnerable to 1080 and at times used 1080 as a control method for less desired mammals such as wolves and coyotes in North America. This is not the case in New Zealand, where the only terrestrial extant endemic/native mammals are two species of bat. This makes the use of 1080 highly efficient in New Zealand, because all introduced mammals cause a degree of destruction to native and endemic fauna and flora, and are susceptible to 1080, while insects, reptiles, fish and birds are less susceptible (Wright, 2011).

Sodium fluoroacetate blocks the Krebs cycle, starving main organs and muscles of energy, with high energy demanding organs such as the heart and nervous system collapsing first, eventually leading to death (Calver and King, 1986). Once ingested, 1080 is metabolised into fluorocitrate ($C_6H_7FO_7$). Fluorocitrate subsequently inhibits two important enzymes, aconitase and succinate dehydrogenase which stops cellular respiration (Krebs cycle) (Calver and King, 1986). Cellular respiration is the mechanism that all aerobic organisms use to produce energy. It involves four steps, Glycolysis, Pyruvate oxidation, Krebs cycle and the Oxidative phosphorylation process resulting in ATP (adenosine triphosphate) production. The Krebs cycle is also referred to as the Citric acid cycle. Here food molecules such as sugars/carbs, fats and proteins that have been metabolised into citric acid are oxidised, creating ATP. ATP is essentially the energy product needed for building, transporting and movement of nutrients between and into cells that sustain life. If the Krebs cycle stops, the surrounding organs starve and eventually the organism dies.

Sodium fluoroacetate (1080) toxicity

Like most poisons, 1080 toxicity varies between species and individuals with a huge range of LD50s (median lethal dose). The weight, size and metabolic rate of different species and individuals affects the rate of metabolism of the poison, altering the toxic loading needed to cause death. Species like possums have a LD50 of 0.47 – 0.79 mg/kg, which means that per kg body weight they need between 0.49 - 0.79 mg of sodium fluoroacetate for a 50% chance of death (Sherley, 2007). Mice (*Mus musculus*) have an LD50 of 8.33 mg/kg, dingos' (*Canis lupus dingo*) 0.10 mg/kg and an Australian blotched blue-tongued lizard (*Tiliqua nigrolutea*) 336.4 mg/kg (Sherley, 2007). Many insects do not metabolise 1080, but simply ingest it and excrete it again, making them temporarily toxic to 1080 vulnerable species.

As 1080 baits get in contact with moisture (soil, atmospheric and precipitation), microbial bacteria are able to move into the cereal pellet and start the digestion process of the sugars, starches and sodium fluoroacetate (Fairweather et al., 2013). In regards to pest control, the optimum time that baits remain 'active' in the field is from 3 - 5 days, (*active, meaning that moisture levels have remained low enough that the toxicity level of the baits have not been compromised*). If the period without rain is less than three days, then there is a chance that the target species have not found the baits or managed to eat a lethal dose by the time baits start to decrease in toxicity. If the forest remains dry for a long period, then there is an elevated chance that pellets are consumed by non-target species including deer, insects and birds, increasing the chance of unwanted by-kill.

DOC uses the guidelines of 100 ml of precipitation having fallen, before the forest is deemed safe for species like birds from poison bait ingestion (Bowen et al., 1995). Sodium fluoroacetate has the potential to persist in dry conditions (Fairweather et al., 2013), or in possum carcasses. The decomposition rate of 1080 occurs very slowly within a larger carcass, and after 75 days toxicity levels can still be still fatal to scavenging dogs (Meenken and Booth, 1997). This means that a warning will remain on forests for up to 4 months to eliminate any secondary poisoning to dogs and any possible ingestion of game meat by humans that may contain traces of sodium fluoroacetate ("*Specifications for Products Intended for Human Consumption*", 2016).

Water contamination after aerial 1080 pest control operations

Bacteria are responsible for the decomposition of sodium fluoroacetate by breaking down compounds. Bacterial activity and abundance vary with moisture and temperature, resulting in lower sodium fluoroacetate breakdown in very dry and cold conditions. Moist forest conditions contain plentiful bacteria that invade the pellets and break down sodium flouroacetate both within the pellet and once leached from the pellet.

The leaching of sodium fluoroacetate from a RS50 cereal pellet decreases rapidly during the first 20 mm of rainfall. Once 100 mm of rain has fallen, only barely detectable levels of sodium fluoroacetate remain (Bowen et al., 1995).

Aerial 1080 pest control operations became routine practice in the 1970s, initially using small fixed wing aircrafts and later helicopters. With the use of aerial distribution, the concern for waterways being polluted became a growing issue among the New Zealand public. In the 1970s it was predicted that aerial 1080 use for pest control is unlikely to have a significant impact on waterways and urban water supplies (Peters, 1975). By the 1990s more sophisticated water testing technologies had been developed. Two 1080 applications, one at 14 kg/ha and the other at 5 - 6 kg/ha, both revealed no contamination of the waterways monitored for up to 6 months after these possum control operations (Eason et al., 1992).

Even with such early field-based evidence and modelling of 1080 in waterways there has been ongoing speculation raising concerns among the general public. As recently as 2016 a research study made headlines with results being exaggerated to the public. The study involved fasting trout before feeding them 1080 baits in captivity. The study found above recommended standard food safety limits of 1080 in the trout's tissues (MPI, 2016). Under these extreme conditions sodium fluoroacetate was found at doses of up to 18 mg/kg of trout, which could translate into a maximum of 0.16 mg/kg of 1080 transfer into a child (23 kg), if the child ate over 280 g of trout within 24 h of the 1 kg trout having consumed a 12 g bait pellet (MPI, 2016). To put this in perspective a lethal dose (LD50) for a human is 0.7 - 10 mg/kg (Fairweather et al., 2013). The study concluded that it is not realistic that such an incident would occur in the wild. This is due to the trout having to consume a whole bait pellet and be caught within hours of the aerial 1080 application and the child would have to eat a reasonably big portion of the fish (MPI, 2016).

In the US, studies have reported that mosquito larvae die at 0.025 - 0.05 mg/L, while fish were not affected at 13 mg/L stream saturations (Eisler, 1995). Additionally, other studies have found that 1080 bait pellets lose 50% of their active compounds within 5 h of falling into the water (Suren, 2006).

Contamination of waterways does occur, though at such low levels and with such a high decomposition rate, that it is near impossible to cause harm to fresh water species and or humans. Freshwater species by-kill rates have been explored both in the field and in the lab, and have shown that low levels of toxicity can occur under forced conditions, but in the wild these results could not be replicated; rather studies have shown no significant by-kill rates. Contamination to a toxic level of waterways is therefore very unlikely, and the chance of toxicity levels reaching lethal doses for humans is extremely unlikely.

By-kill of native birds during 1080 pest control operations

In the 1960 and 70's when aerial 1080 operations were commenced in New Zealand, large numbers of native birds were killed. Placental mammals are the most sensitive group to 1080 poisoning, followed by marsupials and birds. Birds are therefore at risk of both primary and secondary poisoning, as many birds are insectivorous. Secondary poisoning for insectivorous birds is believed to be very low, even with LD50 rates of 0.6 - 2.5 mg/kg for a single dose or an LD50 of 0.5 mg/kg when given daily doses over a month to more sensitive birds (Eisler, 1995).

The most striking difference between these early 1080 pest control operations and the operations conducted today, are the sowing rates and the bait configuration. Originally, carrot shavings were pressed into baits and laced with 1080, resulting in high levels of chaff and small bait fragments. Today the baits are made up of cereal meal containing high sugar content and cinnamon to make them hard and less likely to break up. The early New Zealand carrot bait operations had anywhere from 0.2 - 23% of the baits broken up into less than 5 mm particles, a size ideal for birds to ingest (Powlesland et al., 1999). Sowing rates of above 10 kg/ha or even over 20 kg/ha were generally used (Powlesland et al., 1999), while today it is common practise to use much lower sowing rates ranging between 1 - 3 kg/ha with similar target pest cull results (Elliott and Kemp, 2016).

These new approaches to using lower sowing rates, and high density 'impact-resistant baits' are still being tested for their by-kill rates. A recent study reported undetectable by-kill rates for native birds (Jones, 2016).

Primary and Secondary 1080 poisoning of non-target birds

Sodium fluoroacetate poisoning in birds can occur through direct ingestion of pellets (primary poisoning) or by ingestion of still active 1080 compounds via prey (secondary poisoning). The metabolism and breakdown of sodium fluoroacetate is dependent on the digestive system of the animal that ingests the compound and the accessibility of bacteria. Primary poisoning is to some degree more manageable from a pest control perspective. Modifications to the presentation of the bait can change these ingestion rates. If sown into an environment by hand, helicopter or placed into a range of different bait stations the accessibility and attractiveness will change. Similarly, the smell, texture, colour and density of the bait changes the probability of ingestion. A lot of effort has been invested over the years into this sector, and very positive results obtained, making pest control a lot safer in regards to by-kill rates.

Secondary poisoning for a compound like 1080 is a lot harder to manage. This is because the breakdown of the compound is largely determined by factors that are not in our control, such as adverse weather events (unseasonable snow droughts). Additionally, the rate of insect ingestion of 1080 is a factor that cannot be controlled. Insects have a different digestion system and often do not break down sodium fluoroacetate. This means that they become a carrier, containing low rates of active 1080. It is therefore possible for birds to ingest active 1080 by eating insects. Research in this area has repeatedly found that these probabilities are very low. However, Powlesland et al. (1999) reported finding a North Island robin (NI robins here after) containing 3.80 mg/kg of active 1080 in its tissues. The gizzard of the NI robin only contained insect exoskeletons and no carrot chaffing/bait. Implying that the cause of death was likely secondary poisoning. The same study did find that when the percentage of chaffing was reduced from 23 % to 2.5 % the by-kill rates of NI robins was reduced from 54.5 % to 8.6 % (Powlesland et al., 1999). This implies that by-kill rates of birds like robins may be related to direct ingestion, unless insects are more likely to ingest 1080 from carrot chaffing rather than from the undamaged bait pellets. To our knowledge this has not been tested, but either way, studies have shown a strong relationship with the percentage of chaffing and by-kill rates of birds during 1080 operations (Powlesland et al., 1999).

Current aerial 1080 pest control strategies in beech forests

Pest control operations in beech forests now aim to suppress both rodent and possum populations through primary poisoning and stoats through secondary poisoning. This strategy reduces operational costs as it is only implemented in response to beech mast events. Most 1080 operations are assessed through tracking rates of target species (rodents and mustelids) which DOC aims to reduce to below 5%. A rat tracking rate below 5% in beech forest is expected to supress rat population to below 30% until the next surplus food event (beech mast). It is therefore vital that more research is conducted assessing this approach and the effect it has on predator populations dynamics in the long-term.

South Island robin/kakariwai (*Petroica australis*)

In New Zealand the Petroicidae family (Australasian robins) consist of the 'tomtits' (*Petroica macrocephala*) and the 'robins'. Robins are represented by three species, each species having a clearly defined non-overlapping ranges (Higgins and Peter, 2002). The NI robin or toutouwai (*Petroica longipes*) is restricted to the North Island, while the black robin or kakaruai (*Petroica traversi*) to the Chatham Island group and SI robin or kakariwai is split into two sub-species with *Petroica australis rakiura* restricted to Stewart Island and *Petroica australis australis* to the South Island (Robertson, 2007).

SI robin were described by Anders Sparrman in 1787. According to sub-fossil records, SI robins were found throughout the South Island but are now restricted to the upper and lower South Island regions (Scofield and Stephenson, 2013, Higgins and Peter, 2002), due to habitat modification and introduced mammalian predators (Innes et al., 2010, Higgins and Peter, 2002, King, 1984).

SI robins are classified as unthreatened and of least conservation concern (IUCN, 2017, Robertson et al., 2012, Miskelly et al., 2008). They weigh 22 - 47 g and measure 18 cm head to tail and have sexually distinct plumage as well as easily recognisable juvenile plumage until post juvenile moult occurs within their 1st year (Scofield and Stephenson, 2013).

Territories are strongly defended by males, and pairs generally have a single pair bond that often lasts for several years (Higgins and Peter, 2002). Their natural average life expectancy is largely unknown as it is dependent on predation rates, though the oldest records of robins include individuals between 14 - 17 years (*personal communications*) (Heather and Robinson, 2015).

SI robins have open cup nests usually in main trunk or branch forks in a range of tree species and heights (Higgins and Peter, 2002). SI robin nest in a wide range of tree species as well as on boulders, cliffs and on the ground. Nesting in branches or trunk forks at <10 m, this makes nests accessible for monitoring purposes (Scofield and Stephenson, 2013, Powlesland et al., 2000).

Robin females are the sole parent involved in nest building and incubation (Boulton et al., 2010, Higgins and Peter, 2002). SI robins lay 2 - 4 eggs between September and January. For New Zealand robins each nesting attempt consists of 1 - 14 days of nest building followed by 2 - 4 days of laying, 17 - 20 days of incubation and 19 - 22 days at chick stage (Higgins and Peter, 2002, Armstrong et al., 2000, Powlesland, 1983). SI robin chicks are altricial and nidicolous and raised by both male and females (Scofield and Stephenson, 2013, Higgins and Peter, 2002). They can have up to three clutches (Powlesland, 1983). Most commonly, SI robins raise two chicks per clutch to fledging age (Powlesland, 1983). If nests fail pre-fledging they are able to re-nest within two days (Higgins and Peter, 2002) although generally a longer interval is observed (*personal communications*). Once fledged, fledglings are supported on average for 5 weeks (4.5 - 10), predominantly by the father (Scofield and Stephenson, 2013, Higgins and Peter, 2002). Females can re-nest while the male raises the fledglings of the previous clutch to independence, he then re-joins the female to help feed the next clutch once hatched (Higgins and Peter, 2002).

Robins are identified as intelligent, curious, relatively tame, they can easily be trained to respond to an auditory cues (McLean et al., 1999, Brown, 1997, Powlesland, 1997) and are vulnerable to the trio of mammalian predators, making them an ideal study bird in regards to monitoring pest control impact (Innes et al., 2010, Powlesland et al., 2000, Brown et al., 1998).

Forest birds can be divided into three main groups based on nesting behaviour; 'ground', 'cavity' and 'cup' nesting birds. Each of these groups faces different predator pressures. It is commonly accepted that cavity nesting birds are more at risk of adult mortality by predation during incubation and brooding phases of the nest attempt, if the predator can get into the cavity (O'Donnell, 1996). Ground nesting birds also face threats from ground based mammals such as cats, pigs and hedgehogs, but commonly have more than one exit route from the nest decreasing adult mortality unless flightless (Sanders and Maloney, 2002). By nesting in the open cup nests, eggs and chicks are vulnerable to all three key tree climbing predators; ship rats, stoats and common brush-tail possums, as well as extreme weather events during the nest attempt (Jones, 2016, Powlesland et al., 2000, Brown et al., 1998). As a cup nesting bird, SI robin adult survival is expected to be similar between the sexes, breeding or non-breeding individuals, though females are eaten in low numbers on the nest (Brown, 1997). Outside the breeding season SI robin can easily escape from predators, while during nesting they are vulnerable to all three key introduced predators (Innes et al., 2010).

SI robin nest success and adult survival rates can be compared at different predator densities and sites as well as previous research studies with reasonably large data sets. In regards to assessing the effectiveness of aerial 1080 pest control which often occurs part way through the SI robin breeding season, data on survival and nest success can be compared before and after the pest control treatment effects, and similarly between years since pest control efforts in the absence or presence of beech mast events.

Why study SI robins to measure 1080 pest control effectiveness

NI and SI robins frequently feature in pest control research. They are easy species to monitor and so provide a reliable measure of the effectiveness of the pest control effort. With site and pair fidelity to relatively small territories and curious nature, they make ideal study species that are vulnerable to the three key target species that pest control efforts today aim to minimise (Innes et al., 2010, Powlesland et al., 2000, Brown, 1997). Research reporting figures on productivity in response to translocation to predator free environments also often feature robins (Miskelly and Powlesland, 2013, Dimond and Armstrong, 2007). Additionally, survival data have been collected for genetic research from closed and open populations (Boessenkool et al., 2007). This means that there is a range of survival data already available for NI and SI robins to compare 1080 survival data to.

It is only in recent years that studies can reliably account for beech mast events and take into account the effect of these food booms, that temporarily lift the carrying capacity of many New Zealand forests for introduced predators. By combining both predator densities and productivity and survival rates, a better understanding of the efficiency of pest control efforts is gained. This has only been possible in recent years with advancements in technology such as field cameras that have allowed the identification of nest predators.

Comparing data from control efforts covering many different habitats under different environmental and climatic conditions provides, over time, an invaluable understanding of the ecosystem and food-chain reactions following a pest control effort. Providing guidance to future pest species management and ultimately to an introduced predator free New Zealand.

Tennyson Inlet in the Marlborough Sounds

Tennyson Inlet in the Marlborough Sounds supports mature native forest from its peaks to the coast and hosts one of New Zealand largest recorded rimu and ancient kahikatea. The forest cover changes with altitude from a lowland mixed podocarp forest to predominantly beech forest. The tip of the Stanley peninsula is in private ownership and the Nydia walking track weaves through the base of the peninsula. The native forest cover of the nearby peaks, Lookout Point and Editor Hill are bordered by farmland and a small settlement each before reaching the coast and to the West and South by pine (*Pinus radiata*) forestry and open farmland.

The various ecosystems host a wide range of mammals including goats (*Capra aegagrus hircus*), deer (*Cervus elaphus*), feral pigs, feral cats, possums, ferrets (*Mustela putorius*), weasels (*Mustela nivalis*), stoats, ship rats, mice and likely Norway rats. The surrounding forest is home to 18 species of native forest birds and little blue penguins (*Eudyptula minor*). Five species of shags and a range of introduced farmland birds including small numbers of magpies (*Gymnorhina tibicen*) and California quail (*Callipepla californica*) also use the forest margins.

In Tennyson Inlet predator vulnerable bird species are largely restricted to altitudes of above 500 m. Only few individuals and small pockets of breeding pairs of SI robins, brown creepers (*Mohoua novaeseelandiae*) and rifleman (*Acanthisitta chloris*) are found at lower altitudes that are dominated by introduced bird species and native species that are more tolerant to introduced mammals or hold larger territories such as tui (*Prosthemadera novaeseelandiae*) and kereru (*Hemiphaga novaeseelandiae*).

The Mount Stanley area has had irregular pest control treatments for many years. Records were easily obtainable since 1995 showing that the 2013 aerial 1080 application was a first for that area, with previous pest control operations using other poisons or hand laying and bait stations (Table 2.1, page 34). The Mount Stanley pest control efforts have aimed to protect Powelliphanta (*Powelliphanta hochstetteri obscura*) which are endemic to the area. Pig and possum hunters use the easy access areas periodically and DOC manages goat numbers to reduce browsing, though goats were seen in high numbers from 2012 - 2017.

The study site consists of two areas within Tennyson Inlet, Marlborough Sounds. The Mount Stanley Peninsula at 4300 ha received aerial 1080 pest control on 2 November 2013 (non-beech mast year) and 22 November 2014 (beech mast year). The non-treatment (control area) at c.4900 ha ranged from Lookout Peak (900 m) to Matapehe (935 m) via Editor Hill (1032 m) including the Harvey and Tuna Bay catchments. Both sites reach to 1000 m altitude at their highest points and SI robin populations were found to be present in patches in these higher altitudinal areas. The SI robin study areas consisted of around 100 - 200 ha each. The forest is continuous between the sites and is beech dominated with some rata (*Metrosideros umbellata*) at higher altitudes, containing a mature mix of podocarp along the coastal areas including kahikatea (*Dacrycarpus dacrydioides*), rimu (*Dacrydium cupressinum*) and tawa (*Beilschmiedia tawa)*. Four species of beech are present and altitudinally distributed, with hard beech (*Fuscospora truncate*) commencing around 300 m and predominantly silver beech (*Lophozonia menziesii*) and red beech (*Fuscospora fusca*) at higher altitudes with patches of mountain beech (*Fuscospora cliffortioides*) on the peaks. The distance from peaks to coast ranged from c. 2 - 6 km allowing for dispersal of predators and SI robins between forest types and sites.

The terrain is very steep and rocky in places. There is continuous forest cover from Mount Stanley to the Nelson Lakes National Park and the Murchison Mountains. State Highway 6 is the only gap in forest cover, possibly acting as a deterrent to dispersal as robins are reluctant to cross open terrain (Brehme et al., 2013, Richard and Armstrong, 2010, Huey, 1941). Dispersal of native bird species is plausible and a probable annual occurrence through juveniles' dispersal. Coastlines, roads, public walking tracks and ridge top trails are likely highways for reinvasions of mammalian predators (Brehme et al., 2013, Getz et al., 1978, Huey, 1941). Immigration of predators into the study sites after a pest control efforts is highly likely in Tennyson Inlet.

Chapter Two

General Methods

The 'General Methods' include sections on the study site, aerial 1080 treatments and data collection as this information applies equally to all sections in this thesis (**chapters 3 - 6**). Specific information on data collection methods and analysis applicable to one chapter only will be presented as needed.

DOC Aerial 1080 research projects

This study was part of several research projects assessing effects of aerial 1080 (sodium fluoroacetate) treatments in response to beech masts in different parts of New Zealand: Lake Paringa, South Westland (2009 - 2016), Tennyson Inlet, Marlborough Sounds (2010 - 2017), Tararua Ranges (2011 - 2013) and Kahurangi National Park (2017 and ongoing) (**Fig. 2.1**). All studies and sites used a modified Before-After-Control-Impact (BACI) design with more than one treatment.

Continuous mature native forest (predominantly beech forest) was divided into different pest control treatment areas. This included a non-treatment area which received no pest control with the exception of hunters (deer & pigs) and trappers (possum). One or more different areas in each site received different pest control methods. The different treatments included differences between aerial 1080 treatment applications in regards to timing and or sowing rates or ground treatment in the form of trapping (using DOC200 or DOC250 kill traps).

The Tennyson Inlet study contains two areas, a non-treatment and a treatment area. The treatment area received two aerial 1080 applications: one in 2013 in the absence of a beech mast event and one in 2014 in response to high rodent numbers after a beech mast event. Both aerial 1080 applications used the same sowing rate (1 kg/ha), pellets (RS5 cinnamon lured and laced with 0.15% 1080 6g cereal baits) and contractors to carry out the poison application. The study area is situated in and around Duncan Bay Township at the base of Tennyson Inlet, Marlborough Sounds (**Fig. 2.2**).

This thesis contains some of the data collected during the seven years of research in Tennyson Inlet and focuses on the SI robin adult and nest survival data in response to predator numbers.

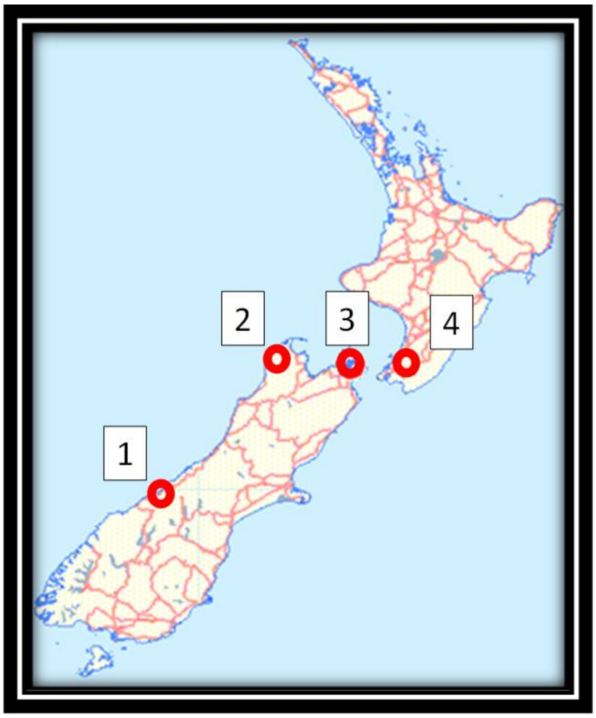


Fig. 2.1: Sites of studies assessing effects of aerial 1080 drops on native bird populations: 1) Lake Paringa study in South Westland, 2) Kahurangi study, 3) Tennyson Inlet study in the Marlborough Sounds and 4) the Tararua study.

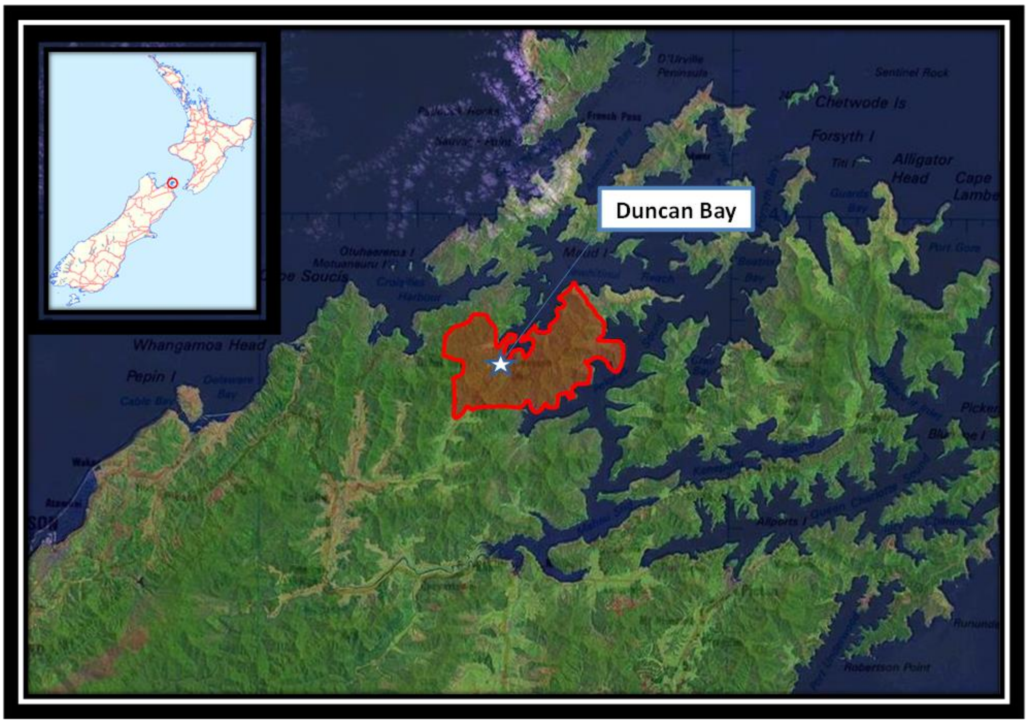


Fig. 2.2: Aerial 1080 research site around the settlement of Duncan Bay at the base of Tennyson Inlet, Marlborough Sounds, South Island, New Zealand.

Tennyson Inlet South Island robin study site (Thesis data)

The Tennyson Inlet SI robin study sites were situated at the most accessible locations at the edge of the SI robin range in both the treatment and non-treatment areas (Fig. 2.3). The treatment area could only be accessed by foot (2 h) or a combination of foot and boat (1 h), while the non-treatment area was accessible by road.

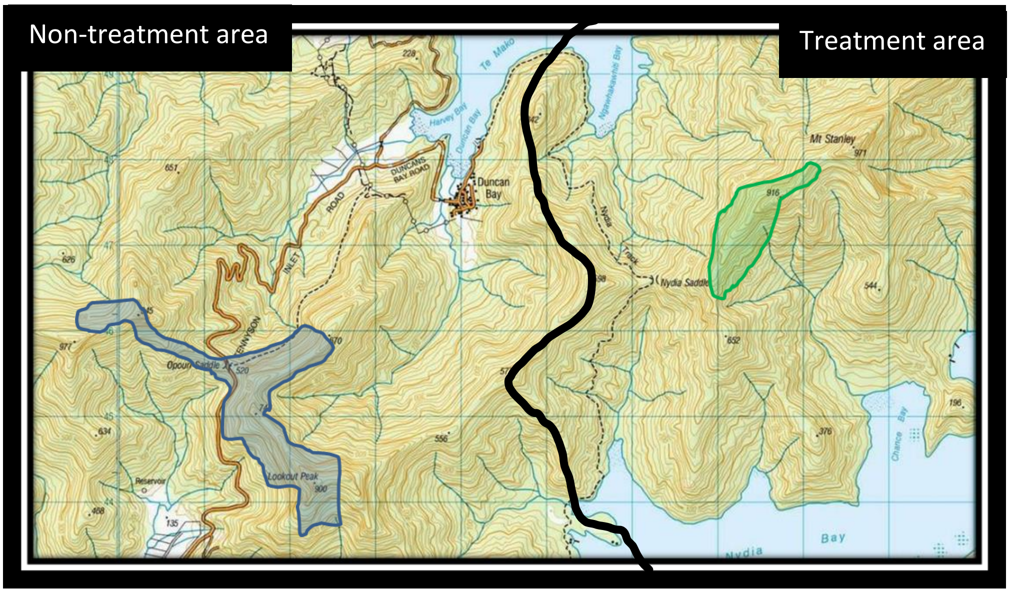


Fig. 2.3: The South Island robin study sites are found within the treatment area (right) and the non-treatment (left) areas of the Tennyson Inlet study situated around the township of Duncan Bay at the base of Tennyson Inlet, Marlborough Sounds. The SI robin treatment area is shown in green and non-treatment area in blue. The non-treatment area had to be a lot larger as robins were scarce and the terrain harder to traverse.

Throughout the Tennyson Inlet area, SI robins were very sparsely and sporadically found at altitudes below 500 m but were at higher densities above 500 m. Due to difficulties with access, a bivy (small bivouac hut) was mounted in 2012 and remained in the SI robin treatment area until the end of the 2015 breeding season. Monitoring in 2016 was conducted with the aid of a tent camp at the same location as the bivy. The SI robin non-treatment area was situated on either side of the Opouri saddle and accessed by the Opouri saddle road, no overnight camping was needed. Ridge trails and old trap lines were used for faster travel within the areas.

A minimum of 10 nests were monitored in each year for each area, and all adults if possible were colour banded for individual recognition. Juveniles were sometimes banded if capture was easy, but due to dispersal potential out of the study areas this was not prioritised. All birds were captured with drop traps, baited with meal worms (*Tenebrio molitor*). These traps were hand triggered by pulling a string, closing the trap over the bird. Birds were individually targeted and captured and processed straight away. No handling or captures were made unless the bird needed banding or had banding related issues such as a lost or damaged colour band or a foot injury. Birds were not treated if ‘natural’ injuries were observed such as after a nest predation event.

The 2013 aerial 1080 operation carried out in Tennyson Inlet was specifically funded for this research project. The aerial 1080 application in 2014 was funded as part of the ‘Battle for our Birds’ campaign, in response to a mass beech fruiting event affecting most of the South Island and parts of the North Island (Fig. 2.4).

All forest systems have different productivity levels and pest re-incursion rates. This affects predator-prey relationships, the severity of the impact caused by introduced predators on native bird species, and the speed at which predators return after pest control efforts. Tennyson Inlet was selected to represent a small coastal mixed forest system strongly influenced by beech mast events that spanned from sea level to 1000 m altitude. Using this type of ecosystem, this project aims to evaluate the benefits gained by the aerial 1080 application for SI robins in the absence and in response to beech mast events.

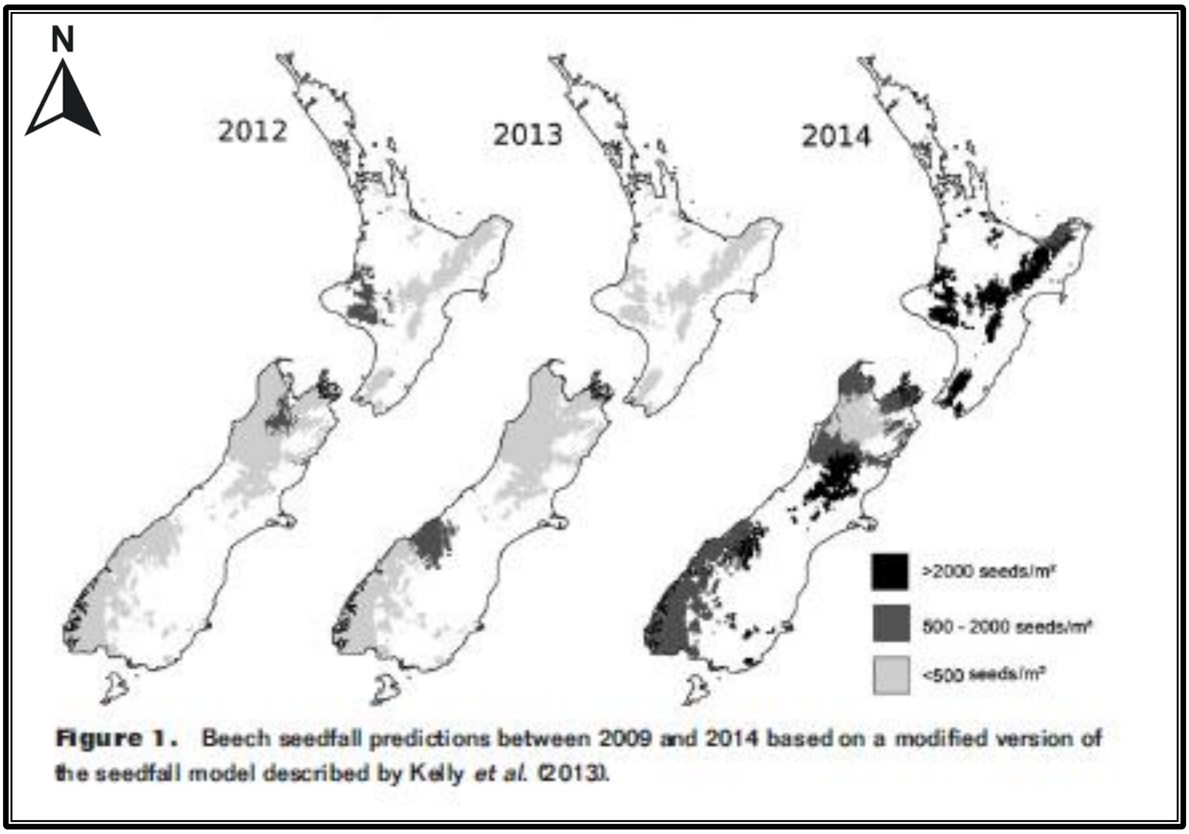


Fig. 2.4: New Zealand wide beech seed fall comparison between 2012, 2013 and 2014 taken from Elliott and Kemp (2016).

History of pest control in the Tennyson Inlet study site

No aerial 1080 applications were conducted in Tennyson Inlet before the 2013 application (Table 2.1). Both the treatment and the non-treatment areas remained open to the public for tramping, possum trapping, pig hunting and deer stalking over the course of the study period. The treatment area received poison warnings after each treatment application. These warned the public of the poison hazard to dogs and advised people not to hunt within the area for up to six months after the application as per DOC and Nelson/Marlborough council regulations.

Table: 2.1: Recorded recent history of pest control efforts in Tennyson Inlet from 1995-2008 targeting possums, goats, deer and pigs. No aerial 1080 applications or pest control efforts targeting rodents had been conducted pre 2013.

Site	Area	Area (ha)	Year	Date	Control method	Type	Prefeed	Target species
Editor Hill	Non-treatment	200.00	1998	15/06 - 14/07	Feratox Bait Stations	Encapsulated pellets (Cyanide 500g/kg)	no	Possum
Editor Hill	Non-treatment	200.00	1999	24/09 - 26/09	Feratox Bait Stations	Encapsulated pellets (Cyanide 500g/kg)	yes	Possum
Tawa Bay	Treatment	1800.00	1999	18/06 - 02/08	Cynide hand laying, trap & bait bags	Cholecalciferol peanutbutter hand paste (8g/kg) & Cyanide paste (	no	Possum
Mt. Stanley	Treatment	48.00	2001	06/10 - 18/05	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Editor Hill	Non-treatment	200.00	2002	08/01 - 12/02	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Mt. Stanley	Treatment	48.00	2002	15-Apr	Pestoff Exterminatior Paste	1080 paste (1.5g/kg)	no	Possum
Mt. Stanley	Treatment	48.00	2002	10/10 - 20/10	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Tennyson Inlet	Both	2157.00	2002	18/07 - 26/07	Feratox & 1080 bait bags & 1080 hand laying	1080 paste (1.5g/kg), 0.15% 1080 cereal pellets 0.5kg/ha & encapsu	no	Possum
Mt. Stanley	Treatment	48.00	2002	15/04 - 16/04	Bait stations	Pestoff Exterminatior Paste (1080 1.5g/kg)	no	Possum
Mt. Stanley	Treatment	48.00	2003	28/10 - 02/11	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Editor Hill	Non-treatment	243.00	2003	29/09 - 16/10	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Editor Hill	Non-treatment	200.00	2003	17/02 - 13/03	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Editor Hill	Non-treatment	200.00	2004	12/10 - 22/11	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Editor Hill	Non-treatment	243.00	2004	19/04 - 24/05	Pesticide hand laying	Cholecalciferol peanutbutter hand paste (8g/kg)	yes	Possum
Ngawhakawhiti Bay	Treatment	790.00	2005	22/06 - 24/06	Pesticide hand laying	0.15% 1080 cereal pellets 0.5kg/ha	no	Possum
Editor Hill	Non-treatment	243.00	2005	23/05 - 23/06	Pesticide hand laying	0.15% 1080 cereal pellets 0.5kg/ha, no prefeed	no	Possum
Mt. Stanley	Treatment	48.00	2005	19/09 - 24/09	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
South of Nydia Saddle	Non-treatment	647.00	2005	01/05 - 24/11	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Editor Hill	Non-treatment	243.00	2005/06	21/11 - 13/12	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Mt. Stanley	Treatment	48.00	2006	11/09 - 19/09	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Editor Hill	Non-treatment	243.00	2006	13/11 - 08/12	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Editor Hill	Non-treatment	243.00	2007	01/10 - 02/11	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Mt. Stanley	Treatment	48.00	2007	13/11 - 21/11	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Mt. Stanley	Treatment	48.00	2008	15/12 - 24/12	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Harvey and Duncan Bay	Non-treatment	1600.00	2008	23/07 -04/09	Cholecaliciferol, trap & hand laying 1080	Cholecalciferol peanutbutter hand paste (10%), 1080 1kg/ha	yes	Possum
Editor Hill	Non-treatment	243.00	2008	18/11 - 10/12	Traps	Victor #1 steel jaw leg hold traps	NA	Possum
Site	Area	Area (ha)	Year	Date	Control method	Type	Prefeed	Target species
Tennyson Inlet	Both	12321.00	1995/96	NA	Hunting	Judas and ground hunting	NA	Goat
Tennyson Inlet	Both	12321.00	1996/97	NA	Hunting	Judas and ground hunting	NA	Goat
Tennyson Inlet	Both	12321.00	1998/99	NA	Hunting	Judas and ground hunting	NA	Goat
Tennyson Inlet	Both	12321.00	1999/00	NA	Hunting	Judas and ground hunting	NA	Goat
Tennyson Inlet	Both	12321.00	2000/01	NA	Hunting	Judas and ground hunting	NA	Goat
Tennyson Inlet	Both	12321.00	2001/02	NA	Hunting	Judas and ground hunting	NA	Goat
Tennyson Inlet	Both	12321.00	2003/04	NA	Hunting	Judas and ground hunting	NA	Goat
Tennyson Inlet	Both	12321.00	2004/05	NA	Hunting	Judas and ground hunting	NA	Goat
Tennyson Inlet	Both	12321.00	2005/06	NA	Hunting	Judas and ground hunting	NA	Goat
Tennyson Inlet	Both	12321.00	2008/09	NA	Hunting	Judas and ground hunting	NA	Goat
Tennyson Inlet	Both	12321.00	2009/10	NA	Hunting	Judas and ground hunting	NA	Goat
Tennyson Inlet	Both	12321.00	2010/11	NA	Hunting	Judas and ground hunting	NA	Goat
Tennyson Inlet	Both	12321.00	2011/12	NA	Hunting	Judas and ground hunting	NA	Goat
Site	Area	Area (ha)	Year	Date	Control method	Type	Prefeed	Target species
Tennyson Inlet	Both	12321.00	2006/07	NA	Hunting	Ground hunting	NA	Pig, Deer & Goat
Tennyson Inlet	Both	12321.00	2007/08	NA	Hunting	Ground hunting	NA	Pig, Deer & Goat

Active 1080 periods during this research study

In Tennyson Inlet rainfall is not the only form of moisture at higher altitudes. Peaks commonly capture passing clouds, increasing humidity to the point of precipitation (wet deposition). This is very common for altitudes above 700 m in Tennyson Inlet (*personal observations*, **Plate 2.1**).

Plate 2.1: Mount Stanley captures and holds cloud cover at its peak for most of the year, creating a very moist forest with moss clad trees and branches. This additional moisture contributes to the precipitation level within the South Island robin study area, though is not captured by the National Institute of Water and Atmospheric Research (NIWA) weather stations situated close to sea level.

NIWA rainfall data for the Tennyson Inlet study area was available from three weather stations within 20 km of Mt Stanley (**Fig. 2.5**). Mt. Stanley is the highest peak within the treatment area of the Tennyson Inlet study area. We estimated the number of days of active 1080 in the forest system based on rainfall data from these three closest NIWA rainfall data sites Okiwi Bay, Maud Island and Pelorus Sound (Agent number: 12635, 13585, 4232). All NIWA sites are near sea level and therefore likely underestimating the wet deposition of the treatment area on Mt Stanley (**Table 2.2**).

The mean rainfall between the three sites over both years is 2.7 mm/day and therefore a period of 36.7 days should receive around 100 mm of rain. The number of rainfall events that were needed to reach a total of 100 mm precipitation ranged from 5 to 16 rain events. The observed median rainfall from the three sites over the two years is 34 days and the average 40.4 days ranging from 33 to 62 days. Thirty-five days was therefore decided on as a safe estimate for the active sodium fluoroacetate period immediately after each aerial 1080 operation, though this does not include additional moisture from low clouds.

Table 2.2: Locations of the three closest National Institute of Water and Atmospheric Research (NIWA) rainfall stations to Mt Stanley. Mt Stanley central within and the highest peak within the 1080 treatment area of the Tennyson Inlet study site.

Name	Agent Number	Latitude (dec.deg)	Longitude (dec.deg)	Height (m)	Distance from Mt Stanley (km)
Okiwi Bay	12635	-41.11300	173.66700	3	8
Maud Island	13585	-41.02900	173.89600	15	15
Pelorus Sd, Crail Bay	4232	-41.10308	173.96406	13	17

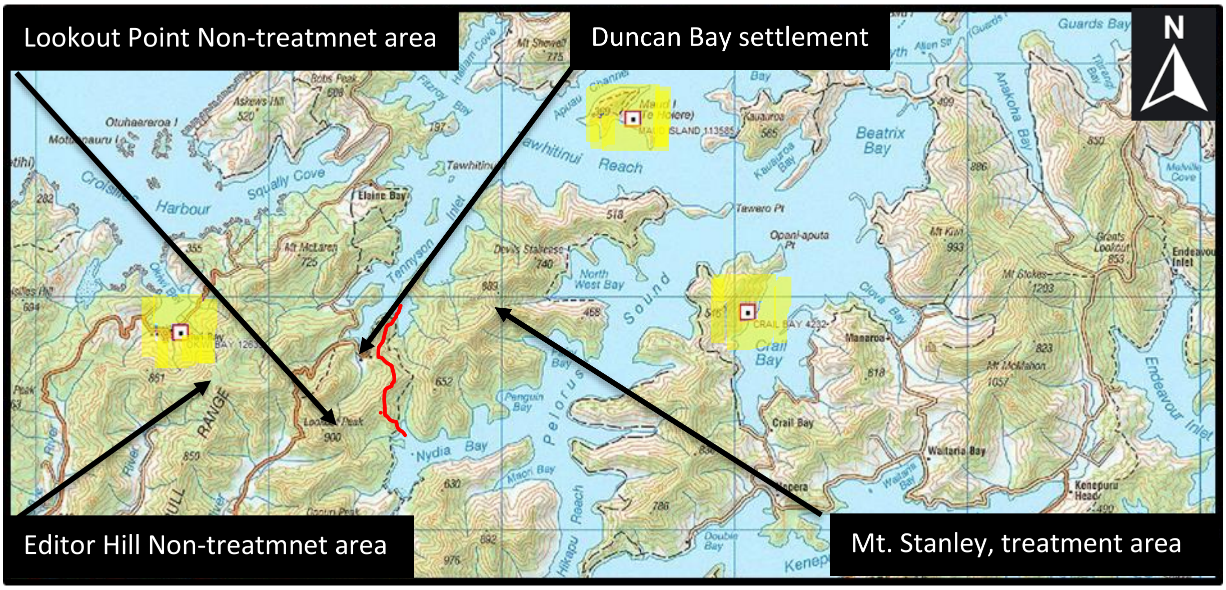


Fig. 2.5: The location of the three National Institute of Water and Atmospheric Research (NIWA) rainfall data collection sites highlighted in yellow. These three NIWA sites were used to predict the active 1080 periods after the pest control operations in 2013 and 2014. The red line on the map indicates the boundary between the treatment and non-treatment area of the study and the highest peaks are named within the treatment and non-treatment areas.

Aerial 1080 applications

The first 1080 application occurred in 2013 and was conducted in the absence of a beech mast, the second in 2014 was conducted in response to a beech mast. Due to 1080 opposition in the local community the initial planned poison application for 2012 was delayed by a year and reduced the data collection opportunities under a treatment effect for the SI robin study.

With a nationwide beech mast event occurring in 2014, a second poison application was conducted around Mt Stanley (treatment area). This provided a great opportunity to compare the efficiency of an aerial 1080 poison applications in two consecutive years, in 2013 without the elevated rodent numbers followed by 2014 with elevated rodent numbers.

The combination of data from Tracking Tunnels (TT) with data from SI robin Estimated Nest Success (ENS) and adult SI robin survival rates provides an excellent opportunity to give a complex insight into the pest control efficiency and predator-prey interactions. TT provided information on the speed and severity of population growth responses of the target predator species (rats (*Rattus rattus* and *Rattus norvegicus*), stoats (*Mustela erminea*) and possums (*Trichosurus vulpecula*)) while ENS and survival data provide an estimation of SI robin population health between and within study years.

A pre-feed was carried out to train pest species onto the cereal pellets and increase kill rates (Nugent et al., 2011). The pre-feed baits were 6.0 g cereal pellets containing a 0.30% cinnamon lure and contained no dye. The toxic application was conducted with 0.15% 1080 laced cereal pellets (RS5) pressed into 6.0 g cereal baits which are dyed green and containing 0.30% cinnamon lure. The sowing rate of the pre-feed and toxic baits was 1.0 kg/ha in both years (2013 and 2014).

Both poison applications were delayed and were not conducted until November. A delay in application for smaller treatment areas is not uncommon. Aerial applications can only be effectively performed for the pre-feed and the toxic applications in mild dry weather conditions. The hierarchical application order is designed to prioritise aerial applications for larger sites and areas of highest biodiversity value in years where multiple pest control operations are conducted. A relatively small site like Mt Stanley at 4300 ha with low biodiversity value and high reinvasion potential due to edge/peninsula effect, public walking tracks, roads and habitat fragmentation, is ranked low priority. Mt Stanley is expected to have a shorter-lived benefit from the pest control effort compared to larger site, where the pest control benefit is often longer lived and therefore returns an overall higher economic value per ha treated.

In 2013 4300 ha were treated with non-toxic cereal pellets on 20 October and 2 November with toxic cereal pellets using two jet rangers in moderate NW wind (**Fig. 2.6**). The same company did the 1080 application in 2014 with the non-toxic pre-feed being carried out on 10 October, followed by the toxic application on 23 November using one Hughes 500D helicopter and one Squirrel AS 350 helicopter in light SE wind (**Fig. 2.7**). Both operations used AgNavGPS navigation systems and a GPS tracking system monitored by DOC to map the helicopters' flight paths. Though opposition had been voiced in the community in 2012, 2013 and again in 2014, these protesters appeared to be a very small minority and none were present on the day of either application.

Additional challenges were encountered during both applications. In 2013 one of the helicopters lost a loaded bucket of pellets into the forest. DOC staff were sent to collect all the spilled pellets by hand, and these were subsequently removed. In 2014 high clouds started gathering around Mt Stanley peak in the afternoon, and aerial sowing was ceased before completion. Fifteen days after the aerial application, three DOC staff were sent to hand sow the remaining area with 1080 pellets, completing the poison application. The hand-laid area of 12.4 ha was outside the SI robin study area and no birds were likely to venture into the hand-laid area. Movement of rodents, mustelids and possums between the delayed hand-laid area and the aerial application area was very likely, and was expected to decrease the efficiency of the poison operation in regards to overall predator kill rates, though the extent of this decrease is unknown.

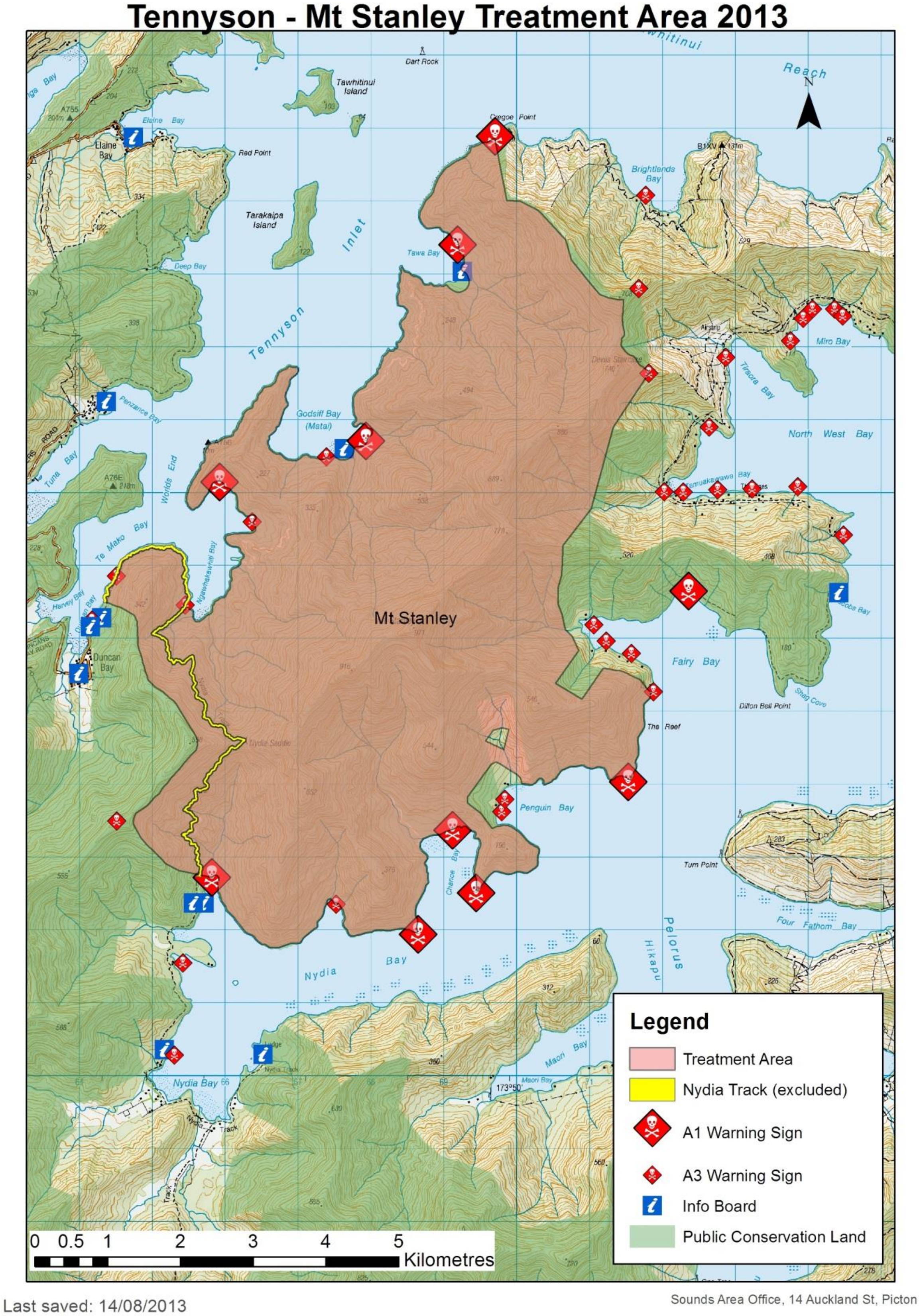


Fig. 2.6: The aerial 1080 treatment area on Mt Stanley, Tennyson Inlet for the 2013 pest control operation in the absence of a beech mast. Map provided by the Department of Conservation.

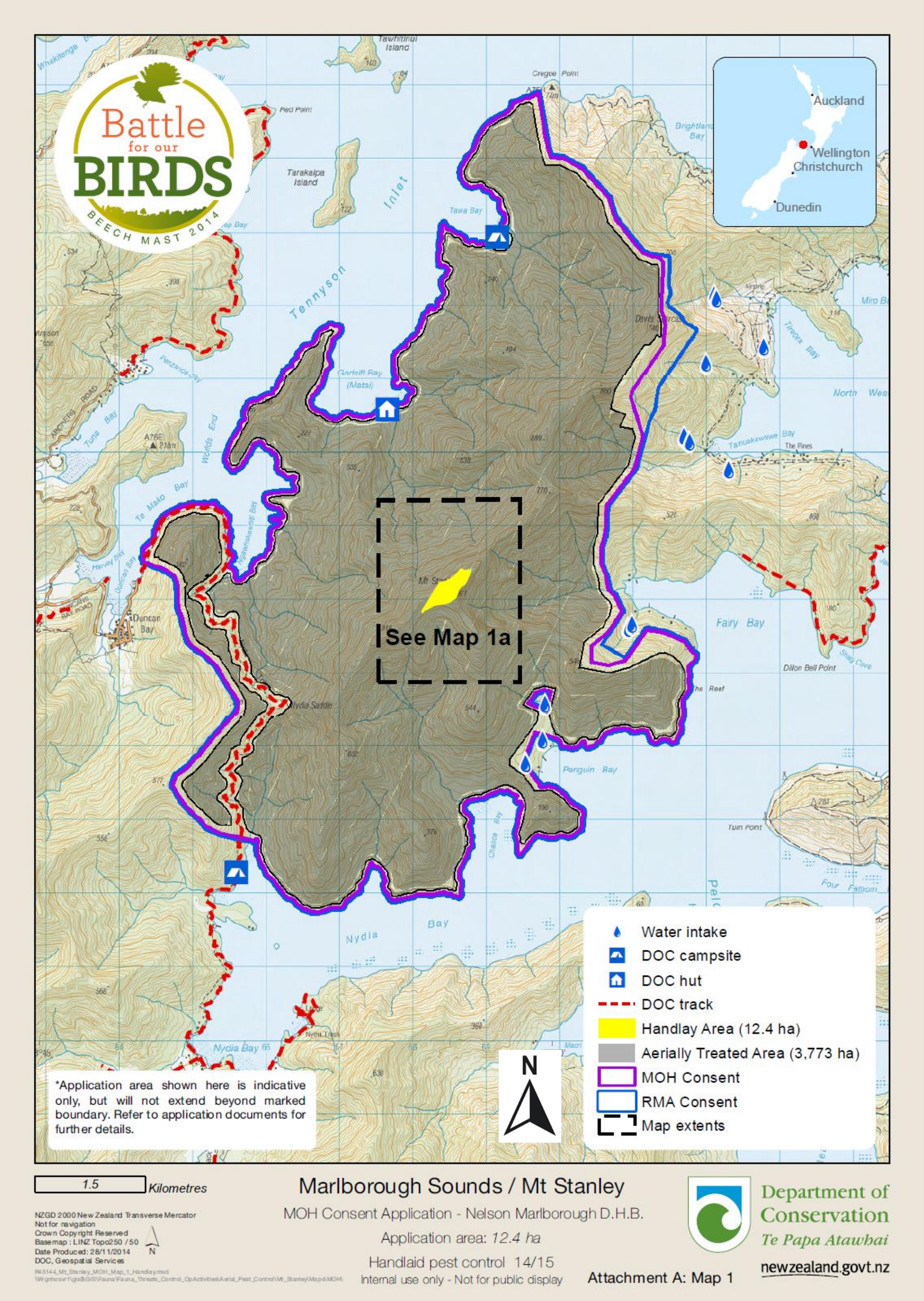


Fig. 2.7: The aerial 1080 treatment area and hand-laid 1080 area on Mt Stanley, Tennyson Inlet for the 2014 poison operation in response to a beech mast. The total area covered 3939 ha, and included 153.6 ha of buffer areas, 3773 ha that received aerially applied 1080 (23 November) and 12.4 ha that received hand-laid 1080 (8 December). Map 1a is not provided in this thesis as the exact boundaries of the hand laid area is not relevant to this study. Map provided by the Department of Conservation.

Theoretical movement of predators between treated and untreated areas after a poison application

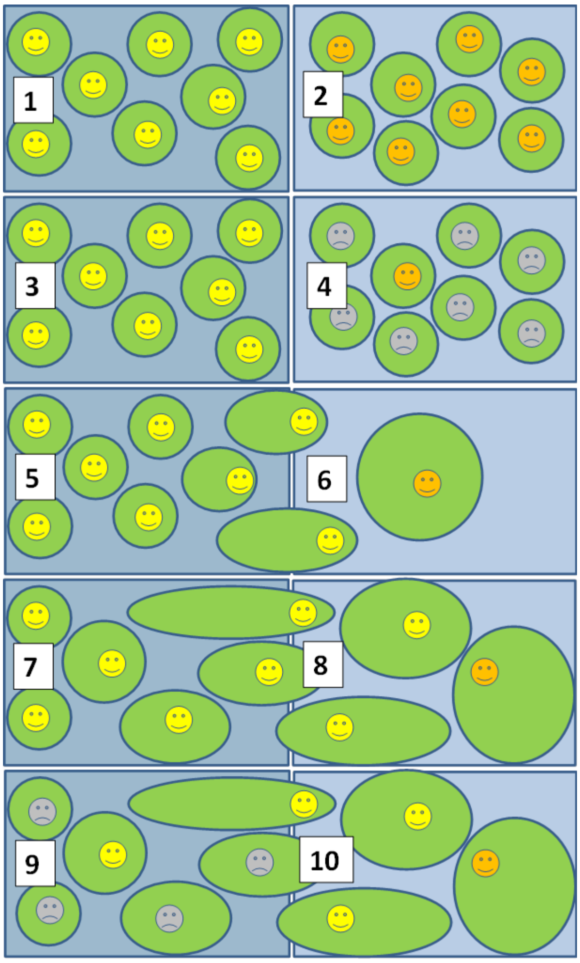
There is evidence supporting the idea that emigration from a high density area to a low density area occurs when resources become strained e.g. (Loe et al., 2009). For many species such as possums, stoats and foxes (*Vulpes vulpes*), dispersal is largely associated with the movement of young individuals on an annual basis as they leave their natal habitats in search of their own territories even in the absence of high density pressures (Byrom et al., 2015, Thomson et al., 2000, King, 1994, King and McMillan, 1982a). Dispersal distances are often sex linked, such as in possums and red deer (Byrom et al., 2015, Loe et al., 2009), and in some cases positively linked to natal density dependence (DD), where individuals of a certain sex will move further if the natal density is high.

Dispersal can also be aided by forest modification. Possums have been observed to disperse along roads and waterways and are more likely to disperse greater distances in modified forests with heavy deer, goat and pig browsing Pracy 1975 referenced in (Cowan, 2000). While re-invasion after pest control efforts is increased for possums if the forest habitat is small, surrounded by agricultural land with a high edges rate (Cowan, 2000).

Predators have also been observed to extend their territories, dispersing into neighbouring territories as they become vacant, though this has been predominantly demonstrated for insects. Mice, rats, stoats and possums likely display similar behaviours when they arrive on an island that is not inhabited with their own species. Unusual behaviour has been observed such as rodents and possums roaming over vast distances unheard of under 'normal' conditions (*personal communications*) and (Russell et al., 2005).

With a delay between the hand laying of 1080 baits and the aerial 1080 application in 2014, pest species had time to disperse from the untreated areas into the treated areas with newly vacant territories. This allowed predators to avoid poison bait encounters and increased the overall percentage of surviving pests within the whole of the treated area (aerial and hand) (**Fig. 2.8**). In theory this should have decrease the efficiency of the 1080 operation conducted in 2014. However, it was thought, as the missed area was only 12.4 ha (0.003% of the treated area) that it would not cause a noticeable difference in the post-treatment tracking rates of target pest species (rats, mustelids and possums).

Theoretical dispersal from one area to another neighbouring area when pest control is conducted



1) Non-treatment or missed area; at a 80% density
2) Treatment at a 80% density
3) Non-treatment or missed area; at a 80% density
4) After treatment (10% survival)
5) Non-treatment or missed area; at a 60% density due to emigration and dispersal into vacant territories
6) Treated area lifted to 30% density due to immigration

The two areas equalise, with larger territories resulting in higher productivity and survival rates due to increased food supplies.

7) Non-treatment or missed area; at a 50% density
8) Treatment at 40% density
9) Missed area treated with delay; decreasing density to 10%
10) Treatment at 40% density due to immigration survivors

Overall kill rate and treatment efficiency is reduced because predators had time to emigrate to empty territories.

Fig. 2.8: A hypothetical depiction of the movement of pest species between a treated and an untreated area, during a pest control operation the proportion of survivors and emigrations is fabricated for the purpose of this demonstration only. Showing changes in density as a small percentage of individuals survive the pest control operation and subsequently disperse into newly available territories. This is a theoretical interpretation of what may have occurred during the 2014 aerial 1080 treatment on Mount Stanley. During the aerial pest control operation, the peak of Mount Stanley was left untreated due to changes in weather conditions prohibiting helicopters to fly over the peak. The peak was subsequently treated with 1080 baits laid by hand.

Data collection compromises

The study faced a trade-off between choosing a study-areas densely populated with SI robins or sparsely populated. It was believed that a high-density population was not facing high predation levels, whereas a marginal population at the edge of the SI robin range was likely experiencing greater predator pressures. If a study area had been chosen within a densely populated area, more nests and pairs could have been monitored, increasing the reliability of the data. However, the study aimed to assess whether aerial 1080 pest control could lift the nest success of SI robins, and it was believed that if any change was to be observed within a short time frame (< 5 years), this would be more visible in a marginal population. The treatment and non-treatment sites in Tennyson Inlet were both chosen at the lowest altitudes of the SI robin distribution (starting at 500 m above sea level).

Climbing trees to mount cameras to monitor nests was also a trade-off between observing higher number of nests and identifying the predatory species responsible for the predation, as tree climbing is labour intensive exercise and costly in training and equipment. Both trade-offs contributed to fewer pairs and nest being monitored: 23 - 62 nests per season (**Table 2.3**). On the positive side, having reliable data on nest outcomes allowed for in-depth analysis of predator-prey relations and population fluctuation changes between predator types in response to beech mast and aerial 1080 treatments.

Table 2.3: Number of nests observed in the treatment and non-treatment areas in Tennyson Inlet from 2012/13-2016/17.

Year	Non-treatment	Treatment	total
2012/13	12	11	23
2013/14	19	17	36
2014/15	26	36	62
2015/16	23	27	50
2016/17	25	21	46
	105	112	217

Marking South Island robins

Sightings were collected from all banded SI robins over the course of the study period during the monitored periods of the breeding season. SI robins were individually banded with a unique metal band and a unique colour band combination (**Plate 2.2**). The monitored period of the breeding season varied between years, starting in August or September and stopping between December and February. Roll-call surveys were carried out in both the treatment and non-treatment area on a monthly basis for the winter of 2016. Only occasional sightings collected while in site for other monitoring purposes were recorded during the winter months for all other years.

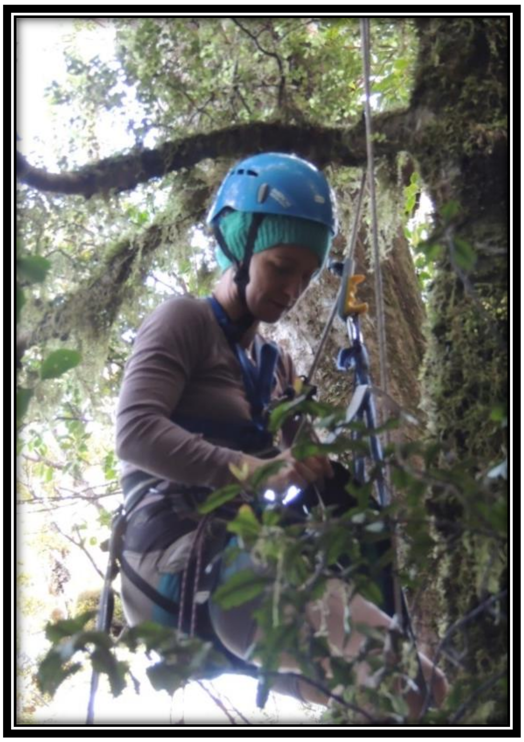
Tree climbing

If safe to do so, all SI robin nests were fitted with a camera, with the exception of late nests in 2015 (December-January). Trees were either accessed from the ground or climbed according to DOC tree climbing protocol (**Plate 2.3**).

If possible, cameras were set up above the nest looking down into the cup (**Plate 2.4**). This was achieved by attaching the camera in the same tree above the nest looking down, beside the nest looking across, or on a nearby branch or tree looking across or down at the nest. The camera was ideally placed 1.5 m from the nest, ranging from 0.3-5 m depending on the surrounding vegetation and access to the nest. Some small branches were removed if they interfered with the camera placement or camera view. This was only carried out if no other options were available to get a clear image of the nest, and kept to a minimum, to decrease disturbance and not change visibility or protection of the nest.

Plate 2.2: Picture shows an individually colour-banded South Island robin in Tennyson Inlet. Individual colour banding was used for adult survival analysis and estimated nest success analysis during breeding seasons from 2012/13-2016/17.
Photo by Mara Bell.

Plate 2.3: Photos show myself climbing trees using rope and abseil gear to access South Island robin nests and set up cameras in the aim of capturing nest predation events.
Left: Photo by Chris Dodd, right: photo by Kathryn Richards.



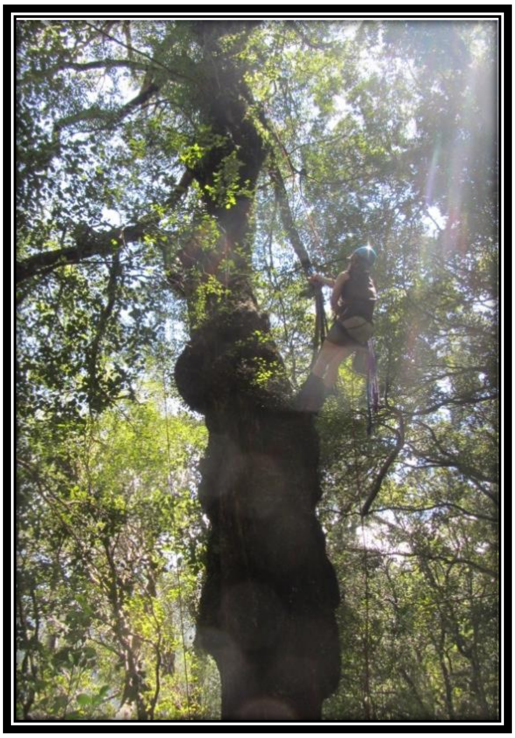


Plate 2.4: Different camera placements were used on South Island robin nests to capture the predator in the act of nest predation depending on nest location and options for camera placements. Nest locations are indicated by the red circles.
Both photos by Mara Bell.

All nests were marked with pink flagging tape on the trunk of the nest tree. A second piece of pink tape with an image of the nest was attached at chest-head height. This made it possible for different team members to readily locate the nest even if it had failed. The assumption was made that our presence, the cameras presence and marking the nest, did not increase predation rates, and was consistent between the treatment and non-treatment areas. The Eastings and Northings were recorded for each nest tree by a hand-held GPS unit (Garmin 58 or 62s) and the elevation recorded with a +/- < 5 m error. A nest description was taken on finding the nest. This description included the tree species, estimated nest and tree height, and the location of the nest within the tree. These descriptions were used to give an image of whether the nest was on the trunk or in the outer branches, if the nest was sheltered in a cavity or exposed in branch forks.

Camera footage

Ltl Acorn 5210A Trail Game Hunting Cameras were used. These were mounted on aluminium brackets. Brackets were either 'Z' or 'L' shaped and came at a range of different lengths so that different camera angles could be mounted. Tape (masking, strapping or insulation) was used to cover different proportions of the light-emitting diode (LED) lights depending on the distance of the camera to the nest, aiming to achieve the best night exposures possible.

Cameras were set on motion triggers with two-three images per trigger. The trigger sensitivity was set according to the distance to the nest and ranged between low and high. SD cards and batteries were optimally replaced every 10 days (range 5 - 14 days). Only rechargeable batteries were used. Powerex batteries (2600 mAh) in conjunction with Maha (MH-C9000) chargers were the preferred option and reliably achieved the longest battery life (ca. 14 days). All images from camera footage were kept from 2014 - 16, but only selected images for the 2012/13 and 2013/14 season were kept.

All camera footage was reviewed using Irfan-View, and all visits by predators and any changes of nest state, such as laying, hatching and fledging were recorded. The exact time of the predation event was recorded, and all predators identified if possible. If the camera was triggered but the images were of poor quality, then the nest failure was attributed to an ‘unknown’ predator, meaning that the camera was triggered by a visitor and the nest subsequently failed, but the images were too blurred to identify the visitor/predator. If a nest failed in the absence of a visitor triggering the camera, the nest failure was not attributed to predation. No further data were recorded after the predation/failure event occurred.

The first active nest day was considered to be two days before incubation was observed to commence or at least one egg seen in the nest cup (the laying period). The total nest days were based off our own observations, and consist of 2 laying days with incubation starting on the day of the last egg being laid (day three). If laying was not observed laying dates were estimated from the hatch date. The incubation period consisted of 18 days with hatching occurring on the 21st nest day (19^{th} day of incubation). This held true for all nests were laying was observed. The chick stage was more variable in length. The most common observed nests at chick stage consisted of 22 days including the day of hatching and fledging. This gave us a commonly observed active nest period from first egg laid to fledging of 42 days in which a predator could cause a nest to fail.

Data analysis

Each chapter will deal with specific data analysis methods as applicable.

- **Chapter 3** this chapter provides background information on rat, stoat and possum numbers over the course of the study period and in response to two aerial 1080 applications.
- **Chapter 4** focuses on a range of exploratory questions from additional data collected from nest cameras during SI robin nest success monitoring. Providing an insight into SI robin and predator behaviours during the breeding season.
- **Chapter 5** is a data analysis chapter focusing on generalised linear models (GLMs) used to analyse nest success (ENS) for SI robins over the course of the study period (2012/13-2016/17) and in response to two aerial 1080 treatments.
- **Chapter 6** is a data analysis chapter focusing on SI robin adult survival during the aerial 1080 operations using mark-recapture analyses in program MARK.

Chapter three

Presentation of data on predator tracking rates through a beech-mast and two aerial 1080 pest control operations

3.1 Abstract

This project aimed to monitor the population response of brushtail possums (*Trichosurus vulpecula*), rats (*Rattus rattus*) and stoats (*Mustela erminea)* over the course of seven years. This time period included two aerial 1080 operations (2013 and 2014) and a beech (*Nothofagus*) mast (2014). By monitoring key predator tracking rates, a measure of pest control success and interactions between predators can be observed. Similarly, the population interactions between the key predators in response to beech masts can be observed.

Both aerial 1080 operations significantly reduced the three key predator species targeted. Stoats showed small fluctuations over the years, had a negative response to aerial 1080 treatment and a positive response to high rodent numbers. Possums showed a strong negative response to the aerial 1080 treatment and a slow annual growth rate thereafter. They did not show any association with the beech mast event of 2014. A positive response in rodent tracking rates to the beech mast event of 2014 in both the treatment and non-treatment area was observed. Rodents negatively responded to the aerial 1080 operation conducted in 2013 and 2014 in the treatment area. With pest control efforts suppressing rodents for the duration of the nesting season in which the application occurred. However, the rodent suppression did not extend beyond those periods.

The key predators were more effectively suppressed in the 2013 aerial 1080 operation, i.e. in the absence of a beech mast. However, the 2014 beech mast drastically lifted rodent and stoat numbers irrespective of 1080 treatment the previous year (2013). The aerial 1080 operation of 1 kg/ha sowing rate conducted in 2014 in response to a beech mast, did not suppress the rat tracking rate below the 5% target. The population growth rate after the 1080 operation in 2014 was also faster, steeper and longer lasting for rodents than expected.

We suggest that sowing rates may have been too low and immigration (due to the small treatment area) too high, for longer lasting predator suppression. Additionally, the mixed podocarp forest system along the coast likely provided a varied and productive food source fuelling rapid rodent population growths after 1080 treatment.

3.2 Introduction

Please refer to 'General Introduction' (**Chapter 1**) for information on the study area, conservation and 1080 pest control in New Zealand.

The data discussed here are published in other sources (Tinnemans et al., 2018, Elliott and Kemp, 2016). However, these data were used to model SI robin survival rates and nest success rates and are therefore essential data underlying subsequent chapters in this thesis.

Beech mast events

Masts of native beech trees (*Nothofagus* spp) are a common event in New Zealand, and have a strong influence on food webs and predator-prey interactions. About 60% of the remaining New Zealand forests have one or more of the four beech species (Wiser et al., 2011) and about 30% of New Zealand's forests are 'pure' beech forests, where beech trees make up the canopy (**Fig. 3.1**). Overall, New Zealand has numerous native masting plants, from alpine tussocks to lowland podocarps such as rimu (*Dacrydium cupressinum*).

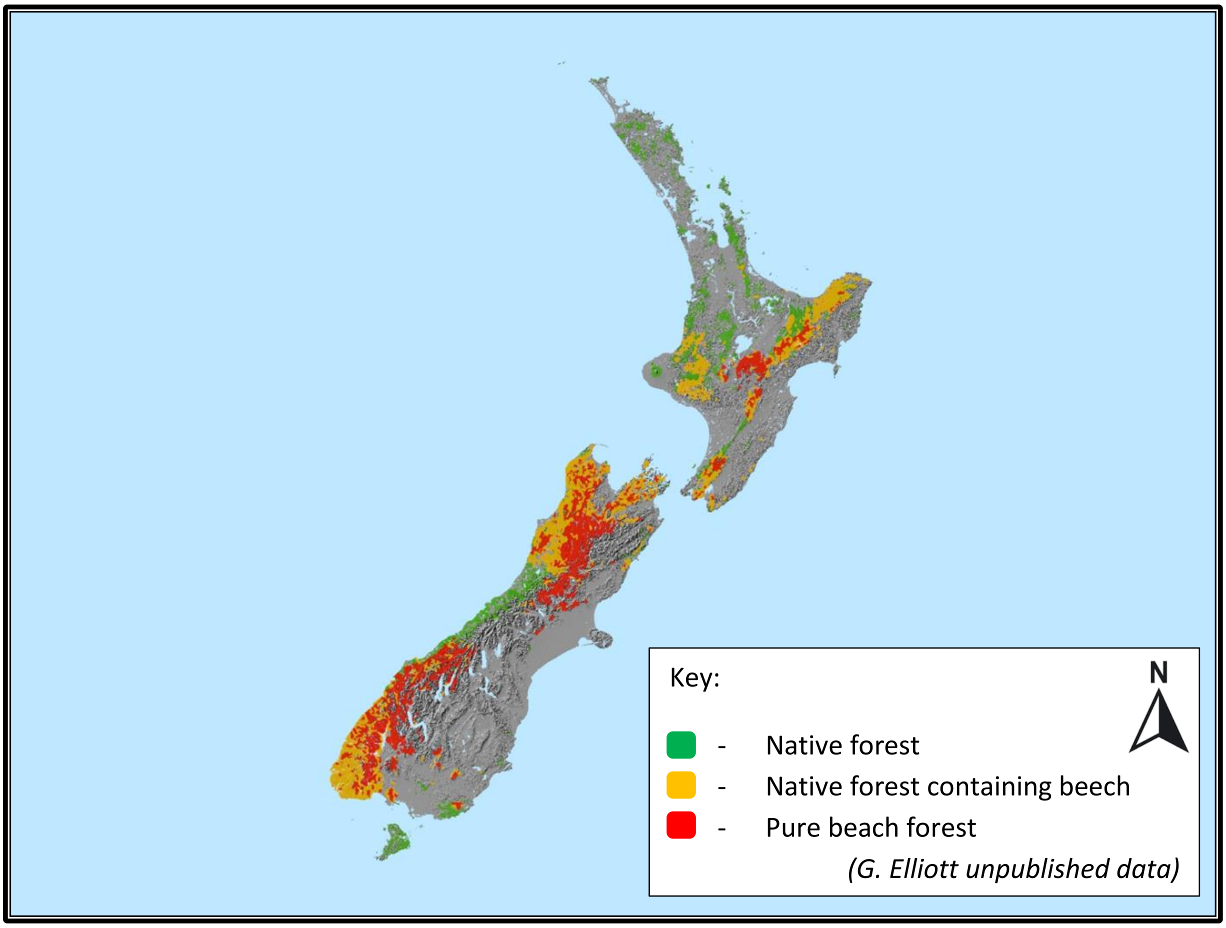


Fig. 3.1: Map of remaining native New Zealand forests showing the importance and distribution of native beech (*G. Elliott, unpublished data*).

Masting refers to a mass seed/fruit production at irregular intervals due to climatic conditions. Since 2013 beech masts can be reliably predicted using the delta T (ΔT) model developed by Canterbury University in conjunction with National Institute of Water and Atmospheric research (NIWA) (Pech et al., 2015). New Zealand beech flower in 'year three' when the mean summer temperature in 'year one' was lower than the mean summer temperature in 'year two'. In the mast year 'year three', huge blooms occur around November, with seed growth and development occurring over late summer, and seeds ripening and falling to the ground from March to August depending on altitude and species.

In 'year three', weather events such as storms, can negatively affect seed setting and ripening. Once seed has fallen it is quantified by the number of viable seeds produced per m^2 and reported as low, medium and high. The beech seeding event is only called a mast in a high seeding year, where >2000 seeds/m^2 are produced. A year of low seed fall has fewer than < 500 seeds/m^2 and a 'moderate year' produces 500 - 2000 seeds/m^2 (Elliott and Kemp, 2016). A beech mast lifts the carrying capacity of many native populations in the ecosystem, including small insects and birds. Today, introduced rodents also benefit from the beech seed which is a valuable food source allowing them to increased reproductive output in 'year four'. A system with high rodent numbers supports elevated reproductive rates for stoats, lifting the stoat population in 'year five'.

Tree species like rimu also respond to the same climatic conditions for their mass fruiting events, though the maturation of the fruit takes two years, fruiting in 'year four'. This creates a secondary mast event the year after the beech mast in forests containing large numbers of beech and rimu. This can elevate the carrying capacity for seed eaters for two successive years (year four and five). These plague events where a forest supports elevated rodent numbers for one or more years is very damaging to native and endemic species preyed on by introduced mammals, and can cause local extinctions.

In 2014, 3.8 million ha (90%) of New Zealand beech forest was predicted to undergo a mass fruiting event with expected pest control costs of $68 million dollars for the affected areas where rat and subsequently stoats numbers are expected to soar (Pech et al., 2015). To target the most 'at risk' areas of the conservation estate, $12 million dollars were quickly made available to the Department of Conservation (DOC) under the 'Battle for our Birds' campaign in 2014 (Elliott and Kemp, 2016). The Tennyson Inlet project benefitted from this funding allocation and received aerial 1080 pest control treatment at a 1 kg/ha sowing rate on 23 November 2014 (Elliott and Kemp, 2016, Pech et al., 2015).

Some areas of New Zealand such as South Westland have in the past experienced a beech mast about every three years according to the ΔT model, while other areas of New Zealand historically experience beech masts events every 5 - 6 years (Pech et al., 2015). When several regions all undergo a beech mast at the same time the event is called a mega-mast or a mass beech mast. Historically, it is uncommon for mega-masts to occur; over the past 40 years only 11 mega-masts have been recorded with smaller localised masts occurring more frequently (Pech et al., 2015). The ΔT model has also been applied to predict beech mast events into the future and predicts that historical irregular beech masts will continue at a similar rate as recorded historically (Pech et al., 2015).

Introduced mammalian predators in New Zealand

Introduced mammalian predators have a significant impact on native fauna and flora, through direct and indirect predation and competition (Innes et al., 2010, Atkinson, 2001, King, 1984). The first rat (*Rattus exulans*) arrived in New Zealand with early Polynesian settlers around 1200 - 1400 AD. Since then, three more rodent species have colonised New Zealand. Mice (*Mus musculus)*, Norway rats (*Rattus norvedicus)* and ship rats (*Rattus rattus)* have all been introduced to New Zealand with early European settlers (King, 1984).

In comparison to these semi-accidental rodent introductions, all three mustelid species -- stoats (*Mustela erminea*), ferrets (*Mustela putorius furo*) and weasels (*Mustelinae sp.)* -- were purposely introduced to control rodents and rabbits (King, 1984). Even at the time of these introductions there was a lot of opposition to their release, with opponents fearing for the long-term survival and population health of native birds.

Possums (*Trichosurus vulpecula*) were deliberately introduced to New Zealand for commercial gains from the international fur trade. This introduction was sanctioned by the New Zealand Government, with 36 batches of possums released at 450 sites across New Zealand between 1922 - 1930 (Hutching, 2008a, Hughes, 1994). At the time it was believed that possums were vegetarian. Although predominantly herbivores, they also consume insects, eggs and nestlings (Innes, 2000, Hughes, 1994) along with 21,000 tonnes of vegetation per day nationally. Possums alone cause $35 million worth of damage to pastures and depleting forest species such as mistletoe in New Zealand (Hutching, 2008b, Nugent, 2000). Heavy browsing weakens forests, making them more susceptible to pathogens, windfall and insects, and can also alter forest composition (Pekelharing and Batcheler, 1990, Payton, 1988, Meads, 1976).

As mammalian predators colonise new areas with high food supplies they can reach unsustainably high population densities, causing catastrophic and sometimes irreversible damage on flora and fauna as seen with canopy forest collapses after colonisation by possums (Innes, 2000). In the case of possums, the average body weight in New Zealand is heavier than for their founder populations in Tasmania, Australia. Interesting genetic research has recently revealed that New Zealand stoats have a more diverse gene pool than in the founder populations of the United Kingdom (Veale et al., 2015). This shows that not only have these predators managed to establish in New Zealand, but their populations have vigorous and healthy gene pools and high productivity

It is well documented that these three key mammalian predators reduce both adult survival and nest success of a wide range of bird species in New Zealand (Innes et al., 2015, Innes et al., 2010, Atkinson, 2001, King, 1984). This predation impact can lead to local extinctions of already small and isolated remnant populations during plague events such as after a beech mast (Pech et al., 2015). Forest types such as lowland mixed prodocarp forests have consistently abundant seeds and fruits for rodents, resulting in very high rodent carrying capacities even in the absence of beech masts. Today, rats perceived to be a key predator responsible for a large proportion of nest failures among New Zealand passerines (Graham and Veitch, 2002, Brown et al., 1998). However, this is not the case for all New Zealand birds. Kaka and ground-nesting birds such as those in braided rivers are not affected by rats (Sanders and Maloney, 2002, Moorhouse et al., 2003), while weka actually benefit from the presence of rodents (Tinnemans et al., 2018), as weka prey on rats and mice.

New Zealand had its own set of apex predators in its pre-human era, with giant birds roaming the forests, wetlands, grasslands and alpine zones attacking smaller birds and reptiles by sight (Hackwell and Bertram, 1999). Giant weka roamed the undergrowth, and nightjars and owls hunted during the night. The Haast eagle (*Harpagornis moorei*) could take a 400 kg moa and had talons bigger than the largest extant predator the Bengal tiger (*Panthera tigris tigris*) (Hackwell and Bertram, 1999). It is not the presence of predators that makes New Zealand endemics so vulnerable; it is the type of predators that has caused the mass declines. Mammals hunt by smell and sound, not vision (Hackwell and Bertram, 1999) and there is no coevolution resulting in a lack of threat recognition by New Zealand natives. For these simple reasons, many New Zealand native species are ill prepared, as their anti-predator responses are generally to sit still and camouflage with their surroundings, making them a very easy target for agile climbers with an excellent sense of smell (mammals).

Monitoring techniques

Monitoring changes in population size and distribution over large areas is only cost-effective with crude monitoring techniques. Techniques such as the use of transmitters, transponders, pit and satellite tags can provide mark-recapture data that give good information on survival, distributions, and density, but are labour-intensive and expensive.

To assess changes in population sizes and distributions a commonly used technique is using trapping/encounter rates. These come in the form of live trapping, camera trapping and capturing the tracking rates of the target species or faecal encounter rates, call rates or visual encounter rates. Such techniques often just give indices of density, and do not give information on survival or individual data such as age and health biometrics. Tracking rates where no individual is identified is the crudest of all measurements, as there is no way of differentiating if one animal was trapped multiple times or multiple animals were trapped on a single occasion.

In New Zealand, tracking rates using foot prints has become the standard technique to monitor rodents and mustelids (King and Edgar, 1977), and later hedgehogs and a range of native species such as weta. Within a framework of rules on where to set trap lines, changes in predator density/behaviour can be compared over time or between areas. Attempts have been made to calibrate tracking rates with actual population densities, territory sizes and other demographic features of interest (Brown et al., 1996b), though no generally accepted results are currently available.

As a measure of before and after pest control treatment applications for mice, rats and mustelids, tracking rates provide a great measure of population increase or decrease, even if direct densities cannot be estimated. Tracking rates are also of some value as a comparison between different sites, though this is less reliable as the populations and ecosystems surveyed are different.

Tracking and trapping rates often vary at different times of the year, due to differences in behaviour of target species. For possums, juvenile dispersal is a normal part of life and occurs on an annual basis around February each year (Efford, 1998). Juvenile possum dispersal varies between the sexes with juvenile females dispersing on average 5.5 km compared to 3.7 km for males (Efford, 1998), whereas adult home ranges rarely change over a life time with only 10 % of possums moving < 500 m to reach another forest patch (Cowan and Rhodes, 1993). Trap shyness or avoidance is commonly recorded for rats and mustelids. Brown et al. (1996b) found that adult ship rats were more trappable than juveniles, and mouse tracking rates increased with the removal of rats over a five-day period. Such variables create error in the tracking rates derived from tracking tunnels.

Even though tracking rates are a crude measure, ship rat tracking tunnel rates were first found to be a useful nest success indicators for North Island kokako with the publication of (Innes et al., 1999), and the standard targets of a <1 – 5 % tracking rate was adopted across the country to measure the success of pest control efforts. Tracking rates of rats have since also been found to be good indicators for nest success of NI robins (Armstrong et al., 2006b).

For possums, which are less attracted to tunnels and too big for rodent tunnels, other monitoring techniques are used. Live or kill traps have been widely used as well as wax tags *("Possum population monitoring using waxtag method*", 2010). Wax tags are used to trap chew marks of a range of species including possums, and like tracking tunnels indicate changes in density and distribution but cannot be used to estimate density and cannot identify individuals.

Aim

To monitor change in introduced predator numbers and distribution over the course of the study period 2010-2016 (possums, rats and stoats).

Assumptions

The assumption is that the treatment area (Mount Stanley) and the non-treatment areas (Editor Hill - Lookout Point) have the same carrying capacities, immigration, emigration and tracking potential of rodents, mustelids and possums.

3.3 Methods

The ‘General Methods’ section of this thesis (**Chapter 2**) contains a detailed description of the study site and the aerial 1080 treatments that apply to this chapter. The following methods solely concern this chapter.

Tracking tunnels

The tracking tunnels used in Tennyson Inlet were installed specifically for this research project. Tunnels were set in lines of 10, with five lines making up a loop that could be completed by one person in a single day. Lines were marked though not cut. Two loops were placed into the treatment area and two into the non-treatment area, covering a range of altitudes from sea level right to 1000 m in both areas (**Fig. 3.2**). All four loops, if possible, were run simultaneously and repeated in February, May, August and November. During pest control efforts an additional set of data was collected to capture a before and after treatment effect. The before treatment survey was aimed to be close as possible to the pest control event and the treatment effect survey was set a month after the pest control event.

For rodent tracking, pre-inked tracker cards were baited with peanut butter at both ends of the tunnel when set, and left in the forest overnight before collection. On collection the tracker cards were replaced with a new set of individually marked and dated cards and baited for stoats with a single piece of fresh rabbit meat placed in the middle of the ink pad. These were left out for three clear nights if possible before recollection.

The stoat monitoring protocol was updated half-way through this study, as stoat tracking rates were too low for analysis purposes in low tracking years. The new method was introduced in 2014, and for one year both were run in Tennyson Inlet, before the old method was made redundant. The new method used salted rabbit meat caged up in wire mesh or a stainless-steel tea balls to stop the bait from being removed by rodents and mustelids alike. The salt increased the life of the bait, especially in hot weather. Under this method the tunnel could be set for two weeks and gathered more stoat foot prints than with the old method, allowing change to be observed when stoat numbers were low.

Wax tags

For possum monitoring, 10 wax tags were used per line, 10 m apart. Wax tags were used in 2013 before and after the 1080 operation, in 2014 after the 1080 operation and in 2017. Each wax-tag operation used newly randomly generated lines. Due to possums having slow population growth and relatively stable densities, only four surveys were conducted in the treatment area and three in the non-treatment area over the course of the study period 2013 - 2017. The number of survey lines was adjusted between surveys due to resource constraints (**Table 3.1**). Some lines had to be moved *ad hoc* while in the field due to terrain or time constraints. This only affected up to two lines per survey and standard procedures were followed *("Possum population monitoring using waxtag method*", 2010. *"Possum population monitoring using waxtag method*", 2008).

All wax tags were removed exactly 7 days after being set. The Residual Trap Index (RTI) is used as a standard measure of possum density. The RTI is a similar measure to the tracking rate collected for rodents and mustelids from tracking tunnels.

As an example, the 2017 lines were grouped into five rounds in the treatment area and three rounds in the non-treatment area (**Fig. 3.3**), which came to 16 person days. Tawa, Matai, Stanley and the Bivvy line were run together on day one, Ngawhakawhiti and

Lookout Point lines were set on day two, and Editor Hill and Tuna Bay lines were run on day three. Lines were set from 24-26/01/2017 and pulled out 31/01-02/02/2017. A density measure for possums was then calculated using the Bite Mark Index (BMI) and converting it to RTI. The standard conversion rate is BMI%*0.24704=RTI (Agencies, 2010).

For rodent, mustelid and possum surveys and extended treatment and non-treatment area was used, making up four survey areas in total (**figure 3.2**).

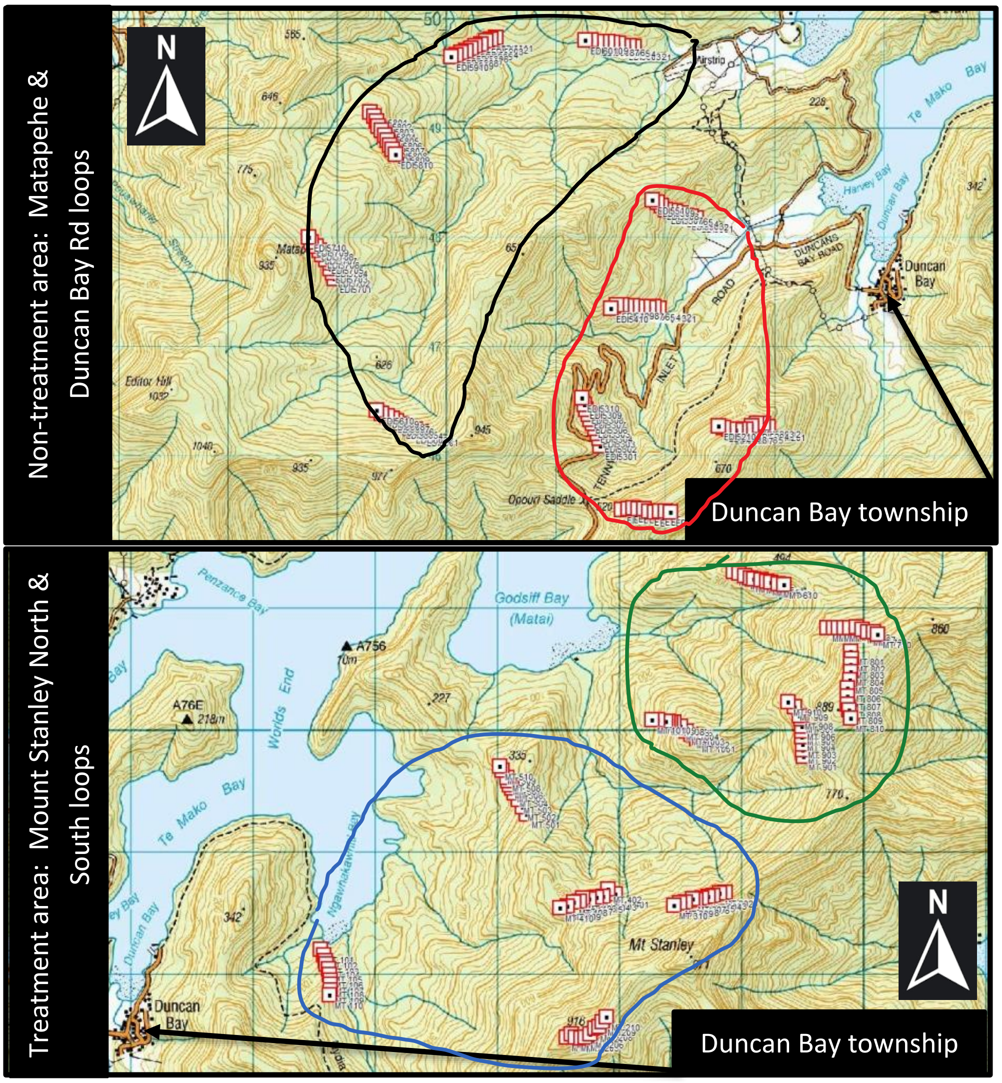


Fig. 3.2: Placement of tracking tunnel lines in the treatment and non-treatment area of Tennyson Inlet. Matapehe (shown in black) and Duncan Bay Road (shown in red) loops are in the non-treatment area and the Mount Stanley North (shown in green) and Mount Stanley South (shown in blue) loops are in the treatment areas. With five lines of 10 tunnels in each loop, giving 10 lines per areas (treatment and non-treatment) adding to a total of 200 tracking tunnels for Tennyson Inlet.

Table 3.1: Brush-tail possum surveys conducted between 2013 and 2017 in Tennyson Inlet. Each line consisted of 10 waz chew tags spaced 10m appart. The starting points for all lines were randomly selected and never repeated. The lines ran North from the starting point were possible, else South, lines were only moved if the circuit could not be completed within a working day.

Survey	Area	Lines	Tags
Sep-13	non-treatment	20	400
Sep-13	treatment	20	400
Jan-13	non-treatment	0	0
Jan-13	treatment	20	400
Jan-14	non-treatment	10	200
Jan-14	treatment	20	400
Jan-17	non-treatment	10	200
Jan-17	treatment	20	400

Timing of rodent, mustelid and possum surveys

The exact timing of surveys was variable among years and before and after pest control treatment efforts. When possible, surveys were run simultaneously between the treatment and non-treatment areas, otherwise back-to-back depending on staff availability and weather conditions. These minor fluctuations are not expected to have an effect on results (Schadewinkel et al., 2014).

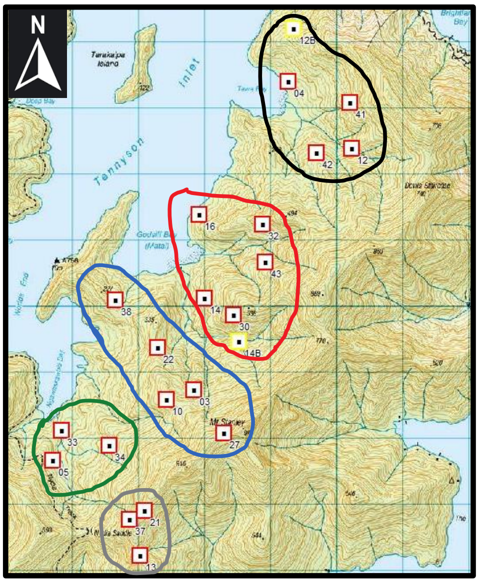


Fig. 3.3: The 20 randomly generated starting points used (red points) for possum wax tag monitoring in 2017 treatment area. Only locations that fitted into four person-day circuits were kept, while other starting locations were discarded (such as yellow points). If possible, the starting points were used as the most southernmost wax tag site with the remainder of the line bearing north. If this was not possible the line was run south from the starting location. This map shows the starting locations for five people days; Matai (shown in black), Tawa (shown in red), Stanley (shown in blue), Ngawhakawhiti (shown in green) and Bivvy (shown in grey). The non-treatment starting points are not shown on this map, this includes the Lookout Point, Editor Hill and Tuna Bay lines.

3.4 Results

Rat tracking rates

Rat tracking rates showed similar patterns in the two treatment areas and the two non-treatment areas. One of the four loops (non-treatment Duncan Bay Road) did not experience the extremely high tracking rates the other three sites experienced following the beech-mast event of 2014, but tracking rates pleaked around the 60% before dropping down to 50% around December when the other three areas peaked between 75 and 95% TR (**Fig. 3.4**).

Of interest is the lack of difference between the treatment and the non-treatment area during the time of the 1080 pest control operation in 2013 (non-beech-mast year). Both the treatment and the non-treatment lines show decreases in rat tracking rates around the time of the 1080 operation.

This pattern did not occur during the beech-mast year of 2014. In 2014 tracking rates drastically dropped in the treatment area whereas in the non-treatment area there was a prolonged period of high rat tracking, right throughout the breeding season. Followed by a reduction during the winter months of 2015. Subsquently the rat tracking rates dropped in the non-treatment area, and were near zero by the end of the SI robin breeding season in February 2016. The observed tracking rate decrease in the treatment area after the 1080 treatments were short lived, and the rat tracking rates never dipped below 5% target. This resulted in a relatively stable high rat tracking rate (45% +/- 6.41) over the 2015/16 and 2016/17 SI robin breeding seasons and the two winters following the 1080 operation in 2014 in the treatment area.

A third point of interest is the difference between the treatment and non-treatment areas from 2010 right through to the 2014 pest control effort. Interestingly, the non-treatment area continuously tracked higher. And lastly there in the non-treatment area rat trackig rates kept dropping naturally for about a year after the 2014 - 15 summer, while in the treatment area rat tracking rates started to rise again imediately after the pest control effort of November 2014.

When the two loops are combined for the treatment area and two non-treatment area (**Fig. 3.5**), the delayed decline in rat tracking rates in the non-treatment is clearly visible. However, the suppression of rats was short lived, which is inconsistent with prediction that rat populations would stay low for at least one to two years following the 1080 operation (**Fig. 3.5**).

When the tracking tunnel lines are split altitudionally, a clear difference between the higher and lower altitudes is observed, with rat tracking rates being consistently higher in at lower altitude (< 500 m) in both the treatment and the non-treatment areas (**Figs. 3.6** & **3.7**).

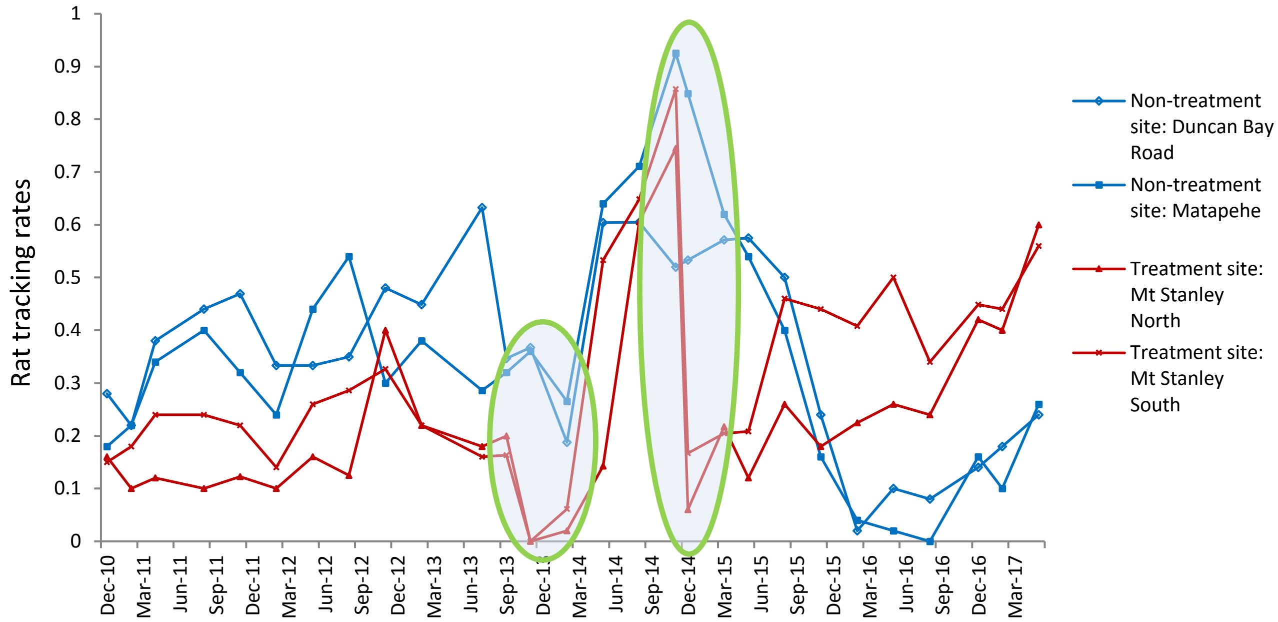


Fig. 3.4: Rat tracking rates forTennyson Inlet study area from 2010-2017. The two tracking tunnel loops are shown in blue for the treatment area and in red for the non-treatment area. Each loop had five tracking tunnel lines of 10 tunnels, these were spread over a range of altitudes. The 2013 aerial 1080 operation occurred on the 2 November, with a pre-feed of non-toxic pellets on the 20 October. The 2014 1080 operation occurred on the 23 November, with a pre-feed of non-toxic pellets on the 2 November. The two aerial 1080 operations and subesecquent impact on rat tracking rates are highlighted by the two green ovals.

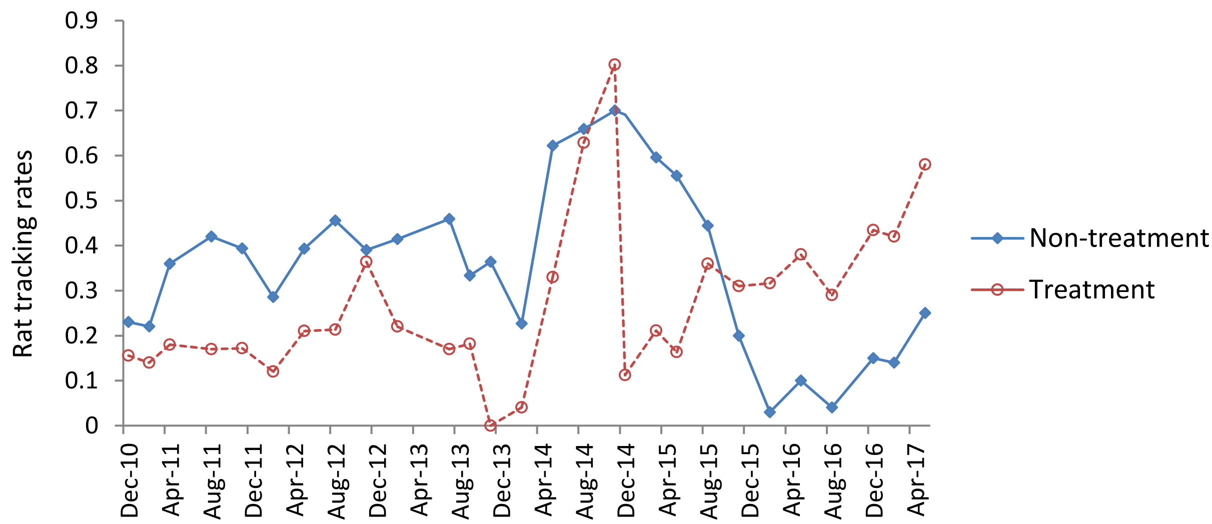


Fig. 3.5: Average rat tracking rates for Tennyson Inlet study area collected from 10 tracking tunnel lines of 10 tunnels each in the non-treatment area (blue) and 10 lines in the treatment area (red). Two aerial 1080 opperations were conducted during this time period the first on the 2 November 2013, with a pre-feed of non-toxic pellets on the 20 October. Followed by a second on the 23 November 2014, with a pre-feed of non-toxic pellets on the 2 November.

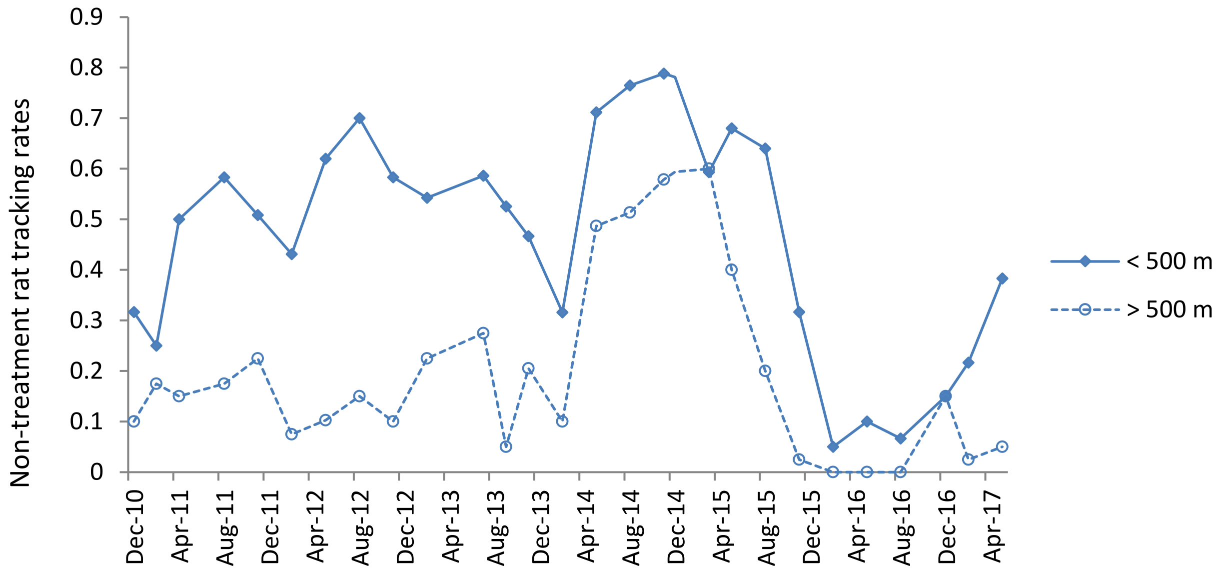


Fig. 3.6: Rat tracking rates for the non-treatment area of the tennyson Inlet study area are presented here split into lower (< 500 m, 5 lines) and higer altitudes (> 500 m, 5 lines) from 2010-2017. South Island robin are only present in altitudes above 500 m in Tennyson Inlet, therefore the higher altitudional tracking rates more relevant when assessing the impact of rats on nest success. During this period, no pest control was conducted in the non-treatment area.

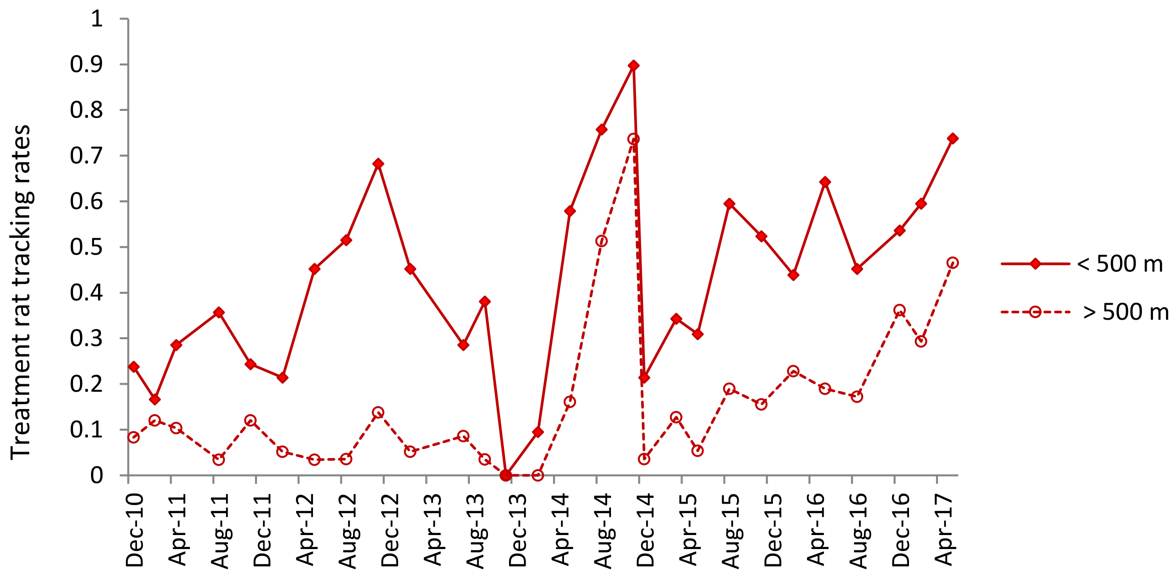


Fig. 3.7: Tennyson Inlet rat tracking rates in the treatment area are split between lower (< 500 m, 5 lines) and higer altitudes (> 500 m, 5 lines) from 2010-2017. The higher altitudional tracking rates more relevant when assessing the impact of rats on nest success of South Island robins as they are only present in Tennyson Inlet in altitudes above 500 m. During this period two aerial 1080 opperations were conducted the first on the 2 November 2013, with a pre-feed of non-toxic pellets on the 20 October. Followed by a second on the 23 November 2014 in response to a beech mast, with a pre-feed of non-toxic pellets on the 2 November.

Stoat tracking rates

Stoat tracking rates varied over time. Between 2013 and 2015 stoats appear to have gone through a population boom, even without a conversion rate between the old and new method this growth is visible in the October 2014 survey which used both methods and obtaned very sililar tracking rates. Stoats appear unable to recover to former population density after the first 1080 application (2013) in the treatment area (**Fig. 3.8**).

When split altitudionally, stoat tacking rates were higher above 500 m than below 500 m in both the treatment and the non-treatment areas (**Figs. 3.9 - 3.10**). Both the lower and higher altidudinal areas showed increased tracking rates in 2015, the year after the beech mast when rat numbers were elevated. In November 2015 there was no stoat in the treatment area above 500 m altitude. This seems unusual though may mean that stoats did not breed in the higher altitudes and only dispursed into the higher altidudional areas after the breeding season as seem by the February survey. The same pattern is repeated in 2016 with an absence of stoat tracking above 500 m altitude in November followed by a peak in February (**Fig. 3.10**).

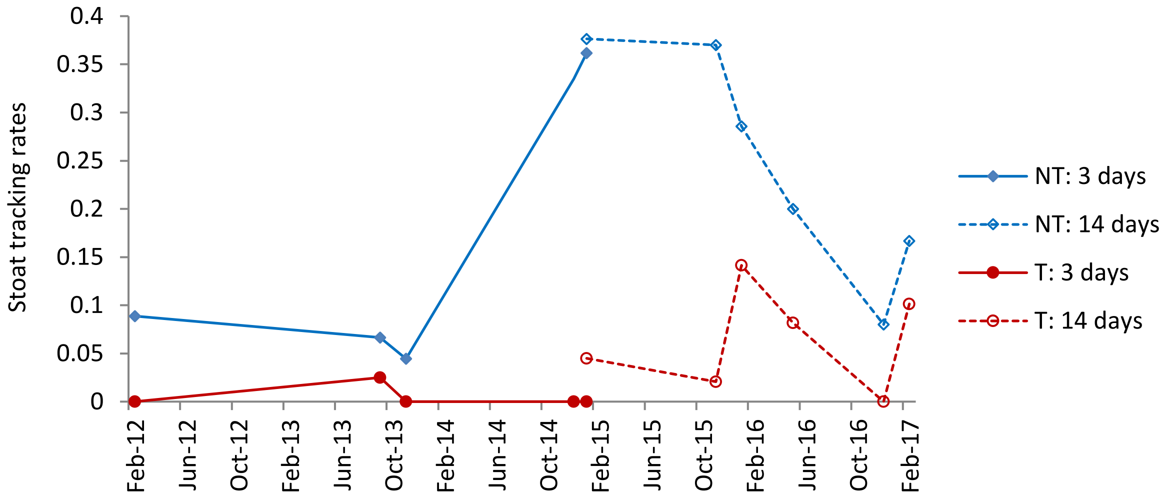


Fig. 3.8: Stoat tracking rates from 2012/13-2016/17. The 10 lines for the non-treatment area (NT shown in blue) are combined here and the 10 lines for the treatment areas (T showin in red) are also combined. No pest control efforts were conducted during this time period in the non-treatment area. The stoat tracking procedure was changed during the study period from a three day to a 14 day tracking period to increase, this is indicated with solid lines for the old method of three days and the new method of 14 days.

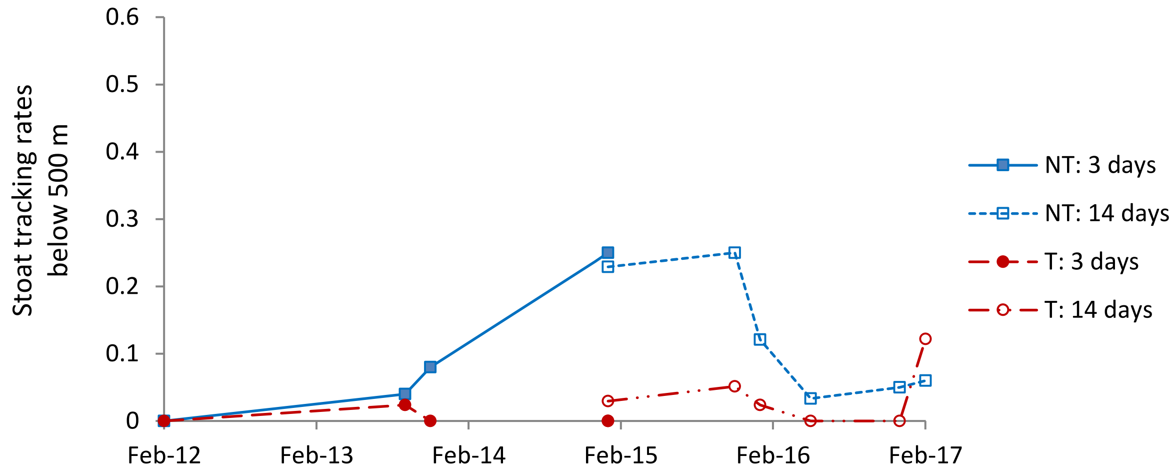


Fig. 3.9: Stoat tracking rates for lower altitude areas (< 500 m) over the study period of 2012/13-2016/17. The 10 lower altitude lines for the non-treatment lines, Matapehe and Duncan Bay Road are combined and shown in blue, and the 10 lower altitude lines for the treatment areas, Mount Stanley North and South combined are shown in red. The stoat tracking procedure was changed during the study period from a three day to a 14 day tracking period to increase, this is indicated with solid lines for the old method of three days and the new method of 14 days.

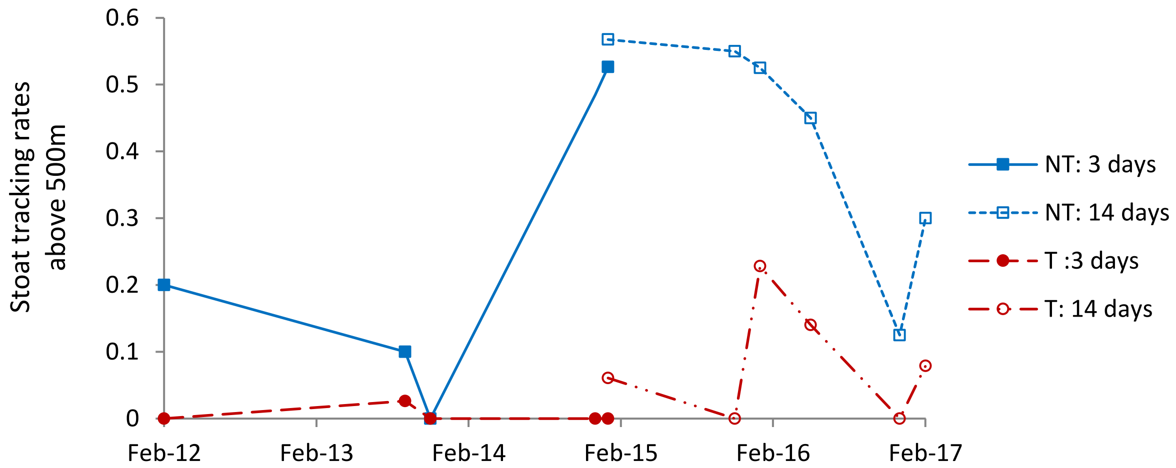


Fig. 3.10: Stoat tracking rates for higher altitude areas (> 500 m) over the study period of 2012/13-2016/17. The 10 lines for the non-treatment and the 10 lines for the treatment area are combined. Notation and colours as for **Fig. 3.9**. Though the higher altitudional tracking rates may be more relevant for assessing the impact of rats on nest success of South Island robins this may not apply to stoats, as they have bigger territories that may cover a large altitudional range. During this period two aerial 1080 opperations were conducted the first on the 2 November 2013, with a pre-feed of non-toxic pellets on the 20 October. Followed by a second on the 23 November 2014 in response to a beech mast, with a pre-feed of non-toxic pellets on the 2 November.

Possum wax tags

As expected possum numbers did not greatly fluctuate over the survey period of 3½ years (2013 - 2017). A slow increase in Residual Trap Index (0.9/yr) was observed in the non-treatment area over the survey period of 42 months between September 2013 and January 2017. A steeper RTI increase was observed in the treatment area after the 1080 treatment of (2.4/yr over a 25 month period from 2014 to 2017) than in the non-treatment area (1.5/yr over the same period). The RTI in the treatment area by the end of the study, three years after treatment, was still well below pre-treatment RTI for possums.

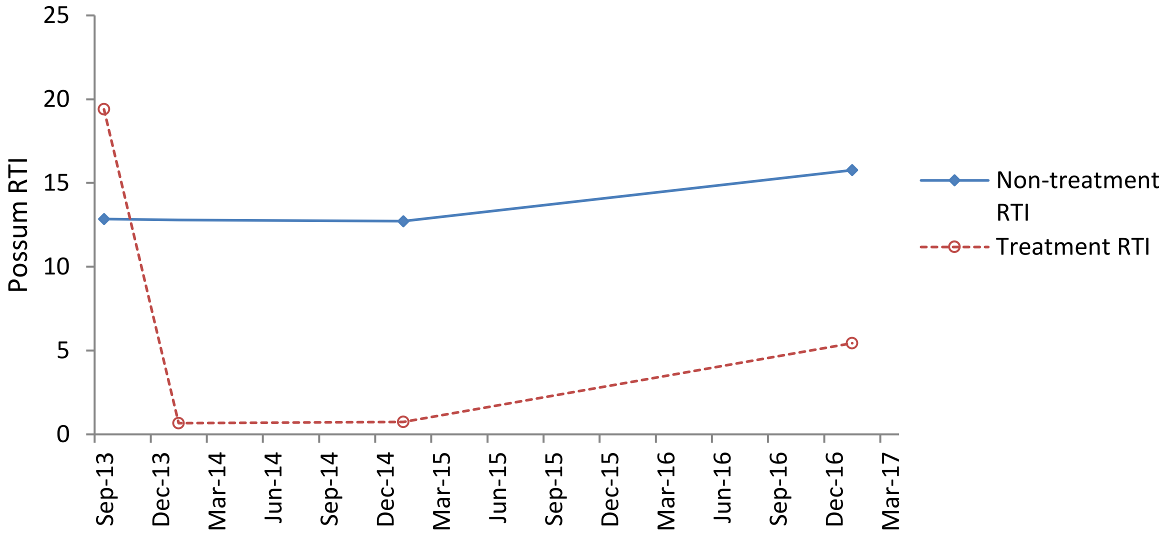


Fig. 3.11: Brush-tail possum Residual Trap Index (RTI) calculated from the Bite Mark Index (BMI) conducted on four occasions in the treatment area and on three occasions in the non-treatment area. The first on the 2 November 2013, with a pre-feed of non-toxic pellets on the 20 October. Followed by a second on the 23 November 2014 in response to a beech mast, with a pre-feed of non-toxic pellets on the 2 November.

3.5 Discussion

The use of wax tags and tracker cards indicated area-specific population fluctuations for the three key predators over the study periods. There were clear differences between years and areas with the beech mast of 2014 lifting rat numbers that year. This was followed by elevated stoat numbers in 2015.

Both aerial sodium fluoroacetate (1080) operations removed predators from the treatment area. The 2013 pest control operation had minimal long-term effect likely due to the beech mast occurring the following year, but was very effective at decreasing all three predators. Possums density never completely recovered in the treatment area after this initial pest control effort in 2013. In contrast, the rat population experienced a population explosion in the treatment area similar to that of the non-treatment area. This was despite the 1080 operation in 2013 having depleted the rat population to low density as indicated by the tracking rate being below 5%.

Stoats appear to have been negatively affected by the 1080 operations. Stoats in higher altitudinal areas appeared to be slow to recover, their density did not appear to have fully recovered by the end of the study period, three years after the 1080 operation. The new lure – salted rabbit meat, indicated seasonal population fluctuations with short lived peaks in February 2015, 2016 and 2017. This implies that the resident stoat population remains low and may actually still be decreasing at the higher altitudes even with rat numbers being reasonably high (~20% tracking rate) from 2015 - 2017.

The two 1080 operations seem to have had different impacts on the rat populations. Rats were not suppressed below the desired 5% tracking rate in 2014 during the beech mast year. The rat population recovered very quickly after both 1080 operations (2013 and 2014). The return of rats after the 2013 pest control effort is most likely related to the beech mast of 2014, though the quick recovery after the 2014 pest control effort is lees easily explained.

There is a possibility that the sowing rate of 1 kg/ha was too low, or that the remaining rats still had ample food allowing them to recover. Tennyson Inlet and especially the Mount Stanley peninsula are not pure beech forest and contain species like rimu (*Dacrydium cupressinum*) at lower altitudes. Rimu mast the year after beech due to a two-year seed maturation cycle, so may have provided additional food that allowed the rat population to recover quickly. The treatment area is also very small 4300 ha (Elliott and Kemp, 2016), and it is likely that immigration played a major role in the recolonization of the treatment area for all three mammalian predator species. Additionally, the Mount Stanley treatment block contains large buffer areas in which no pest control was conducted. This included coastal areas, private land, huts and a public walking track, from which recolonisation by rats was very likely. Additionally, there is evidence that after a 1080 pest control operation, competition for food resources is reduced. Rats and possums appear to compete, as they have overlapping diets with a high proportion of insects, seeds and fruit. Possum removal therefore may reduce competition for food, allowing for a faster rat-recovery rate. Juvenile rats were found to be most reliant on insects while adult rats on seeds (Sweetapple and Nugent, 2007). After a pest control effort, rat recovery rates were correlated to the abundance of fruit and seed which made up 74% of the rats' diet (Sweetapple and Nugent, 2007).

The key difference between the 2013 and 2014 operation was the failure of the 2014 operation to suppress rats to below the desired 5% tracking rate while the 2013 pest control effort succeeded. This implies that the rat density may have been too high for a 1 kg/ha sowing rate to adequately suppress the rat population. Interestingly, other areas experienced the same failure to suppress rats during the 2014 mast year (Elliott and Kemp, 2016). Further research is underway and treatment options are being explored in the hope of finding a pest control regime that supresses' rats in a mast year and keeps number low for several subsequent years in coastal mixed forest sites like Tennyson Inlet.

Chapter Four

Insights on predator and South Island robin behaviour from nest cameras

4.1 Abstract

Using cameras at nests provided accurate information on nest outcomes, times and causes of failure. One-hundred and sixty-seven South Island robins (*Petroica australis*) nests were monitored by camera between 2012/13 - 2016/17 at Tennyson Inlet, Marlborough Sounds, New Zealand. From 83% of nests the nest outcomes was captured by the camera.

Rats (*Rattus* spp), stoats (*Mustela erminea*), possums (*Trichosurus vulpecula*), long-tailed cuckoo (*Urodynamis taitensis*) and possibly a mouse (*Mus musculus*), were all found to prey on SI robin nests. Diurnal differences in hunting practises were observed, with stoats taking nest solely by day and rats by night. Nests were most commonly preyed on by rats, with rat predations making up 60 % of all identified failures. Rats were also the sole predator to cause death to brooding or incubating females.

Natural failure events made up 3 % of nest failures. The number of live fledglings produced per nest attempt was on average 0.52. This varied between years and was correlated with predator pressures. The number of fledglings produced increased in the later part of the breeding season. The re-nesting interval did not appear be affected by the nest stage at which the previous nest failed, and the interval between nesting attempts decreased in the later stages of the breeding season.

The use of nest cameras assisted in assessing the effectiveness of aerial 1080 applications by identifying the nest predator and changes in predator dynamic. Without the use of cameras, the cause of nest failure events could not have been accurately observed nor the ability to identify rats as the biggest threat to SI robins in the Marlborough Sounds.

4.2 Introduction

The use of cameras for nest observations

Finding and monitoring nests is a vital part of ornithological studies especially if an understanding of population health is to be obtained (Jehle et al., 2004, Armstrong et al., 2002, White, 2000). SI robins are easily trained, as they are naturally curious and respond to a tapping noise if they associate this with being rewarded with a meal worm (*Tenebrio molitor*). As males feed their females during the breeding season (Higgins and Peter, 2002), nests can readily be located once birds are trained to respond to tapping. A male will feed his female on the nest or call her off to feed her (Boulton et al., 2010, Higgins and Peter, 2002). Females spend the majority of time on the nest during incubation (Powlesland, 1983), and for the early chick period, progressively decreasing the brooding time with chick age (Boulton et al., 2010, Higgins and Peter, 2002).

Nest observations are traditionally conducted by observing the nest from a distance, and interpreting the nest age and state, based on behavioural observations (Powlesland, 1997). To determine the time of a nest failure event, either the failure event is observed or nest success is estimated in a way that account for uncertainty in the timing of failure (Johnson, 1979). Nest age can be estimated based on observed nest building, laying, commencement of incubation or hatch dates, egg candling and aging of chicks (Gage and Duerr, 2007).

In regards to nest failures, if the nest can be accessed, then the nest contents can be examined to determine the cause of failure (Brown et al., 1996a, Moors, 1983). However, such observations are not reliable when attempting to identify the predator responsible for the failure event (Brown et al., 1998, Brown et al., 1996a, Major, 1991). In recent decades additional methods were trialled to help identify nest predators. With technological development, motion-triggered video and still footage became the method of choice for endangered and threatened species management (Little et al., 2017, Sanders and Maloney, 2002, Brown et al., 1998, Major, 1991). Cameras can accurately identify the nest outcome and cause of failure, they are small, light, have a long battery life, are waterproof, and use infrared to capture images of predators at night. All images are linked to a trigger time, allowing the time of failure as well as the predator responsible to be captured. Limitations include fog (which blares out the image) and a need to climb the trees to mount the cameras within reasonable distance from the nest to retrieve reliable footage. Complete reliance on cameras cannot be obtained, so weekly or fortnightly nest checks are still necessary to maintain battery life and check general camera function.

SI robin nest predators and threats to incubating/brooding females

New Zealand robins have been the focal point of many research studies, assessing predator impacts and their breeding behaviour (Higgins and Peter, 2002). From our literature search only five species have been observed to attack robins during the nesting period causing loss of eggs, chicks and adult females (Brown, 1997, Jones, 2016, Powlesland et al., 2000). The five predators are ship and Norway rats (*Rattus norvegicus*), possums, stoats and ruru (*Ninox novaeseelandiae*) (Jones, 2016, Powlesland et al., 2000, Brown, 1997) though mice (*Mus musculus*) and cats (*Felis catus*) are mentioned in Higgins and Peter (2002). Long-tailed cuckoos (*Urodynamis taitensis*), falcons (*Falco novaeseelandiae*), weka (*Gallirallus australis*), weasels (*Mustelinae spp.)* and ferrets (*Mustela putorius furo*), also likely predators do not feature in published literature as nest predators of NI and SI robin nests.

SI robin productivity

The majority of SI robin studies have been conducted as part of pest control evaluations or as monitoring techniques before and after translocation (Jones, 2016, Schadewinkel et al., 2014, Armstrong et al., 2000, Etheridge and Powlesland, 2001). This means that few studies were conducted in hard to reach areas or in larger forest networks. This includes an emphasis on fragmented landscapes, islands and habitat at lower altitudes. Research has been conducted linking time of year to periods of higher productivity and juvenile survival rates (Drummond, 2017, Dimond and Armstrong, 2007, Higgins and Peter, 2002), and a positive correlation was found between invertebrate biomass and robin productivity (Boulton et al., 2008). Invertebrate availability is affected by forest makeup, temperature and water availability. The higher the competition for food resources the harder survival becomes, especially for non-generalist species with limited and small territories. Rodents, stoats and possums all consume and compete with SI robins for insects, especially weta (*Hemideina sp*), which make up a significant part of their diets (McQueen and Lawrence, 2008, Sweetapple and Nugent, 2007, Martinoli et al., 2001, Miller and Miller, 1995, Murphy and Dowding, 1994). Pest control should alleviate competition and increase productivity and juvenile survival, not just through reducing predation risk but also by increasing food supply.

Re-nesting capabilities of SI robins

Few data have been recorded on the time between nest attempts (Powlesland et al., 2000, Armstrong et al., 2000, Powlesland, 1983), and it is largely unknown if a female is quicker or slower to re-nest after a failure at egg stage versus chick stage. Females in general take a lesser role after fledging than males (Higgins and Peter, 2002). If three chicks are fledged females tend to feed one of the three fledglings while males two (Higgins and Peter, 2002). When larger clutches are fledged the females post fledging involvement presumably increases the number of days between nest attempts as her energy levels will go to raising fledglings rather than starting a new nest attempt. When two chicks fledge, the male either takes on both or the fledglings are split between the parents (Higgins and Peter, 2002).

The time interval between nest attempts is probably affected by the nest stage and age at which the previous nest ends. It is possible that a female is biologically more ready to relay when the previous nest successfully fledged, rather than if the previous nest fails part way through the incubation or chick rearing stage. Similarly, the re-nesting ability may be affected by time of year.

Different predators appear to cause nest failures at different nest stages (Brown et al., 1998), and may differ in whether they kill the female as well as the eggs or chicks. To estimate the productivity potential of SI robins with variation in timing of pest-control treatments, the females' ability to re-nest after different predation events needs to be understood. If her ability to re-nest is linked to nest age at the time of failure, or time of year, priority may have to be given to adjusting the timing of pest control operations, especially when SI robin reproductive outputs are compromised due to the loss of females during nest predation events.

4.3 Method

The 'General Methods' chapter (**Chapter 2**) contains a detailed description of the study area and the aerial 1080 treatments that apply to this chapter. The following analytical methods specifically concern aspects of the current chapter.

Data used for analysis of camera footage for identification of predator, time of day and attacks on females

Data used for this section were sourced from preserved camera footage. From the five breeding seasons monitored, all captured photos were available for 2014/15 - 2016/17, with only selected images available for the 2012/13 and 2013/14 breeding seasons. Nest descriptions and observations stored in both the database and personal note books were also used, to give a complete picture of the observations made at each nest failure event. Time of day for the nest failure events was taken from camera footage. The timing of sunrise and sunset is not consistent for dates between years. For ease of analysis the time of sunrise and sunset was used for all nests as Wellington (41° S, 175° E) standard time 2016. Where appropriate, observations from the treatment and non-treatment areas were lumped, such as the 2013 breeding season pre any pest control treatment.

Nest location types

All nests were allocated one of the eight nest site descriptions. Nest site descriptions were gave an indication on how restricted the females escape route was, during a nest predation event.

Nest location types:

- **'*branch*':** a nest on a branch was rated as having the highest escape potential as it gives multiple angles of obstruction-free escape for the female
- **'*fork*':** a nest on a branch or trunk fork leaves multiple escape routes in the opposite direction of the predators approach
- **'*ledge*':** a nest perched on the side of a trunk or branch leaves multiple escape routes in the opposite direction of the predator's approach
- **'*top*':** a nest perched in the top of a tree fern may have some escape obstruction due to the array of fronds
- **'*twigs*':** a nest in the outer branchlets has restricted escape potential
- **'*nook*':** a nest in an open cavity such as a shallow knot hole or inside an open but hollow branch. The predator entry and the females escape route are at the same location, and though the area is large, it could restrict the females escape potential
- **'*vegetation*':** a nest under *Astelia* bushes or similar could reduce escape potential due to collision or entanglement with vegetation during escape
- **'*hollow*':** a nest found in a hollow branch or trunk with a single entry/exit point has low escape potential as the escape direction is the same as the direction from which the predator approaches.

Data sources for productivity and re-nesting ability of South Island robin females

Time intervals between nest attempts were calculated based on the observed failure or fledge date and the approximate first day of the commencement of nest building of the subsequent nest. These values are approximate, as few nests were observed from the start of nest building right through to the day the nest ceased. During the breeding season one or two checks a week were conducted for all breeding birds. Pairs likely to re-nest were prioritised, in the hope of capturing new nests early.

Both literature and our own observations indicated that the time taken for nest building varies from about 1 - 7 days (Higgins and Peter, 2002). This variability was not included in our approximations of re-lay intervals and all nests were strictly, unless observed otherwise allocated two days for building.

Data from the treatment and non-treatment areas were collated for this section of the analysis. As a 1080 treatment is unlikely going to affect the speed at which a female re-nest. Due to reduced data collection at the start and end of the breeding season our dataset is somewhat limited. January-February nest observations were only conducted in 2015/16 and 2016/17. The 2015/16 breeding season was the only entire SI robin breeding season observed from late-August to mid-February. This limited data for the earlier and later part of the breeding season.

Active nests can either fledge or fail

Each nest once deemed active, could either fledge or fail. For our data, no individual nest was able to fail more than once. The cause of the failure was identified based on the event prior to the failure, even if multiple predators visited only a single predator was identified as causing the failure.

4.4 Results

Camera captured attacks on South Island robin nests

In total 167 of the 210 confirmed active nests (80%) were fitted with cameras (**Table 4.1**). Over the course of the study period each year, the treatment area had a lower proportion of nests fitted with cameras than the non-treatment. Between the years, proportions of nests fitted with cameras varied from 54% to 100%. The cameras captured the nest outcome in 139 of the 167 nests (83%) fitted with a camera (**Table 4.2**). The remaining 28 cameras failed to capture the nest outcome due to battery failure, camera failure and human error.

The camera captured nest outcomes include: 50 fledging events, one human-caused abandonment, 13 natural abandonments and 75 predation events, including native and introduced predators. The predation events were caused by possums (12), stoats (16), rats (45) and ruru (2). The human-caused nest failure was removed from the data presented, leaving a dataset of 50 fledging events and 88 nest failures.

Changes in predator composition and predator caused failures appear to be visible (**Figure 4.1**). Before aerial 1080 operations possums cause of nest failures in the treatment though not after, while rats the main cause of nest failures after aerial 1080 operations and in the year of the beech mast. Two years after the beech mast in the non-treatment area stoats become a dominant threat to SI robin nestlings, this is not observed in the treatment area.

Table 4.1: The number of South Island robin nests fitted with cameras over the course of the study period 2012/13 to 2016/17. The treatment area has consistently fewer nests with cameras than the non-treatment area.

Area	Year	Total nests	Camera used	% of nest with cameras
Non-treatment	2012/13	12	11	92
Treatment	2012/13	11	10	91
Non-treatment	2013/14	18	17	89
Treatment	2013/14	17	15	88
Non-treatment	2014/15	25	20	80
Treatment	2014/15	35	23	64
Non-treatment	2015/16	21	18	78
Treatment	2015/16	26	14	52
Non-treatment	2016/17	24	22	88
Treatment	2016/17	21	17	81
Total non-treatment	**all**	**100**	**88**	**88**
Total treatment	**all**	**110**	**79**	**72**
Total	**all**	**210**	**167**	**80**

Table 4.2: Camera-captured predation events causing South Island robin nest failures in the non-treatment and treatment area over the course of the study period (2012/13 - 2016/17) in Tennyson Inlet. From the remaining nests, 50 fledged, one failed due to human disturbance and 13 failed due to natural abandonments.

Camera-captured predator	Total occasions	Treatment	Non-treatment
ruru	2	1	1
rat	45	26	19
stoat	16	3	13
possum	12	3	9
Total	75	33	42

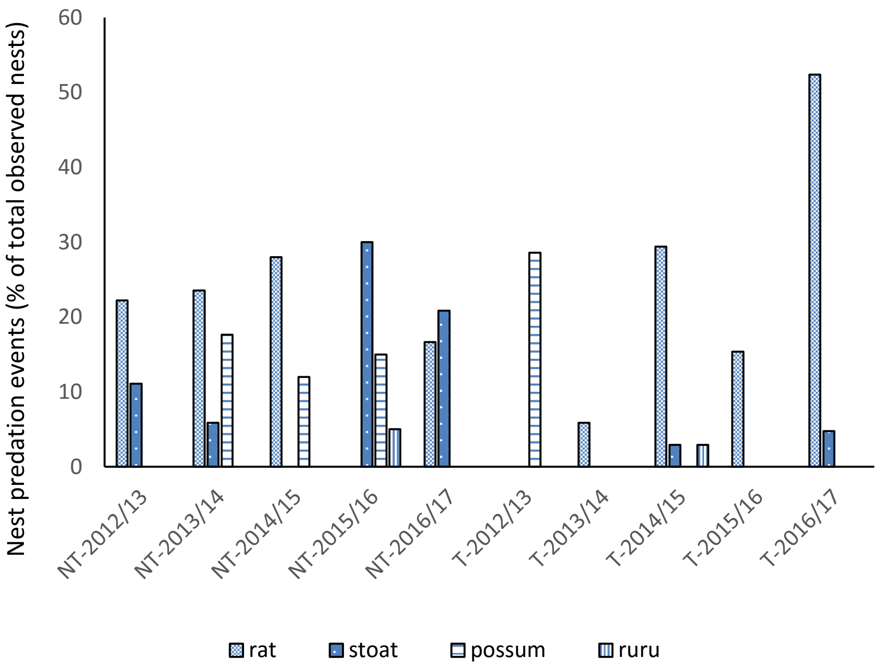


Fig. 4.1: Predation events as a percentage of all observed nests over the course of the study period (2012/13 to 2016/17) which caused South Island robin nest failures. Presented here in years and by area; non-treatment (NT) and treatment (T). Changes in predator presence and composition appears to be visible before and after beech mast (2014/15) and before and after aerial 1080 treatments which were conducted in November of 2013/14 and 2014/15 in the treatment area only.

Camera-captured timing of nest attacks by different predators

Sufficient data were available from 68 nests to assess the time of day at which a nest attack by a predator occurred. All stoat nest attacks occurred during the day and all rat nest attacks occurred at night (**Fig. 4.2**). Possums, which are nocturnal, predominantly attacked nests during night-time hours, with one day time event. Similarly, ruru are also predominantly nocturnal, but one of the two nest attacks observed occurred during the day, the other by night (**Table 4.3**). The two unexpected day-time attacks occurred close to the shortest night of the year (22 December).

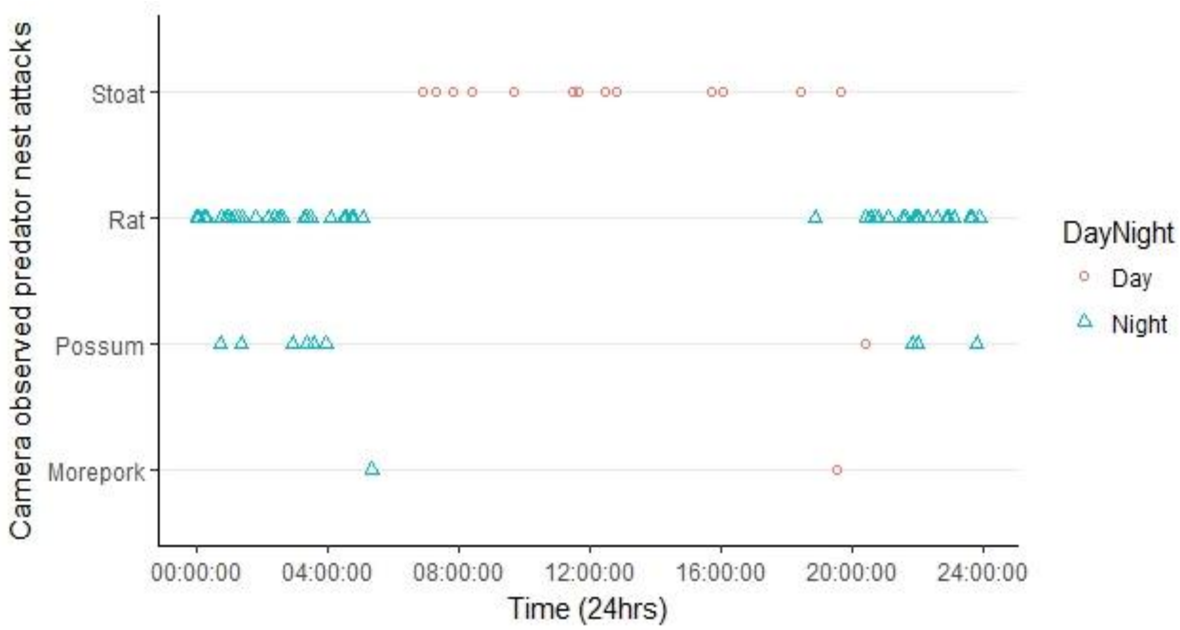


Fig. 4.2: The 68 nest predation events over the course of the study period (2012/13 to 2016/17) divided into 'night' and 'day' events and by predator type. The daylight hours change dramatically over the nesting season (September – February) with ~ 11 h 9 min of daylight on 1 September and ~ 15 h 10 min on the longest day 22 December (based on Wellington standard time for 2016). Attacks after sunset are coded as 'night' and attacks after sunrise as 'day'.

Table 4.3: Predation events on South Island robin nests that occurred outside the expected active period of the predator. These constitute 10% of predation events by possums and 50% of those by ruru (see Table 4.2).

Predator	Time of nest attack	Date	Time relative to sunlight	Day/night
possum	20:27	16-Dec-2012	23 min before sunset	day
ruru	19:34	1-Jan-2015	1 h 23 min before sunset	day

Changes in predation pressures on South Island robin nests

Nest failures attributed to the three key introduced predators (possums, rats and stoats, from here on referred to as the trio) accounted for 54% of the 72 monitored nest outcomes in the non-treatment area from 2012/13 – 2016/17. In the treatment area they account for 46% of 65 monitored nest outcomes.

In the absence of pest control the trio were responsible for 45 – 62% among years. The highest year was 2014/15, this coincided with the year of the beech mast, at 62%. The impact of the trio on SI robin nests was very similar among the years. However, the relative contributions of nest attacks by rats, stoats and possums varied between them.

Rats were on average responsible for 32% (SD 0.012) of all nest failure events between 2012/13 and 2014/15 in the non-treatment area, comparatively stoats for 8% (SD 0.083). The beech mast in 2014/15 changed these proportions, one year after the beech mast rats were responsible for 0% of SI robin nest failures and stoats 40%, two years after rats were responsible for 25% and stoats 31%. These nest threat changes among the trio are even more pronounced when natural nest failures and unidentified nest predation events are ignored.

In the treatment area this rise in stoat-caused failures was not observed, though the trio were responsible for 58% of the 19 nest outcomes during the beech mast before the aerial 1080 drop, after the 1080 operation one of 11 nests failed due to a rat. One year after the pest control effort rats were responsible for 31% of the observed nest failures and 61% in 2016/17 two years after the pest control effort. Stoats contributed 0% 1 year after and 6% 2 years of failures after pest control. In the presence of 1080 pest control, rats remained the key cause of nest failures.

Females killed on the nest

Camera-confirmed attacks on female SI robins were limited, with cameras not reliably capturing contact between the female and the predator. For these reasons, notes on the females' wellbeing and feather remains in the nest were used to identify additional incidents. It is likely that our estimation of non-fatal attacks on females is lower than in reality, while the number of females killed due to predators is reliable.

Of the 167 camera observed nests 138 nests had sufficient notes to assess the nest outcome, we collected camera footage of eight females (5.80%) being attacked during a nest failure event, 2012/13 - 2016/17. Each event occurred at night and was caused by a rat. If the nest was attacked by a rat, the female had a 13% chance of death, with the remaining females often escaping with just feather loss (**Table 4.4 - 5**).

Table 4.4: Over the course of the study period (2012/13 – 2016/17), 5.80% of South Island robin females received injuries during a nest predation event.

	Observed	Total (%) N = 138
Female deaths	6	4.35
Injured females inc. death	8	5.80

Table 4.5: During 75 predation events enough footage and notes were collected to determine the impact on the females-health, after a predation event. Rats caused 60% (N=45) of these predations and were the sole cause of death and sole predator to inflict injury to females during a nest predation over the course of the study period (2012/13 - 2016/17).

	All predations	Rat caused predations
Female deaths	8.00	13.33
Injured females inc. death	10.67	17.78

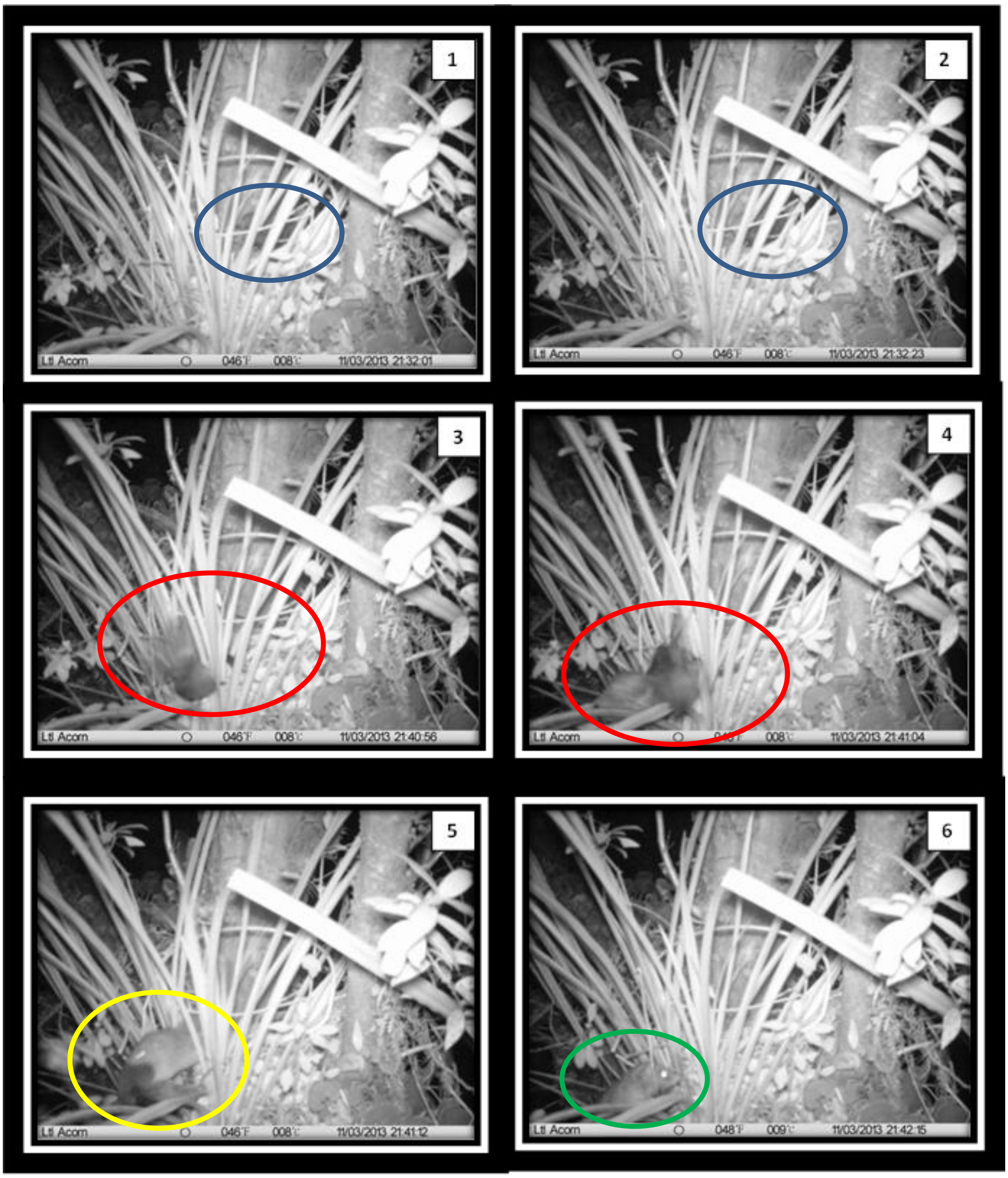
The footage and remains at the nest were all different among the eight observed incidents. In some events only the tail feathers remained in the nest, indicating contact between the rat and robin (**Plates 4.2**), in other events the female was seen to flap her wings for some time before escaping (**Plates 4.1**) and on two occasions the female's partially-eaten body remains were found in the nest (**Plates 4.3**).

Not all females were killed on the nest during the event. Four females subsequently disappeared, presumably dying due to infections or fell easy prey to subsequent predators, whereas two recovered and remained in the breeding population.

The female shown in **Plate 4.1** (xM/VP) appears to have died from sustained injuries soon after the attack, as she was not observed again after these images were captured. The rat had hold of the female for 8 s, as seen on camera footage. It appears that the rat was unable to keep his/her hold on the female robin, as she flapped her wings and eventually managed to escape. Her body was not recovered.

Of the two females subsequently seen after attacks, OM/YY was seen alive two days after the attack and female PM/YP was seen alive five days after the attack. However, in both cases the females were subsequently never seen again, and likely died due to injuries sustained. Female OM/YY was observed to have severe injuries to her eye and head (**Plate 4.2 - 3** and **Table 4.6**) and reluctant to fly, hopping along the forest floor while her mate fed her. No notes where recorded on the appearance of PM/YP, suggesting she appeared normal when last seen five days after the nest attack.

Rats were responsible for 30.6% of all nest failure events and 100% of all female deaths during these events, and killed ¾ of all the females they attacked. The proportion of females killed on the nest appear to depend on the nest location. Nest locations chosen by SI robins in our study areas were divided into eight different types. Four nest locations were assumed to restrict the female's escape (*twigs, nook, vegetation* and *hollow*). These restricted nests made up 33% of the 185 nest locations chosen by SI robins (**Table 4.6**). In restricted nest locations females were at a 12% higher risk of injury or death during a nest predation event than in more exposed nests.



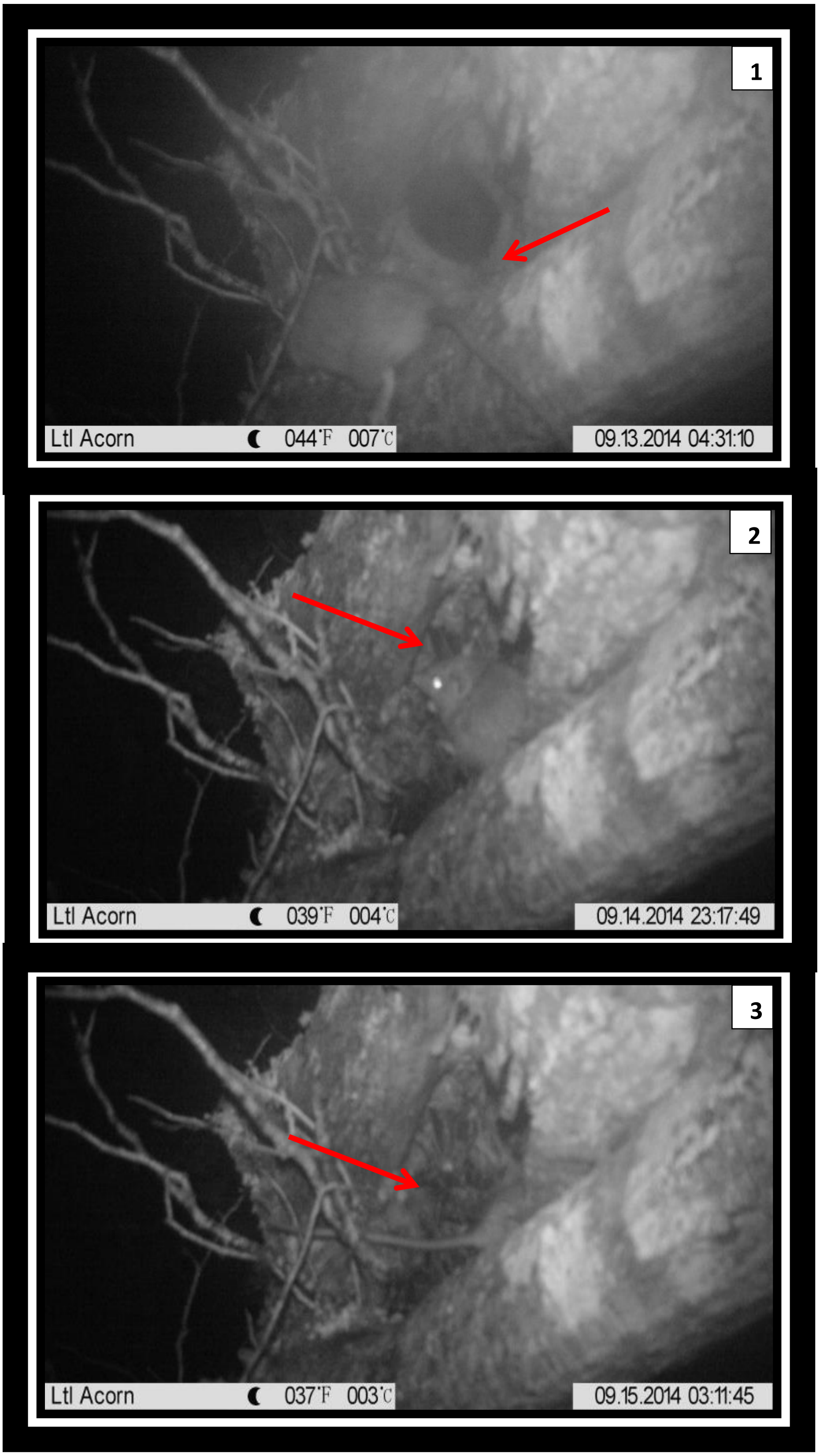
Plates 4.1: Photo sequence of a rat attack on a South Island robin nest (ID: RC411) on 3 November 2013. The robin female (ID: xM/VP) was not seen again so is presumed to have been killed or fatally injured. Contact between the rat and the female robin is evident from the photo sequence, with a total of 8 seconds where the rat presumably holds the female before she presumably gets away. *Image 1 & 2*: female robin incubating at 21:32:01 – 21:32:23, shown in blue oval. *Image 3 & 4*: female robin struggling to escape 21:40:56-21:41:04, shown in red oval. *Image 5*: rat holding the female robin around the body for a minimum of 8 s 21:41:12, shown in yellow oval. *Image 6*: rat ‘returns’ to eat the nest contents 21:42:15, shown in green oval.

Plates 4.2: Photos taken two days after a nest attack occurred. Images show a trail of South Island robin body feathers in and around nest (ID: RT711) on 3 November 2016. Female OM/YY was found the same day with severe injuries to eye and head; she was never seen again. A rat was captured by camera attacking the nest from above.

Plate 4.3: Photos of female OM/YY two days after her nest was attacked by a rat (see Plate 4.2). Visible damage included severe damage to the back of the skull, with bald patches visible, and loss of the right eye. During observation the female stayed on the ground, with her male feeding her.

Table 4.6: Over the course of our study period (2012/13 – 2016/17) a total eight predation events on camera-observed nests involved South Island robin females. ‘Restricted’ nest locations are nest sites that limit the number of directions in which a female can easily escape the nest, if spooked or under attack.

Female ID	Predator	Live/die	Body recovered	Nest stage	Nest location
xM/VP	rat	Killed	no	7 day chicks	Vegetation (under *Astelia)*
PM/YP	rat	Killed	no	incubation	fork
xM/WW	rat	Killed	yes	incubation	hollow trunk
xM/LK	?	Killed	yes	incubation	nook
xM/KW	rat	Lives	N/A	incubation	nook
OM/RB	rat	Lives	N/A	12 day chicks	hollow trunk
OM/YY	rat	Killed	no	incubation	hollow trunk
xM/BP	rat	Killed	no	incubation	hollow trunk



Plates 4.4: Images of female xM/WW killed on the nest (ID: RT506) 13 September 2014. *Image 1*: 13 September, female incubating and rat arrives 04:31:10 (arrow pointing at female's bill). *Image 2*: 14 September, female tail feathers visible with rat eating in nest. *Image 3*: 15 September, mass of feathers visible and rat seen in nest.

Table 4.7: A summary of the observed nest sites selected by South Island robin females and the associated risk that the female is attacked during a nest predation event involving a rat. Nest locations are divided into eight location types and the escape potential for the female ranked from extremely high to very low.

Nest location	Escape potential	No. of nests	females attacked	Risk level (% of nests)
Branch	extremely high	3	0	0.00
Fork	very high	74	1	1.35
Ledge	high	38	0	0.00
Top	high	11	0	0.00
Unrestricted		**126 (67.38%)**	**1**	**0.8%**
Twigs	some	4	0	0.00
Nook	some	14	2	14.29
Vegetation	moderate	7	1	14.29
Hollow	low	36	4	11.11
Restricted		**61 (32.62%)**	**7**	**11.5%**

Productivity of South Island robins in the Marlborough Sounds

Over the course of the study period (2012/13 - 2016/17), 210 nests were observed, of which 59 (28%) fledged at least one chick, with a minimum of 111 chicks fledged in total. As our study was not focused on productivity, and not all nests per female were monitored. Our productivity data presented here are based on the number of fledglings produced per observed nest attempt. The average number of fledglings per observed nest was 0.52. The lowest rate (0.15 fledglings) was observed in the non-treatment area in the breeding season of the beech mast (2014/15). The highest (0.96 fledglings) were observed in the treatment area in 2015/16 the season after the beech mast and 1080 treatment (**Table 4.8**).

Two pest control efforts were carried out. The first occurred on 2 November 2013/14, the second on 23 November 2014/15. Both events are about halfway through the SI robin breeding season, (the first eggs was laid in late August and the last in mid-January). This division allows for a before and after 1080 treatment comparison for the number of fledglings produced per nest. The number of fledglings per observed nest attempt is consistently higher in the treatment area, and both the treatment and non-treatment areas show an increase from the early breeding season (before), to later part of the breeding season (after) irrelevant of 1080 treatment (**Table 4.9**).

Table 4.8: The average number of South Island robin fledglings produced per observed nest attempt in each breeding season in the treatment and non-treatment area.

Year	Non-treatment		Treatment	
	No. nests	Fledged/nest	No. nests	Fledged/nest
2012/13	12	0.83	10	0.30
2013/14	19	0.37	14	0.86
2014/15	26	0.15	34	0.47
2015/16	26	0.35	27	0.96
2016/17	25	0.72	27	0.29
Total	108	0.44	106	0.59

Table 4.9: The average number of South Island robin fledglings produced per observed nest attempt before (early breeding season) and after 1080 treatment (later breeding season) in both the treatment and non-treatment areas.

Area	Year	Before treatment		After treatment	
		No. nests	Fledged/nest	No. nests	Fledged/nest
Non-treatment	2013/14	12	0.10	7	0.86
	2014/15	16	0.13	10	0.20
Treatment	2013/14	6	0.38	8	1.50
	2014/15	23	0.26	11	1.00

Re-nesting capabilities of South Island robins

In total we made 109 observations of a SI robin re-nesting during a breeding season, 56 from the treatment and 53 from the non-treatment area. There was no reason to believe that the interval between the nest attempts should be different between the two areas, though more nest attempts are likely to occur with increased failure rate/predator pressure. For this reason, the data were combined from all years, treatment and non-treatment areas. Of the 109 re-nest observations, 21 re-nests occurred after a fledging event.

The average interval between two observed nesting attempts was 8.5 days (+/- 0.71 SE) and ranged from zero to 46 days (**Figure 4.3**). There is a significant difference between the time the female took to re-nest depending on the previous nest's outcome (t = -3.4385, df = 22, p = 0.002). The average number of days between re-nests rose to 15.3 from 6.9 days when the previous nest was successful rather than having failed.

In 2016 female M/LO used both an adjacent male and her regular partner to raise her fledglings. She took no active role in the raising of her fledglings. She appears to have mated with the adjacent male for the 2nd clutch and with her partner for the 1st, 3rd and 4th clutch. The males split the three fledglings from her first clutch, with the adjacent male taking on one and her partner two fledglings. This allowed her to raise four successful clutches, each of two or more chicks within a single breeding season.

In another case, a single male stepped in and helped to raise a clutch from chick stage onwards after the father disappeared. On another occasion, a father disappeared during the incubation stage and a single male claimed the territory, though he was never seen to assist with the clutch. She managed to raise one chick to fledging, though the chick-nest stage was 3-13 days longer than expected.

It should be noted that the two longest time periods between nest attempts (re-nest intervals) were 38 and 46 days. These appear to be outliers. As the mean and mode for the re-nest interval after a successful nest was 16.2 and 16.0 days respectively, there is no concern for non-normality in the data distribution, and the outliers were not removed.

We found a relationship between the re-nest interval and the previous nest age (t = 2.686, p = 0.008) (**Figure 4.4**), where the re-nest interval increased in length with age of the previous nest. The later in the breeding season the shorter the re-nest interval (t = -2.940, p = 0.007) (**Figure 4.5**).

A positive correlation between the re-nest interval and the number of fledglings produced in the previous nest was observed (t = 5.324, p < 0.001) (**Figure 4.6**). This means that the time/energy demand on the female is likely higher if the nest fledges multiple chicks rather than one. Similarly, from our data if a single chick fledges or the nest fails, the focus for the female appears to be to re-nest, decreasing the re-nest interval.

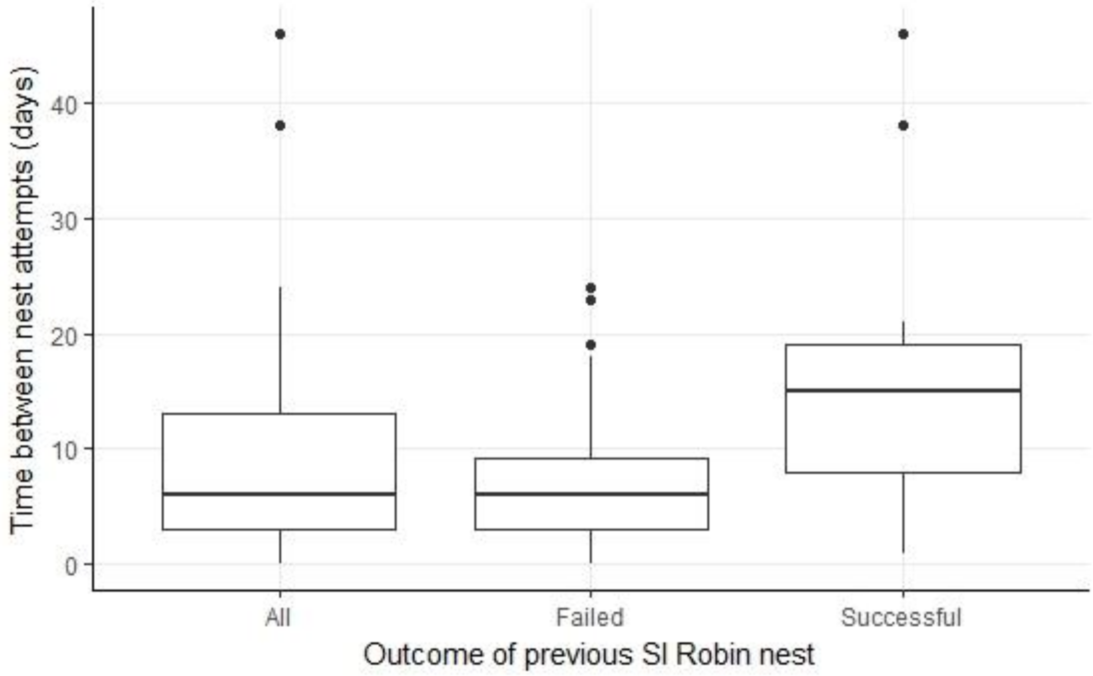


Fig. 4.3: Re-nest intervals after a previous nest failed or fledged. ‘Failed’ means the previous nest failed (N=88) and ‘Successful’ means the previous nest had a successful outcome, and fledging at least one chick (N=21).

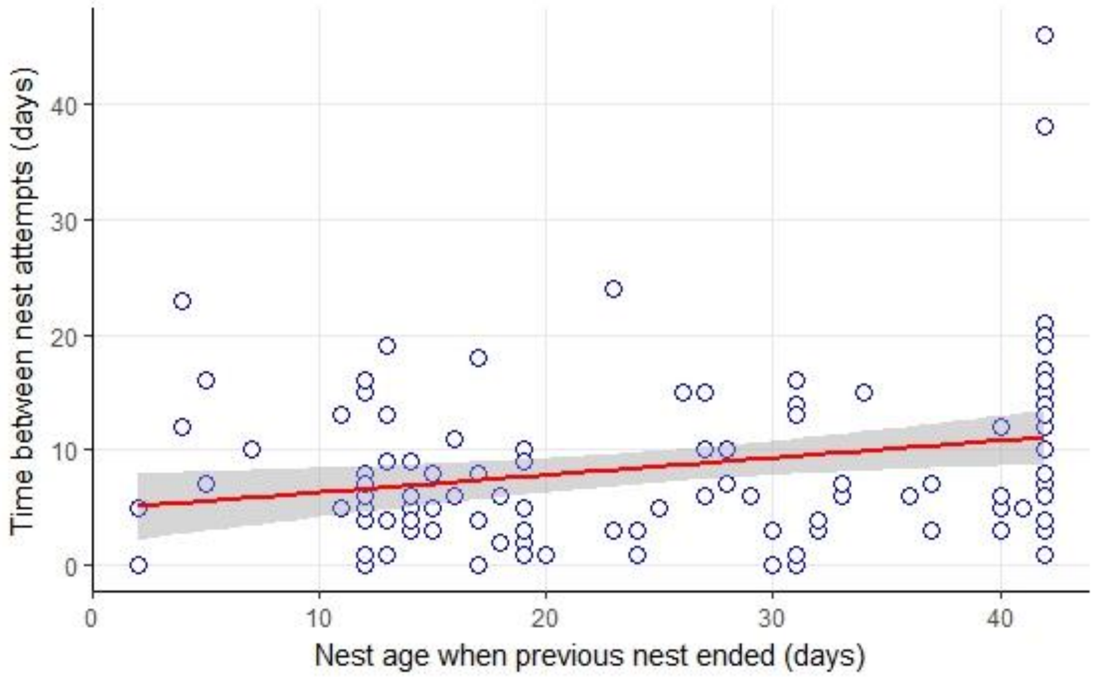


Fig. 4.4: Relationship between the previous nest age at which a nest attempt ceased and the re-nest interval before the next nest is commenced for South Island robins (N = 109).

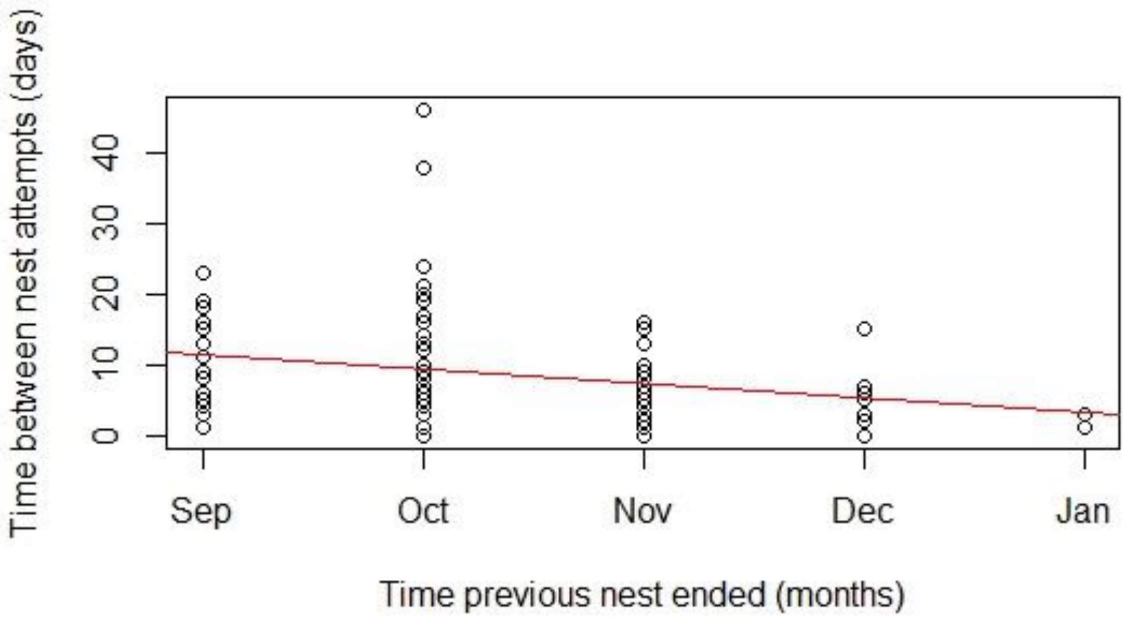


Fig. 4.5 Relationship between the month in which the previous nest attempt ceased and the length of the re-nest interval (N = 109).

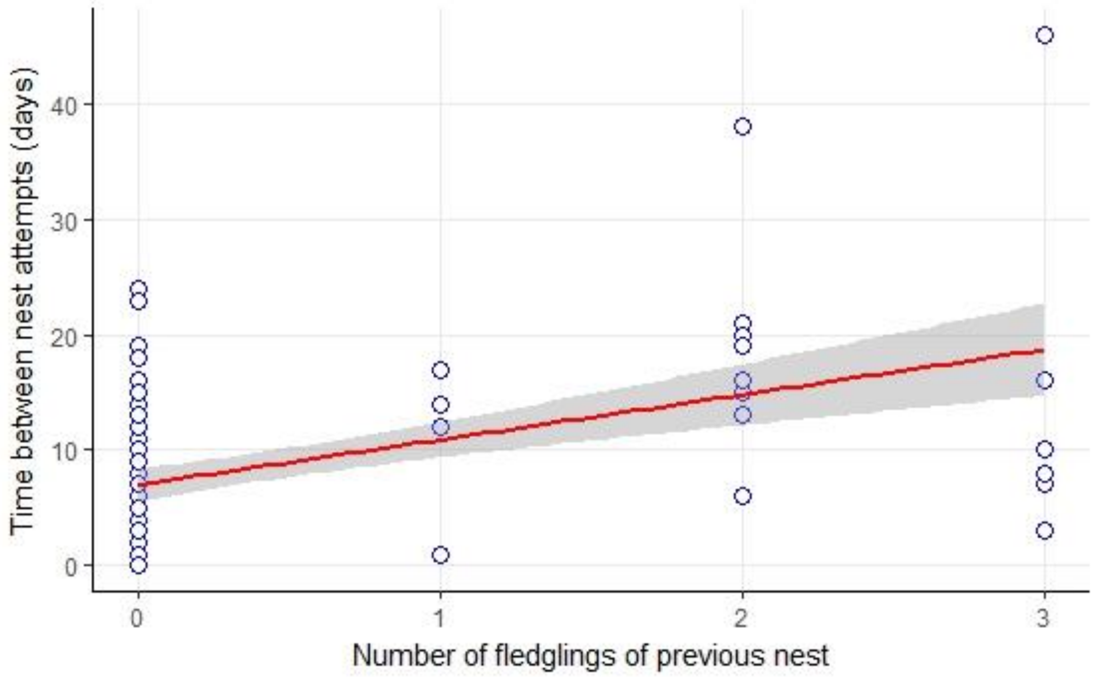


Fig. 4.6: The number of fledglings fledged from the previous nest attempt has an effect on the Re-nest interval for South Island robins in relation to the number of fledglings fledged from the previous nest attempt (N = 109).

Unusual visitors at South Island robin nests

We had no nest failures caused by more than one predator arriving at a nest site at the same time. However, we did observe two failure events for the same nest. Nest ID: RC520 initially failed due to a rat on the 22 November 2014 and on the 25 November possibly due to a mouse. On the 22 November a rat destroyed the nest content (one egg) leaving an empty nest cup, this was recorded as a failure and used for further nest success analysis. The female then returned to her empty nest and laid additional eggs allowing for a second failure to occur at the same nest site.

This second nest attempt was not used for further analysis and was not included in our nest success analysis. Images captured on the 25th are not 100% clear, but a rodent is easily identifiable, while the rodent appears round in body shape and is estimated at ~ 6 cm (**Plate 4.5**), uncertainty remains on the identification of the rodent. Was it a small rat or a large mouse?

The second unusual nest predator was a long-tailed cuckoo (*Urodynamis taitensis*) visiting a SI robin nest on 21 December 2015 (nest ID: RC616) (**Plate 4.6**). Oddly in this case incubation continued until 28 December when the nest was abandoned. The nest content decreased from four eggs to one between the 21 and 30 December without further predators caught on camera, leaving a degree of uncertainty as to the cause of the nest failure.

Frequent visits from korimako (*Anthornis melanura*) and titipounamu (*Acanthisitta chloris*) to SI robin nests were captured by camera after a nest failure event. It is unclear to why korimako visited SI robin nests, though titipounamu were interested in feathers and were often captured doing multiple return trips removing feathers from the failed nest. We did not capture the 'stealing' of feathers of an active SI robin nest.

Plate 4.5: A small rodent approaching a South Island robin nest (ID: RC520) on 25 November 2014, eye of rodent clearly visible above short arrow on the left of image. The nest had already failed once due to a rat three nights prior, but the female had laid 2 - 3 additional eggs before this small rodent caused a second and final failure at this nest site. It is possible that this image is the first capturing a SI robin nest predation by a mouse. The images are not clear enough for verification purposes, and with the rodent being estimated at 6 cm from nose to start of tail, it is possible that it was actually a small rat.

Plate 4.6: A long-tailed cuckoo apparently removing eggs from a South Island robin nest (ID: RC616) on 21 December 2015. The nest content went from four eggs to one. The nest remained active with incubation continuing until 28 December. During the week following the visit from the cuckoo no other disturbances were noticed and the female appears to abandon the nest by her own free will. It is therefore possible that the remaining egg was infertile or the egg of the cuckoo, see **Plate 4.7**. No special attention was given to the remaining egg when the camera was removed, leaving infertility or cuckoo egg up for debate.

Plate 4.7: A comparison between the eggs of long-tailed cuckoo and South Island robins. **Left**: Image of a long-tailed cuckoo egg measuring 22.4 x 17.2 mm laid near Wellington in 1964. **Right**: Image of a South Island robin egg measuring 25.5 x 18.9 mm laid near Kaikoura and collected by Douglas Flack in 1971. Both images are © Te Papa by Jean-Claude Stahl and freely available through TePapa online resources.

4.5 Discussion

The use of cameras for nest observations

Cameras proved a valuable tool to assess the efficiency of the aerial 1080 operations and compare nest outcomes between the areas and years. We had a high success rate from cameras, with 83% of set cameras capturing the nest outcome.

Proportionately the treatment area had fewer mounted cameras than the non-treatment area each year of the five-year study. This is largely due to differences in canopy height and therefore accessibility of nests. The proportion of camera malfunctions to capture the nest outcome was similar between the two areas.

Camera captured timing of nest attacks by different predators

Stoats, possums and rats are thought of as nocturnal predators (Sullivan, 2004), though increasing numbers of observations have been made of stoats hunting or at least exploring prey options during the day (Little et al., 2017). These diurnal behaviours possibly reflect the predator's hunting strategy. Stoats may hunt more by sound, and are reported to commonly take nests at chick stage (Jones, 2016), while rats predominantly take nests at the egg stage. Our observations support earlier finding of these patterns, though possibly more strongly than has been observed in other areas and species, as we observed no stoat predations at night and no rat predations during the day.

Camera captured attacks on South Island robin nests

Our observations concur with other studies, that pest control treatments successfully suppressed nest failures attributed to introduced mammalian predators (see **Chapter 5** for analysis of nest success rates). We found that the proportion of nests attributed to mammalian predators was lower after an aerial 1080 application in a non-mast year then in a mast year. The rate of natural failures varied between years. This means that without identifying the cause of the failure and accounting for natural failures the treatment effect may be falsely interpreted. Also, the proportion of nest failures attributed to mammalian predators one year after the mast (2015) in the treatment and non-treatment area was very similar. However, the species responsible for the nest failures were different. In the treatment area the main predator one year after the beech-mast and aerial 1080 treatment was the rat, the same year in the non-treatment area most nest failures were caused by stoats. This reflects the relationship and flow on effects of the beech mast events. In year one, an increased food supply due to the beech-mast, temporarily supported an elevated rodent population into year two. This was followed by an increase of stoats in year three due to the increase stoat food supply (rats and mice).

In the treatment area the use of the 1080 removed the stoat plague, though a fast recovery of the rat population after the pest control operation was observed in year two. This is not the expected pattern for pest control in beech forests, where and rodent recovery usually takes several years.

Rats were responsible for 31% of all nest failures observe (treatment and non-treatment combined). In the absence of 1080 treatment rats made up 44% and stoats 33% of all failures caused by the trio of introduced predators. Rats were solely responsible for 23% of SI robin nest failures over the study period including all natural failures, abandonments and predators, with stoats causing 18% of all nest failures and possums 13% in the absence of pest control. Moors (1983) found that the majority of SI robin nests failed at egg stage. He went on to report that the primary cause of nest failures were mustelids, in particular stoats and weasels, these accounted for 77.9% of nest failures. The failures were attributed to each predator species based on sign left after a nest predation. Empty nests were attributed to mustelids (stoats and weasels) and egg fragments for rats and mice (Moors, 1983).

Moors' results are a reflection on sign left in nests after a predation event, not an identification of the nest predator responsible for the nest failure. A later study using a still camera revealed that rodents leave egg fragments behind in only 59% of nest predation events (Major, 1991). This means that the proportion of robin nests taken by rats is likely to have been underestimated by Moors (1983) or other authors not using cameras.

In more recent studies with cameras a clearer picture is forming of different bird species being sensitive to one or multiple predators, both on and off the nest. Weka (*Gallirallus australis)* for example are sensitive to stoats (Tinnemans et al., 2018). The predominant predators on a native species will vary between sites and times. For example, SI robins were found to suffer more stoat caused nest failures at chick stage than at egg stage (Jones, 2016). This differs from our observations, where most failures occurred at the egg stage and were attributed to rats.

In summary, it is important to note that our rat numbers increased to very high levels two years after the beech mast year pest control effort in our treatment area, exceeding their proportionate contribution to nest failures of all other study years. This is likely due to the low 1080 sowing rate in conjunction with extremely high rodent numbers and high immigration due to a small treatment area.

Variables like the number of years a study was executed over, the forest type, altitude of study area and the number of years since pest control efforts or a mast event are likely all large contributors to the predatory composition. From this study and others before it, we would advise that these variables are recorded and accounted for to provide more information to ultimately understand the true impact of introduced predators under different conditions on New Zealand's native bird species. The species identified as the key nest predator may simply be the most common predator at the particular time and place and therefore a key predator may vary over time and or between sites.

Females South Island robins killed on the nest

Other studies have reported that robin females become prey during nest attacks, though at very different rates. A 5.7% kill rate of NI robin females was observed in the Pureora forest in the absence of pest control (Powlesland et al., 2000) while another study found 41.6% of female NI robins killed in Kohara, Rotorua, also in the absence of pest control (Brown, 1997). We observed eight attacks on SI robin females out of 138 camera observed nest attempts, and these resulted in six female deaths. From our data, rats were responsible for 31% of all nest failures. We found that if a rat attacks a SI robin nest, that there was a 18% chance that the female was injured during the nest predation and a 13% chance that the female was killed by the rat. In most cases the females were able to vacate the nest without injury or contact with the rat.

Rats were the only cause of female loss on the nest and were found to exclusively hunt by night. It is possible that the behavioural trait of hunting primarily by smell and under the cover of darkness catches females unaware. Robin females may simply have a slower response rate to the disturbances at night or a greater reluctance to leave the nest than during the day.

On further investigation we compared the nest locations, giving each nest a rating on how restrict the female's escape route was (see above for classification of nests into "restricted" and "unrestricted"). We found that few (33%) SI robins selected restricted sites for their nests. However, if a restricted nest site was selected, then this nest site had an 12% chance of leading to an injured female. Rats were the only predator seen to cause mortality to nesting SI robin females. Therefore, in years of high rat numbers females using restricted nest locations become more vulnerable. There are no data on the habits and consistency of nest site selection by SI robin females or whether they select nest sites depending on predation rates or environmental factors such as aspect or major weather patterns such as El Niño.

Interestingly stoats took SI robin nests exclusively by day and at the chick stage, even though they have been found to hunt and take bird nests by night for other species like mohua (*Mohoua ochrocephala*), rock wren (*Xenicus gilviventris*), kaka and titipounamu (Little et al., 2017), all cavity nesters. SI robins use cup nests. It may not be worth the effort for a stoat to hunt for nests by night when the reward is relatively low and open cup nests easier to locate during the day due to loud chicks. A recent study on rock wren (*Salpinctes obsoletus*) nest predations found that stoats investigated nests by day before returning by nest to kill the nest content when the brooding parent was present (Little et al., 2017). Our observations never captured a stoat or a rat observing a nest without destroying at least part of the nest content. We did however, on multiple occasions observe a possum 'miss' a nest while ascending or descending a tree. The difference between SI robins and rock wren is the nest type. Rock wren use cavities with only one small entrance trapping the brooding adult, whereas SI robins have open cup nests where escape of the brooding female is possible.

Re-nesting ability of South Island robins

NI robins in Pureora forest were reported to take about two weeks to re-nest (Powlesland et al., 2000). Also, the number of clutches a robin is able to raise successfully has been reported as varying between 1 - 3 for NI robins (Armstrong et al., 2000, Powlesland et al., 2000) and 1 - 4 for SI robins (Heather and Robinson, 2015).

We found that SI robins in Tennyson Inlet on average took 8.5 days to re-nest (N = 109 nests). Females took 8.4 days longer to re-nest after a successful nest outcome (15.3 days), rather than after a nest failure (6.9 days). We also observed a significant difference in the re-nest time interval between nests with multiple fledglings compared to fledging just one chick. This appears to reflect the work allocation between the male and female post fledging.

Males will always look after a fledgling but females tend to only care for fledglings if there is more than one or they are the sole parent. In the case of three fledglings the male tends to look after two and the female just a single fledgling. Our data are not complete and with the absence of post fledging survival rates, understanding the time demands on either parent past fledging is unreliably discussed here. We have no way of knowing if the time taken to re-nest is dependent on the number of fledglings raised to independence rather than just the number of fledglings fledged, which is the figure reported and discussed here.

In our study, there is a difference between rats and stoats in regards to the nest age at which these predators attacked nests; stoats took chicks and rats eggs. This raises the question of how damaging losing a clutch of eggs is compared to chicks and if, the time needed by females to prepare for a subsequent nest is similar no matter when, the previous nest failed. We found that the number of days a previous nest had been active for was negatively correlated with the time it took for a female to re-nest, extending the re-nesting time interval by 0.42 (+/- 0.16 SE) days per active nest day. If this is true, then it has a direct impact on pest control management of different predators. In this case, the year after a beech mast usually has increased stoat and reduced rodent pressures. Therefore, a robin nest is more likely to be attacked at chick stage after a longer time investment by the robin pairs. From our data the impact of losing nests at the chick stage is compounding the impact by increasing the inter-nest attempt time periods, theoretically resulting in fewer nest attempts per breeding season not just due to nests failing later but also as the re-nest interval is longer.

In regards to pest control efforts often occurring part way through a SI robin breeding season, we found that the re-nesting time interval was decreased by 3.7 days in the later part of the breeding season than in the earlier part. This is valuable information as aerial 1080 pest control efforts, especially in nation-wide beech-mast years, are often delayed from the start to the middle of the SI robin breeding season.

Productivity of South Island robins

Productivity differences in previous reports are generally linked to the number of predators at the site, showing that the higher the predator numbers the lower the productivity (Dimond and Armstrong, 2007, Powlesland et al., 2000). We are unable to report on productivity but can report on the number of fledglings produced per successful nest.

We observed small differences between years for the number of fledglings produced per nest attempt and between the treatment and non-treatment areas. We found an increase in the number of fledglings produced per nest attempt in the latter part of the breeding season in all years and both areas irrelevant of pest control. The sole exception being the non-treatment area in 2014, here the lack of rise in fledglings produced in the latter part of the breeding season corresponds with high rat numbers due to the beech-mast.

The pest control effort alone did not appear to elevate the number of fledglings produced, rather the lack of pest control during a mast year depressing fledgling production in the non-treatment area. This was only observed in the 2014/15 season when the pest control effort and the beech-mast affected predator numbers. The pest control effort in a non-mast year (2013) had no such effect on the number of fledglings produced. We therefore suggest that food availability or competition is possibly the primary driver of the number of fledglings produced per nest, while predation pressures are likely the key driver for the number of successful nests per female.

If our observations are correct future productivity studies should consider including a measure of insect availability to provide a greater understanding of the direct and indirect effect of beech masts and pest control on SI robin productivity. We suggest that there may be a negative impact on the number of chicks that survive to fledging age due to high rat numbers competing for food resources with SI robins in the later part of the breeding season. If this is true, nest success as a sole measure to evaluate the impact of high predator numbers is unreliable, especially in the breeding season following a beech mast.

Unusual visitors at South Island robin nests

With the use of cameras, two possibly new predators have been identified for SI robins, the long-tailed cuckoo and the house mouse. Mice are mentioned as possible predators in Heather and Robinson (2015). No mention of long-tailed cuckoos as SI robin nest predators was found.

4.6 Conclusion

Using cameras has aided the understanding of predation on SI robin nests. Cameras allowed the identification of predator types and the proportion of natural nest failures to be differentiated from predator-caused nest failures. They also provided observations on hunting behaviours of stoats, possums and rats, revealing that rats were the only predator to cause adult mortality during a nest attack. Interesting footage was captured of a long-tailed cuckoo and possibly a mouse taking eggs from a SI robin nest as well as the use of feathers from failed nests by titipounamu.

We observed 4.35% of breeding females being lost due to rat nest predations. However, we found higher productivity in the later part of the breeding season with more fledglings produced per nest attempt and shorter intervals between nest attempts. Although the production of more fledglings in the latter part of the breeding season is beneficial to the population, this benefit may be reduced by low fledgling survival rates. Both Drummond et al. (2019) and Dimond and Armstrong (2007) found a higher mortality rate in late season fledglings, then for early season fledglings.

With 1080 operations often delayed into November, half the SI robin breeding season is already behind them. In Tennyson Inlet, no new nest attempts occurred after 12 January. For another nest attempt to be successfully completed before the 12th, about 58 days are required, including 42 days for the nesting period (from laying to fledging) and 16 days to re-nest (average of 15.3 days re-nesting interval after a fledging event). Therefore, the latest possible date at which a pest control effort could allow two successful SI robin breeding attempts would be ~15 November, assuming that the pair is ready to re-nest at the time of the pest control operation.

Chapter Five

Estimated nest success and predation rates on South Island robin nests in the Marlborough Sounds

5.1 Abstract

New Zealand is internationally renowned for the impact that introduced predators have had on its vulnerable biodiversity. Rodents, stoats and possums are recognised as the trio largely responsible for the population declines observed in birds over the last century.

We assess the effectiveness of the most widely used aerially applied 1080 regime used in New Zealand. The 1kg/ha 6g cereal pellets (RS5) laced with 0.15% of 1080 are used to protect native wildlife from introduced predators, here we assess the benefits to South Island robin (*Petroica australis*). The study was carried out in a 39ha native forest study area at Tennyson Inlet in the Marlborough Sounds, receiving two 1080 applications during the study (2013 and 2014). SI robin nest success was monitored over five breeding seasons (2012/13 - 2016/17), to provide insight into predator-prey relationships and predator control efficiency.

No negative impacts were observed on daily survival rates of SI robin nests during the two 1080 applications. Aerially applied 1080 pest control increased the survival rates of SI robin nests from 33 to 77% in 2013 and 8 to 54% in 2014. Rodent numbers quickly rose after the beech-mast triggered 1080 application, reaching 31% one year and 43% tracking rate two years later. This was reflected in the Estimated Nest Success (ENS) dropping to 38% one year and 4% two years after the pest control effort.

5.2 Introduction

Please refer to 'General Introduction' (**Chapter 1**) for information on study area, conservation, 1080 pest control in New Zealand and South Island robins.

Nest success

Measuring nest success is widely used as an assessment of avian population health, even though it only looks at one aspect of population health and often only over a short time period (Lettink and Armstrong, 2003). Many studies report apparent nest success: the number of successful nests divided by the total number of nests found. This method overestimates nest success as it fails to account for nests that fail before they are found. This method is unable to account for variation in nesting success during a breeding season or at different stages within the breeding cycle. This becomes more important for species that have long breeding seasons such as SI robins (September to February) or are affected by changes in predator pressures, such as during beech mast events. (Armstrong et al., 2002).

The Mayfield method (Mayfield, 1975, Mayfield, 1961) was devised to eliminate the overestimation of nesting success inherent in apparent nest success. It estimates a daily survival rate (DSR) based on probability of nests surviving during each day they are observed. Estimated nest success (ENS) is calculated by raising DSR to the power of nest days.

DSR & ENS based on the Mayfield method are comparable between years, areas, studies and species (Lettink and Armstrong, 2003), though there are inherent biases, assuming that nests fail half way between when they were last seen active and when they were first found failed. A method developed by Johnson (1979) incorporates uncertainty about fail or fledge dates into the estimate and removes this source of bias. Even more sophisticated methods of estimating nest success using program Mark (Dinsmore et al. 2002) or generalized linear modelling (GLM) software overcome the biases of the apparent nest success and the Mayfield method, and also enable easy inclusion of explanatory variables in models of nest success (Johnson 2004). I used the GLM approach to estimate DSR and ENS.

DSR of nests are unlikely to be constant from egg laying to fledging as changes in smell and noise levels change as chicks age, affecting their detectability by predators. Similarly, the DSR of nests is unlikely to be constant between the start of the breeding season (September) and the end of the breeding season (February) or constant between years. I accounted for these variabilities by adding them as parameters to our GLM models.

The main question concerned the effectiveness of pest control. However, the study was conducted over a limited number of years (five). This is a short time period without replication where results may be influenced by factors other than the effectiveness of pest control. In particular, season, weather, and food abundance may mask the effect of the pest control. For this reason, both ENS and estimated predation rates (EPR) were estimated and compared. EPR excludes nest failures caused by natural events such as storms, right-censoring the data of naturally failed nests to the day before their observed fail date. This allows comparison of nest success between years, focusing only on nest failures caused by introduced mammalian predators targeted by the pest control effort.

5.2 Method

Monitoring nests

Nest success of SI robin was estimated from nests monitored over five breeding seasons (2012/13 - 2016/17) for nests where the both father and mother were banded. Nests where either or both parents disappeared before a band could be applied were omitted from analysis. For a nest to be included in nest success analysis, the nest had to be identified as 'active'. A nest was deemed 'active' only if the nest contained eggs or chicks, or if prolonged observations of incubation or chick feeding behaviour was observed by either a camera or an observer. Nests that were found after failure had occurred, were omitted from analysis.

All nests were if possible monitored by camera and by weekly to fortnightly checks. Nest failures were attributed to predators only when there were photos of an identifiable predator. Predation events were not inferred from examination of nest contents. No attempt was made to distinguish ship (*R. rattus*) and Norway rats (*R. norvegicus*). Nest outcomes were either identified with a predator name, 'fledged', 'unknown predator', 'abandoned' or 'human caused failure'.

When chicks or eggs disappeared and only partial or no images captured of the predation event, the nest outcome was classified as 'unknown predator'. Nests that were observed to fail in the absence of a predator were classified as 'abandoned'. A nest was classified as 'human caused failure' if the nest failed immediately after setting up a camera or a nest check event and the nest content remained the same.

Predation events caused by introduced predators could be separated from failures caused by native predators and abandonments from photos. This allowed us to analyse not just the Estimated Nest Success (ENS) but also the Estimated Predation Rate (EPR). As the aim of this study is to assess the efficiency of removing introduced predators through aerially applied 1080, changes in EPR is a more valuable statistic than ENS which also includes natural nest failure events.

ENS of SI robin was used to compare the effectiveness of the aerial 1080 pest control application in the treatment area and non-treatment area. Additional comparisons were made between the 1080 application in 2013 (in the absence of a beech mast event) and the 1080 application in 2014 (in response to a beech mast event). The comparisons were made by including and excluding appropriate parameters from our GLM models.

The nests we observed generally had 22 chick days, 18 incubation days and a laying period of three days including the days on which the first egg is laid. For calculating ENS from DSR we assumed that robins spent 42 days nesting, not 38 as was used by Powlesland (1983), who omitted the initial egg-laying period. Compared to other studies our ENS results will be lower due to the longer nest period used.

The nesting period of 42 days was divided into and early and late stages. This was to account for changes in nest detectability as chicks get bigger noisier and eat/defecate more. We used the 25th nesting day (5-day old chicks) as the cut off between early and late stage nests, as chicks become more active around that time.

Nest survival analysis assumes that daily survival rates are homogenous, that all nest ages and nest fates are correctly estimated and that finding and checking nests does not affect nest fate. We could not always accurately determine nest fate from observations or camera footage and we right-censored some nest records to account for this. When the outcome of a nest was unknown, we right-censored the data by ignoring nest observation days after the last visit, when it was clear that the nest was still in use.

Based on previous research, nest success is negatively affected by introduced mammalian predators (rats, stoats and possums), and predator numbers are affected by time between treatments and beech-mast events (Wright, 2011, Armstrong et al., 2002, Etheridge and Powlesland, 2001). In this study, treatment effects included two pest control operations and a single beech mast event which either decreased or increased predator pressures within and between breeding seasons for SI robins. To account for these effects, the study was divided into seven meaningful time periods (**Table 5.1**).

An additional time period of 35 days was added immediately after each 1080 application, making nine time periods in total. These two additional shorter time periods allowed for a minimum of 100ml of rain to have fallen after a pest control effort, accounting for the active 1080 period (**Chapter 2**).

Table 5.1: The study period of five years and two areas was divided into seven-time categories for analysis to account for before and after 1080 treatments (2013 & 2014) and beech-mast event (2014), and nine categories to assess the effect of active Sodium fluoroacetate in the forest on ENS of South Island robins.

Time periods Codes	 Description	Years	1080 application	Beech mast
1	Before 1080	2012	no	no
2	Before 1080	2013	no	
3	Active 1080		yes	flowering
4	After 1080		same yr	seeds in trees
5	Before 1080 & mast effect	2014	one yr after	seeds on ground
6	Active 1080		yes	
7	After 1080		same yr	
8	1yr after 1080	2015	one yr after	one yr after
9	2 yrs after 1080	2016	two yrs after	two yrs after

Generalised Linear Models (GLM's)

DSR was estimated using generalised linear models (glms) with binomial errors and a complementary log-log link function following the methods of Rotella et al. (2004) and Bolker (2014). Potential explanatory variables included in the models were:

Site	- either treatment or non-treatment
Season	- the early or late part of the breeding season
Age	- nest age in days from 1 - 42
Age^2	- nest age in days from 1 -42 squared
Time:area	- allowed to account for the year and the treatment effect
Rodent tracking	- tracking rates from Tracking Tunnel results from above 400m were used and values were both extrapolated and interpolated between observed tracking rates and aerial 1080 operations.

In addition, a random effect PairID was included in all models to account for any lack of independence in the success of nest attempts by the same pair. A suite of glms with plausible combinations of the explanatory variables were compared using AICc (Burnham and Anderson, 2002). All analysis was carried out using the package lme4 (Bates et al., 2015) in R version 3.4.2 using R studio.

5.3 Results

Summary of nest success data

A total of 200 nests were used to analyse ENS, with over twice as many nest failures (N=136) as successful nests (N=64). This included 147 nests in spring (before 14th November) and 53 in summer (after 14th November). The 200 nests contributed a total of 3529 observed active nest days, of which 96% came from the early nest stage (<23 days old) and 4% from the later nest stage when chicks were larger and noisier.

Over the five breeding seasons, 51 males contributed on average 26 nest days (range 1 - 207) and 46 females contributed on average 77 nest days (range 1 - 295). This combines to a total of 67 pairs contributing an average of 53 nest days each (range 1 - 195) (**Table 5.2**). There was no statistical difference between the number of nest days contributed by father, mother or pair, over the course of the study (**Figure 5.1**). There was also no statistical difference in the number of nest days per father, mother, pair or nest, between the treatment and non-treatment area (**Figure 5.2**). Of the 200 nests 62% (123) failed due to a predator, including 52 nests where the predator could not be identified. Only 7% (15) of nests failed due to natural causes. Natural causes included abandonments and predation events by native predators such as ruru (*Ninox novaeseelandiae*).

A total of 172 nests were used to analyse EPR with adjusted nest outcomes. This included 3230 nest days with 129 nests in spring (before 14th November) and 43 in summer (after 14th November). This difference is due to the removal of nests where the cause of the nest failure could not be attributed to an introduced predator, a native predator or a natural abandonment.

Table 5.2: Number of South Island robin fathers, mothers, pairs, nests and observed nest days used for analysis from the treatment and non-treatment areas in Tennyson Inlet over the course of the study period 2012/13 - 2016/17.

Area	Pairs	Mothers	Fathers	Nests	Observed Nest days
Treatment	35	24	23	106	1856
Non-treatment	32	22	28	94	1673
Total	67	46	51	200	3529

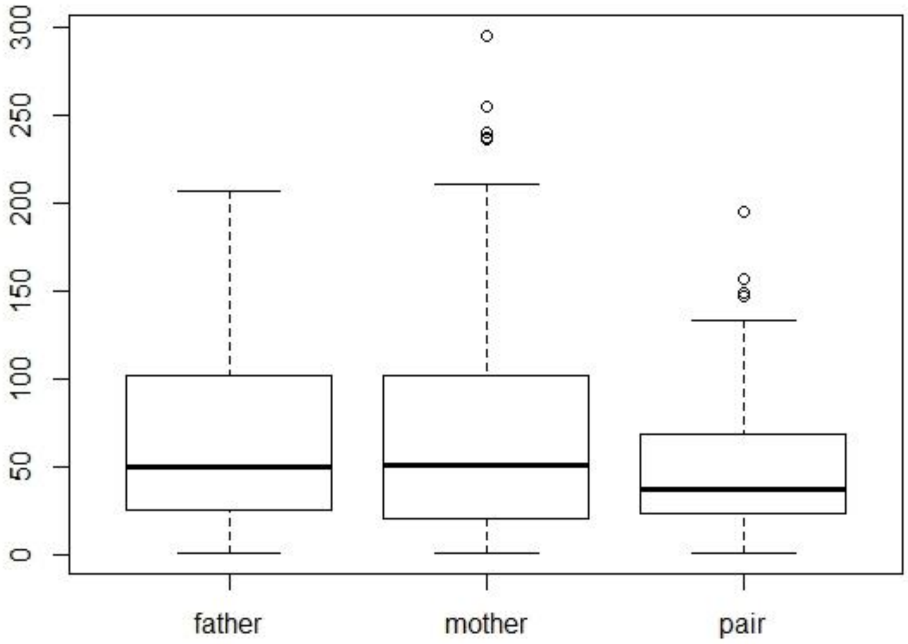


Fig. 5.1: Box and Whisker plot displaying the data range of observed nesting days per father (N=51), mother (N=46) and pair (N=67) over the study period 2012/13-2016/17.

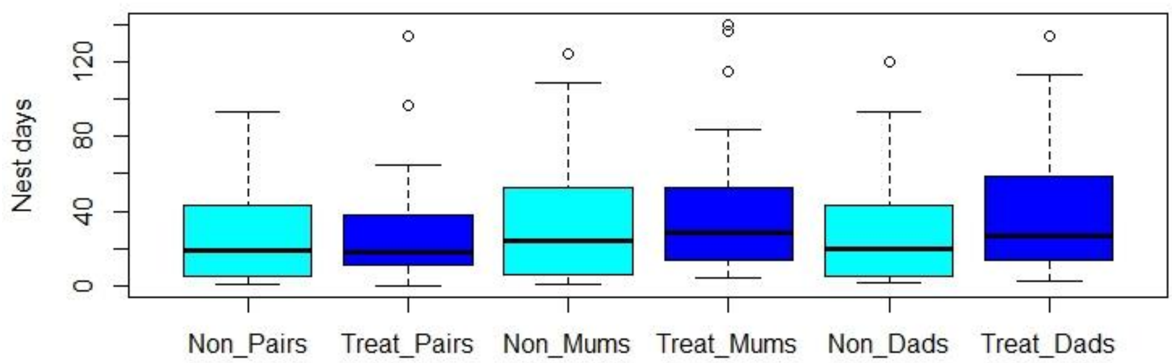


Fig. 5.2: Box and Whisker plot displaying the distribution of the number of nest days observed for mothers, fathers and the pair for each of the areas, over the course of the study period 2012/13-2016/17. The non-treatment area (Non) in light blue and the treatment area (Treat) in dark blue.

Nest failures during the active 1080 periods of the 2013 and 2014 pest control efforts

Two nests failed due to abandonment within 35 days of 1080 application. Nest RT531 failed within 24 hours of the 1080 operation when the eggs were close to, or just hatched. Both adults were found alive subsequently. Nest RT412 failed about four days after the 1080 operation while the eggs were being incubated. The female disappeared between four and five days after the 1080 operation and was never seen again. There is no direct evidence that the failures were caused by 1080, but if the robins ate any of the 1080 baits their illness or death could cause a nest to fail. A nest may also have failed due to helicopter disturbance or impact from a falling bait.

Factors affecting DSR analysis approach one: using Estimated Nest Success (ENS) data

The most parsimonious model shows a relationship between daily survival, using two explanatory variables; nest age as a linear variable and time within the breeding season as a quadratic variable (**Table 5.3**). We found a treatment effect as the time:area model fitted the data better than time+area model. The additional variables of nest age and a quadratic term for season improved the model.

The nine term time:area model was better than the seven term time:area, this indicates that nest success is different immediately after a 1080 operation (**Fig. 5.3**). However, this effect was not consistent between the two 1080 drops. Relative to the non-treatment, nest success was worse in the active period after the 2013 operation, but better in the active period after the 2014 operation. Additionally, a model that treated the 2013 treatment area nest success data as non-treatment was a better model – indicating that the 2013 treatment had a negligible effect on ENS.

Model 'seven' was therefore the most parsimonious model on which to predict ENS for SI robins (**Fig. 5.4**). The aerially applied 1080 pest control effort increased the SI robin nest survival rates including a 95% confidence limit from 34% (8 - 62) to 77% (15 - 96) in 2013 and 8% (2 - 21) to 55% (19 - 80) in 2014. In the non-treatment area nest success remained relatively stable before and after the pest control efforts of 2013 and 2014.

The increased ENS in the treatment area was short-lived dropping to 38% (18 - 58) in 2015/16 and down to 4% (0 - 15) in 2016/17 – two years after the pest control effort. In the non-treatment the ENS rose after the rodent plague observed in the 2014/15 season to 25% (9 - 45) in the 2015/16 breeding season and 26% (10 - 45%) in the 2016/17 season.

Table 5.3: Assessing the relative importance of possible explanatory variables to predict nest success, including nest age from laying to fledging and time within the breeding season over the course of the study period 2012/13-2016/17 as linear, quadratic or categorical variables. Models are listed in order of their AICc values with the favoured most parsimonious model at the top.

Factors	k	df.resid	AICc	ΔAICc	Deviance
time:area+(1|PairID)+age+season+season^2	22	3507	1022.95	0	978.66
time:area+(1|PairID)+age+season+season^2	18	3511	1027.16	4.21	990.97
time:area+(1|PairID)+age	16	3513	1028.87	5.92	996.72
time:area+(1|PairID)+season+season^2	17	3512	1029.84	6.89	995.66
time:area+(1|PairID)	15	3514	1030.32	7.37	1000.18
time:area+(1|PairID)+age+age^2	17	3512	1030.60	7.65	996.42
time:area+(1|PairID)+split-season	16	3513	1032.20	9.25	1000.05
time:area+(1|PairID)+stage	16	3513	1032.33		1000.17
time:area+(1|PairID)+season	16	3513	1032.33	9.38	1000.18
time+area+(1|PairID)+age+season+season^2	12	3517	1035.43	12.48	1011.34
(1|PairID)	2	3527	1037.64	14.69	1033.64

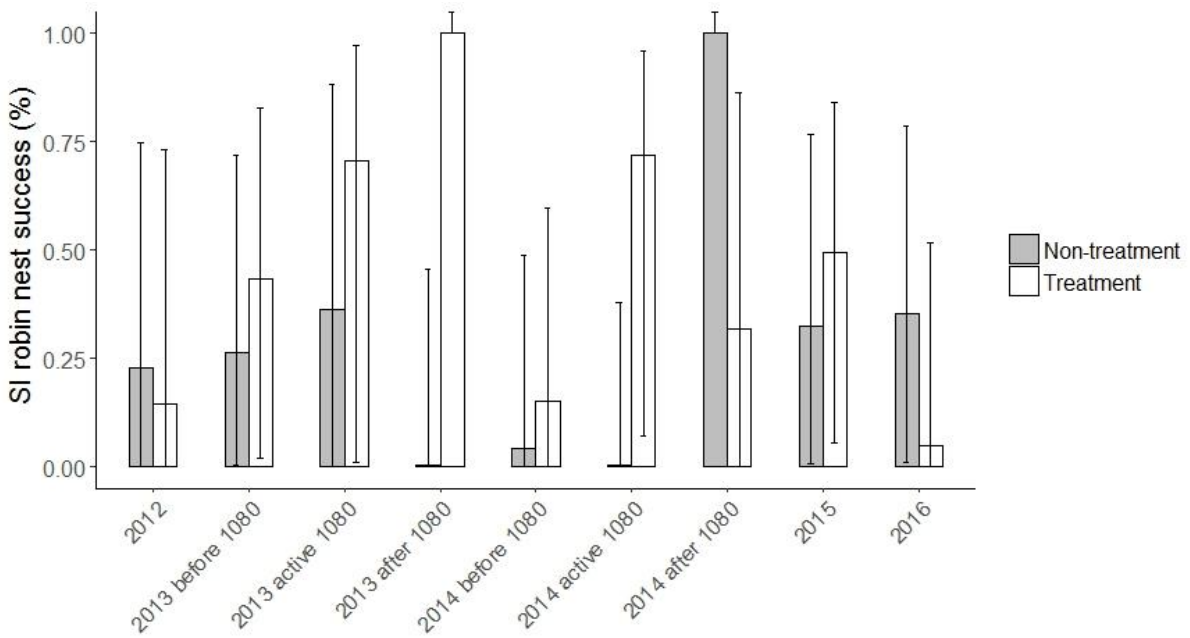


Fig. 5.3: South Island robin estimated nest success (ENS) and associated 95% confidence limits are given for each of the nine time periods, including the 35-day period immediately after a 1080 operation. A rise in ENS during the active 1080 period of 35 days in 2013 and 2014 was observed, removing the concern that SI robin ENS may have been adversely affected by the presence of Sodium fluoroacetate (1080). The pest control operations took place on the 2 and 23 November in 2013 and 2014 respectively. The ENS in the non-treatment area is low for three consecutive time periods from November 2013 to November 2014. In 2016, two years after the beech mast induced pest control operation the ENS has decreased in the treatment and is still slowly increasing in the non-treatment area.

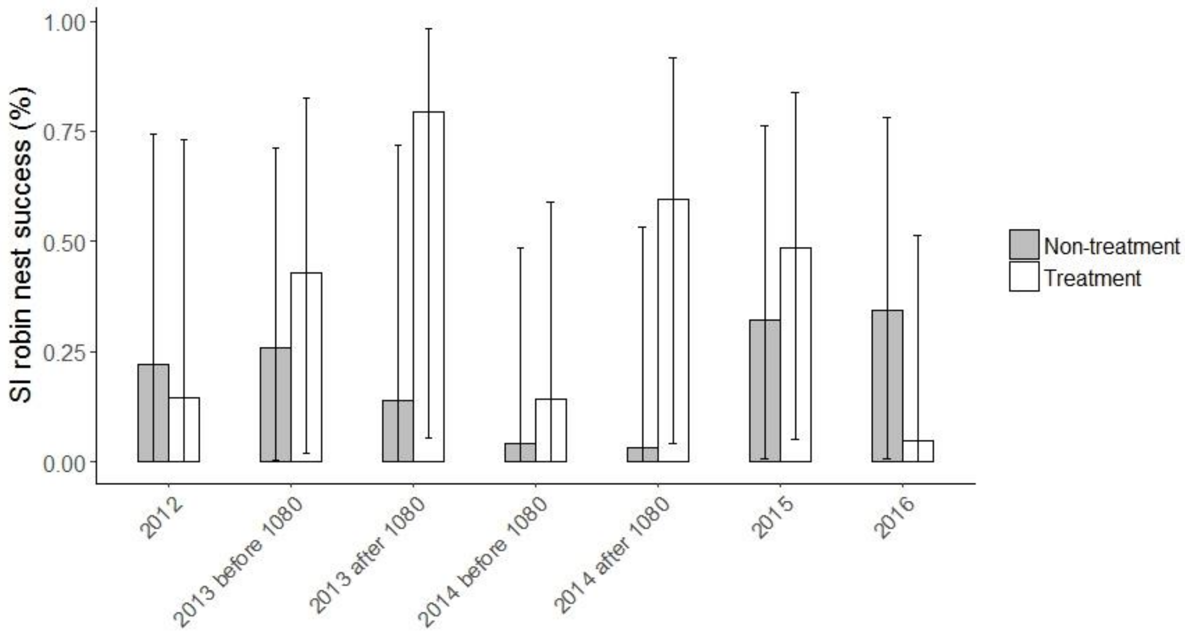


Fig. 5.4: South Island robin estimated nest success (ENS) and associated 95% confidence limits are given for each of the seven time-periods. The time categories represent different predator pressures on nests due to the time since a beech mast event in both areas and the time since the last 1080 pest control efforts in the treatment area. ENS in the non-treatment shows a decrease in ENS in response to the beech mast event in 2014 with ENS already decreasing during the beech flowering season. Comparatively the ENS in the treatment area where 1080 was applied in both 2013 and 2014 spikes after the pest control efforts and decreases in 2015 and again in 2016, one and two years after the 1080 was applied. The ENS in the treatment area before the 2014 1080 application is low, showing that the effect of the beech mast was stronger than the pest control relief provided before the beech fruiting event in 2013.

Factors affecting DSR analysis approach one: using Estimated Predation Rate (EPR) data

The same models were run again. This time the dataset consisted of only mammalian caused nest failure events (Estimated Predatory Rates or EPR), with remaining nest failures being right censored to explore whether the observed treatment effect was due to the 1080 operations or due to stochastic events such as food, disease or unusual weather events. Results were then compared to those of the ENS model.

The most parsimonious model was ‘2’ containing nest age as a linear variable. A treatment effect was observed under EPR, with the time:area model fitted the data better than time+area model, implying that the 1080 drop was the most likely cause for the increased ENS observed. The ENS in the treatment area was increased by 18% after the 2013 1080 drop and by 56% after the 2014 1080 drop. This shows that the rise observed after the two 1080 operations in ENS was not due to the treatment effect.

Table 5.4: Assessing the relative importance of possible explanatory variables to predict nest success, including nest age from laying to fledging and time within the breeding season over the course of the study period 2012/13 - 2016/17 as linear, quadratic or categorical variables. The data was right censored for natural nest failures so as to code only introduced mammalian (rodent, mustelid and possum) nest predations as failure events. This allows for an assessment of efficiency of the aerial 1080 pest control effort without stochastic noise in the dataset. Models are listed in order of their AICc values with the favoured most parsimonious model at the top.

Model	Factors	k	df.resid	AICc	ΔAICc	Deviance
1	treatment:area+(1|PairID)	15	3515	797.91	0	767.76
2	treatment:area+(1|PairID)+age	16	3212	798.33	0.42	766.16
3	treatment:area+(1|PairID)+split-season	16	3214	799.5	1.59	767.33
4	treatment:area+(1|PairID)+season	16	3214	799.83	1.92	767.66
5	treatment:area+(1|PairID)+stage	16	3214	799.91	2.00	767.74
6	treatment:area+(1|PairID)+age+season	17	3513	800.11	2.20	765.92
7	treatment:area+(1|PairID)+age+split-season	17	3513	800.16	2.25	765.97
8	treatment:area+(1|PairID)+age+age^2	17	3213	800.32	2.41	766.13
9	treatment:area+(1|PairID)+season+season^2	17	3213	800.4	2.49	766.21
10	treatment+area+(1|PairID)	9	3221	807	9.09	788.94
Null	(1|PairID)	2	3228	813.87	15.96	809.87

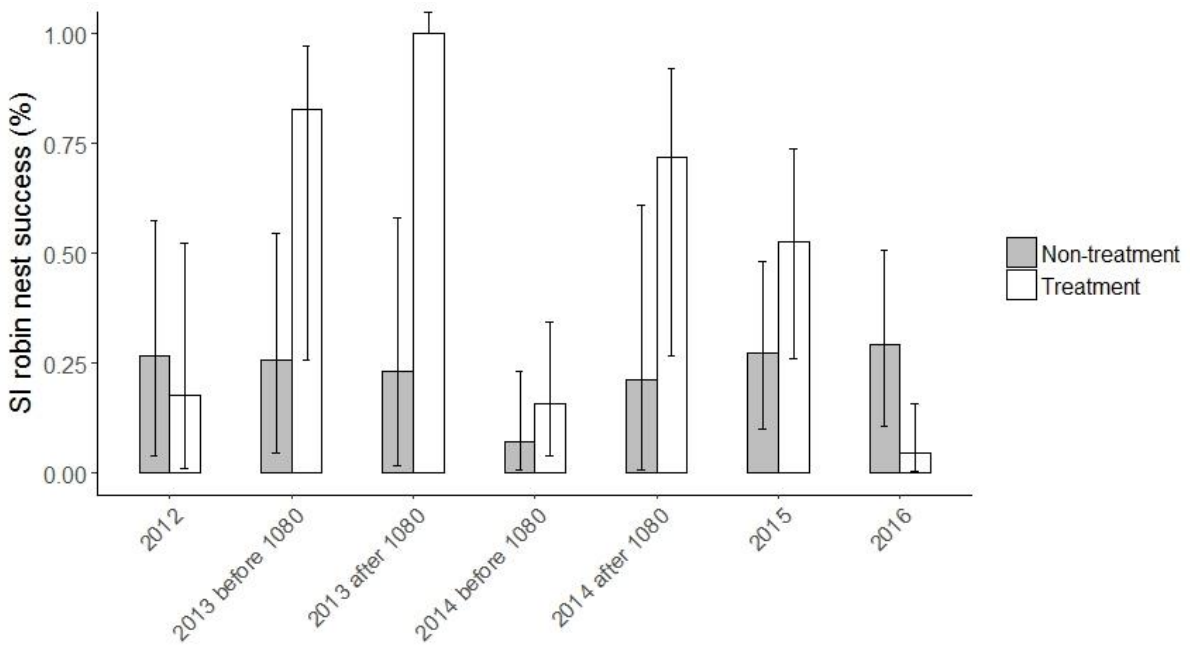


Figure 5.5: South Island robin estimated nest success (ENS) under a modified dataset containing only mammalian caused nest failure events (estimated predation rate, EPR) and associated 95% confidence limits are given for each time period. The time categories represent different predator pressures on SI robin nests due to the time since a beech mast event in both areas and the time since the last 1080 pest control efforts in the treatment area. Dataset modifications consisted of failure events caused by natural predators, abandonments and due to weather events having been right censored. The ENS in the treatment area where 1080 was applied in both 2013 and 2014 spikes after the pest control efforts and decreases in 2015 and again 2016, one and two years after 1080 had been applied. The ENS in the treatment area before the 2014 1080 application is low, showing that the effect of the beech mast was stronger than the pest control relief provided before the beech fruiting event in 2013.

Factors affecting DSR analysis approach two: using rat rates to predict ENS

When modelling ENS, time can be treated as a continuous variable. As we are particularly interested in the changes of predatory threat, we used time as a categorical variable. By breaking time into categories with similar levels of predators depending on time since a beech mast and/or predator control effort and separated the treatment and non-treatment areas. This is reasonably complicated, as there are so many time categories (seven). A simpler approach is to replace time directly with a measure of predator pressures. Mustelid, rat and possum tracking rates are our best estimations on the severity of introduced predator levels and ideally, we would use a combination of all three predator rates on which to model SI robin ENS. In our case, only data from rat tracking rates was reliably available for the duration of the study period. Rodent tracking rates should account for the beech mast event, time since the beech mast event, as rodent numbers respond to food availability, and account for the 1080 pest control operations. By using rat rates, we can simplify the models from seven-time categories on which to predict SI robin ENS to just the single variable accounting for rat tracking rates.

The categorical variables for nest age (stage) and time within the breeding season (split-season) were valuable variables to include to in the models to predict nest success (**Table 5.5**). The combination of the two variables within a single model did not improve model fit, rendering a model containing only nest stage as most parsimonious.

Table 5.5: Assessing the relative importance of possible explanatory variables to predict nest success based on rat rates. Explanatory variables assessed include, nest age from laying to fledging and time within the breeding season over the course of the study period 2012/13 - 2016/17 as linear, quadratic or categorical variables. Models are listed in order of their AICc values with the favoured most parsimonious model at the top.

Model	Factors	k	df.resid	AICc	ΔAICc	Deviance
1	ratrate+(1|PairID)+stage	5	3495	886.22	0	867.21
2	ratrate+(1|PairID)+split-season	5	3495	886.60	0.38	876.59
3	ratrate+(1|PairID)+stage+split-season	7	3493	889.10	2.88	875.07
Null	(1|PairID)	2	3497	955.93	69.71	1041.49
4	ratrate +(1|PairID)+age	3	3497	1033.55	147.33	1027.54
5	ratrate +(1|PairID)+age+age^2	4	3496	1035.42	149.20	1027.41
6	ratrate +(1|PairID)+season+season^2	4	3496	1041.36	155.14	1033.35
7	ratrate +(1|PairID)+season	3	3497	1044.30	158.08	1038.29
8	ratrate+(1|PairID)	3	3514	1047.50	161.27	951.93

Summary of results

The additional variables of nest age and time within the breeding season differed in their importance depending on the dataset and analysis approach used. When ENS data was used, linear nest age and binomial time within the breeding season were most important, while under EPR neither nest age nor time within the breeding season significantly improved model fit. When using the rat tracking rates instead of time categories only nest age split into early and late failures was important for predicting ENS. There were complications with models failing to converge even after scaling the variables nest age and time within the breeding season when using rat tracking rates to predict SI robin nest DSR.

5.4 Discussion

The use of aerial 1080 targeting rats to protect native wildlife is a new pest control approach and its efficacy is still poorly known. We hoped to test if a single aerial 1080 application at the most frequently used sowing rates of 1kg/ha in response to a beech mast would reduce rodent numbers enough to increase nest success of SI robins.

Both 1080 operations increased daily survival rates of SI robin nests (44 & 46%). Although 1080 treatment increased nest success, the benefit of the 1080 operations was limited by their timing. Both 1080 applications were conducted very late in the breeding season (02/11/2013 and 23/11/2014). This provided around two months in which SI robins were able to re-nest and raise chicks in a relatively predator-free environment. This is a very short time. The last observed nest started laying on the 12 January, meaning that in most cases only a single successful nest attempt was possible after the 1080 operation occured. Furthermore, we monitored relatively few nests after the 1080 operations (129 nests < 14 November < 43 nests) and this has reduced the inferential power of our findings.

The observed improvement in nest success was short-lived. Only one year after the 1080 application the ENS had reduced by 17% and two years after the operation to 4% a 51% drop in ENS from the 2014 treatment year. Other areas have had similar fast rodent recoveries following 1080 operations (Elliott and Kemp, 2016) though in some areas, rodent suppression persists for much longer. Possible explanations for variability in rodent recovery rates include food abundance, the size of the treatment area, exclusion zones, and the low 1080 sowing rate.

Death or illness from ingestion of 1080 is a concern for native species management. Although we observed two nest failures that may have been caused by 1080, there was no detectable negative impact on DSR for SI robin nests during the 2013 or 2014 operation. nineteen native bird species have been recorded being killed by 1080 including 12% (N=17) NI robins during an operation in Pureora Forest Park (Veltman and Westbrooke, 2010, Empson and Miskelly, 1999). However cereal baits have replaced carrots in most 1080 operations and the pellets have been modified to decrease fragmentation when they hit branches or the forest floor (Wright, 2011). Sowing rates of > 15kg/ha were used during the 1980's, while todays sowing rates are mostly 1 - 2kg/ha (Wright, 2011). Our study and other studies (Tinnemans et al., 2018, Jones, 2016, Schadewinkel et al., 2014, Powlesland et al., 2003, Westbrooke et al., 2003) found that low sowing rates of 1 - 3kg/ha had a low risk to non-target species such as SI robins, reporting no or few individuals perishing due to the pest control operation.

The use of cameras for nest observations meant that the perpetrator was reliably identified in 83% of predation events. This allowed us to estimate the predation rate (EPR) and compare this to the ENS. We found that the ENS and the EPR both showed an increase in nest survivals after the pest control effort for both 1080 operations. The EPR showed a more reliable though less pronounced increase in the improvement to the daily survival rates of SI robin nests than ENS. This comparison provided us with confidence that the increased ENS observed after the 1080 operation was due to the removal of mammalian predators and not due to chance.

All analyses of ENS and EPR used a period of 42 nest days rather than the 38 days used by Powlesland (1983). We suggest that the number of days chicks take to fledge is associated with food availability. Previous studies of SI robin nest success were mostly at lower altitudes where the climate is less severe and insects likely more abundant. We suspect that the extended nestling period we observed as typical of areas at higher altitudes. Our estimates of nest success will be comparatively lower than those observed by other studies using a shorter nestling period.

Rat tracking rates was a good predictor of nest success, even though rats only caused 32% of the nest failures. Possums and stoats caused an additional 19% of nest failures. This is not a surprising finding. Although rats were not the only cause of nest loss, their abundance is almost certainly correlated with the abundance of stoats and, so rat abundance is likely also a good predictor of nest failure rates caused by both predators combined.

Nest success monitoring is very labour intensive, and many areas of New Zealand are remote and expensive locations in which to conduct species monitoring projects. By using rodent tracking rates, a rough estimate of SI robin nest success over larger areas of New Zealand could be extrapolated.

Overall, no adverse effects were observed on SI robin nest survival from aerially distributed 1080. SI robin nest survival increased after each 1080 application, with the application following the beech mast in 2014 more effective in the short term. However, this did not continue to lift nest success in following breeding seasons. It is possible that one 1080 operation every beech mast will be sufficient to remove nest success loss, due to beech mast events, but alternatively they may require more frequent predator-control to combat fast recovery rates of rodents in small treatment areas. Further research is required, to determine the frequency of aerially applied 1080, necessary for maintenance of robin population health.

Chapter Six

Adult Survival of South Island robins in Tennyson Inlet, Marlborough Sounds

6.1 Abstract

Pest control efforts are a vital component for the long-term survival of native biodiversity in New Zealand. Improvements are constantly being made to pest control techniques and application methods. This study aimed to assess the effectiveness of aerially applied 1kg/ha 6g cereal pellets (RS5) laced with 0.15% of sodium fluoroacetate (1080) in regards to adult South Island robin (*Petroica australis*) survival rates. Survival rates were monitored over five successive breeding seasons (2012/13 - 2016/17), through a beech mast (2014) and two aerial 1080 applications (2013 & 2014).

We found evidence that adult SI robin survival rates can be predicted by rat tracking rates, with a quadratic function being a better predictor than a linear function. During the active sodium fluoroacetate periods, two birds went missing but this appeared to be consistent with the expected survival rates for that time of year. No models containing explanatory variables associated with the timing of 1080 operation or the active phase immediately after a 1080 drop, performed better than a model containing only an intercept. Robin sex and distinguishing between the breeding and non-breeding season did not improve the model fit.

We found no evidence to imply that sowing rates of 1kg/ha have a negative effect on SI robins. Contrary, our models found a negative relationship between rat tracking rates and adult survival irrelevant of sex. Given that aerially distributed 1080 pellets are effective at suppressing rats (**Chapter 4**), our data support that the distribution of aerial applied 1080 can improve the survival rates of adult SI robins.

6.2 Introduction

Please refer to 'General Introduction' (**Chapter 1**) for information on study site, conservation and 1080 pest control in New Zealand.

Adult and juvenile robin survival rates

Survival rates for New Zealand robins (*Petroica australis* and *Petroica longipes*) are often based on short-term studies, with factors such as density, inbreeding, predator abundance and seasonality being investigated. Most of the data on survival rates come from North

Island robins (NI robin *P. longipes*) in association with translocations (Armstrong et al., 2006a, Armstrong et al., 2006b, Armstrong and Ewen, 2002). There is a distinct lack of information about New Zealand robin survival rates during beech mast years, or long-term studies investigating how survival rates respond to predator fluctuations over several years and after pest control operations.

New Zealand robins are relatively long-lived with some of the oldest records for NI and SI robins being 16 - 17 year (*ref Heather and Robinson field guide, NZ banding records and BioWeb database).* Annual adult survival rates between 0.39 and 0.81 (Armstrong and Ewen, 2002, Drummond, 2017, Powlesland, 1983) have been recorded which suggests that the average life expectancy is much lower than 16 - 17 years. These studies identify a range of factors that might affect survival including male rivalry, population density, predator abundance and stresses associated with translocations.

Juvenile New Zealand robins have been found to have a lower survival rate than adults in their first summer season, though by their first winter juvenile survival rates are similar to that of adults (Drummond, 2017, Powlesland, 1983, Flack, 1979 & 1976). Jamieson et al. (2007) found variations in juvenile survival rates depending on how inbred the individuals were. Offspring of sibling pairings had the lowest juvenile survival rates. Also on Tiritiri Matangi Island, Dimond and Armstrong (2007) found that juvenile survival rates depended on density. In comparison (Drummond, 2017) reported juvenile survival rates varying between years, irrelevant of density.

Annual adult survival rates are similar between males and females in areas where pest control occurs (Armstrong et al., 2002, Drummond, 2017, Powlesland, 1983). However, both sexes may have higher survival rates in winter (non-breeding season), than in summer (breeding season) (Powlesland, 1983, Flack, 1979 & 1976). Powlesland (1983) did observe different survival rates between the sexes, though only during the winter, with females having the higher survival rate. This may be an indication that predation may not be the only driver of SI robin winter survival rates.

SI robin survival rates in the Tennyson Inlet are likely to be influenced by a range of factors including predator abundance and seasonality. Inbreeding, is unlikely to be a factor affecting survival, as although the Tennyson robin populations are discrete, robins can disperse through continuous forest to connect with other small populations. No assessments have been made on the inbreeding of SI or NI robin population pockets found within a large continuous forest ecosystem, as found in the Marlborough Sounds.

Survival analysis using mark and recapture methods

Survival is most often estimated using mark-recapture methods involving repeated surveys of marked individuals. Frequently, survival is estimated using program MARK, a free mark-recapture statistical modelling program designed to cope with modelling survival rates for both open and closed populations, when death has not been observed (White and Burnham, 1999). The use of MARK is an effective alternative when body counts or radio tracking individuals is impractical (Davidson and Armstrong, 2002, Armstrong and Ewen, 2001, Armstrong et al., 2001).

Using a variety of competing models R-MARK can estimate differential survival and detectability for groups (such as sexes) and periods of time. And the use of mark-recapture analysis allows for mortality estimations during the 1080 operations (November 2013 and November 2014) as a quantitative measure of by-kill. The model containing the smallest Akaike information criterion (AIC) with a correction for small datasets (AICc) can be used to select the most parsimonious model among competing models.

Underlying assumptions for mark-recapture data include (Lettink and Armstrong, 2003, Cooch and White, 2018):

1) Every marked animal present in the population at the time (i) has the same probability of recapture (p_i).
2) Every marked animal in the population immediately after time (i) has the same probability of surviving to time (i + 1).
3) Marks are not lost.
4) All samples are instantaneous, relative to the interval between occasion (i) and (i +1), and each release is made immediately after the sample.

6.3 Methods

Please refer to 'General Methods' (**Chapter 2**) for information on study area, methodology on aerial 1080 applications and active 1080 periods post application as well as general data collection methodologies.

Data collection for estimating adult survival

Robins were captured, individually colour-banded, and re-sighted at every opportunity during the study. During the spring and summer this happened almost continuously. The start for spring-summer field work varied between August and September, and finished between December and February. Sightings included birds on nests as well as birds at large in our study area. Breeding adults were checked every 7 - 10 days and searched for if not immediately found.

Non-breeding birds such as single males were only checked on a monthly to bi-monthly basis, to see if their relationship status had changed. Systematic searches for all banded robin were only conducted at the start of each breeding season and pre/post 1080 treatment applications (treatment area only). Surveys of the whole study site (treatment and non-treatment) were also conducted on a monthly basis during winter 2016 and during a final pre-breeding survey in August 2017.

Rat and stoat tracking data

Both rat and stoat tracking data were collected from tracking tunnels run routinely as described in **chapter 3**. Tracking data from tracking tunnels above 500 m altitude in the treatment and non-treatment areas were used to provide an indicator of predator abundance. We assume that higher tracking rates will result in higher predation rates on robins.

Rat tracking rates vary not only with population density, but are confounded by territory size, age, sex and health of the individual as well as food availability. We were not able to account for these confounding variables. Tracking tunnels were run four times a year, and we linearly interpolated between two surveys to obtain tracking rates between quarterly surveys.

Mark-recapture analysis

R-MARK was used to estimate survival rates and explore the effect of sex, season, mast driven rodent and stoat plagues and 1080 operations on SI robin survival. Birds were considered to join the population when banded or when birds banded as juveniles became adults. For the purpose of analysis and in attempt to standardise survey efforts, captures were analysed at monthly intervals over the study period, meaning 37 monthly blocks were created.

Plausible explanatory variables investigated included variables for sex, season, rat and stoat abundance in the form of tracking rates. Variables associated with the timing and the active periods immediately after pest control operations were also investigated to assess possible effects of the pest control effect on SI robin survival rates up to two years after the pest control effort and during the by-kill period of 35 days immediately after the pest control effort. A 35-day period was based on precipitation records available for the area (see **chapter 1**).

The variables were included or excluded from the models to judge their relative importance at estimating adult survival (**Table 6.1**). To test the possible direct and indirect effect of 1080 – the possibility that birds were killed by 1080, we classified the two months surrounding the 1080 drops as “active” (November and December 2013 and 2014 in the treatment area only).

Overdispersion in the data was examined using program RELEASE within R-MARK which produces an estimate of overdispersion, c-hat. Competing models were compared using QAICc (Quasi Akaike’s Information Criterion) – a modification of AICc that incorporates c-hat and is used to account for overdispersion (Burnham and Anderson, 2002).

Table 6.1: Explanatory variables explored in the mark-recapture analysis. These account for the treatment effect (aerial 1080 pest control), time before and after pest control and beech mast and the sex of South Island robins. Additionally, rat and stoat tracking rates were explored as explanatory variables of SI adult survival rates and an additional variable was added to account for the month-long period after each 1080 operation in which SI robin survival rates may have been affected by sodium fluoroacetate (1080) poisoning.

Explanatory variables	Description
rat	Rat tracking rate in tunnels above 500 m altitude
Rat^2	Rat tracking squared for constructing curved quadratic relationships between survival and rat abundance
Sex	Male or female
Since1080	Time (in months) since last 1080 drop
Since1080^2	Time (in months) since last 1080 drop squared for constructing curved quadratic relationships between survival and time since most recent pest control effort
Sincecat	Time since 1080 divided into 4 classes (0-2 months, 2-6 months, 6-12 months, > 12 months)
Site	Treatment or non-treatment area
Stoat	Stoat tracking rate
Stoat^2	Stoat tracking rate squared for constructing curved quadratic relationships between survival and stoat abundance
Active	The month after a 1080 drop
Time	37 monthly time periods

6.4 Results

Descriptive statistics on banded adults in the monitored population

During the study 134 individual robins were observed between the two study areas, 67 in each area (non-treatment and treatment area). Males made up the majority of banded adults, with 80 males and 54 females (1.5:1 ratio). The non-treatment area had a higher number of bachelor males than the treatment area with a sex ratio of 1.8:1 in the non-treatment area and 1.3:1 in the treatment area.

In each survey period un-banded birds were banded and added to the monitored population (**Fig. 6.1**). This ranged from zero to 14 individuals added per monthly period between the area. Progressively fewer birds were added to the monitored population. The exception was the 2016/17 breeding season when additional birds were added to increase the monitored population in preparation for a third 1080 operation. Funding was subsequently not secured, and no further pest control operations were conducted in the treatment area.

The number of birds added in each breeding season reflects immigration, productivity of the previous year and juvenile winter survival rates. To what extent the observed 'new birds' reflect the mortality rate of the previous year's adults or the nest success and survival rate of the previous year's fledglings, remains unknown and is not analysed here, as insufficient data was recorded on juveniles and age of recruited individuals.

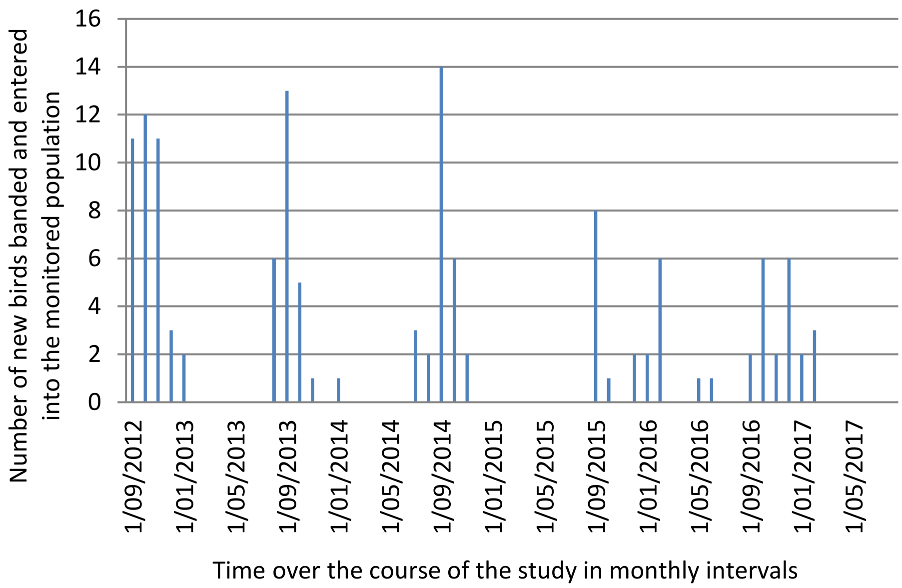


Fig. 6.1: Number of adult South Island robins banded and added to the monitored population on Mt Stanley (treatment) and Lookout Point (non-treatment) study areas.

South Island robins that went missing during 'active' 1080 periods

Sodium fluoroacetate is poisonous to SI robins if directly ingested or if enough of the non-metabolised compound is ingested through a high insect diet. We accounted for a possible by-kill period of 35 days after which enough rain had fallen to wash remaining Sodium fluoroacetate from bait pellets (**see chapter one**).

The 2013 1080 pest control operation had a maximum possible active period of 35 days, from the 2 November 2013 to 7 December. In which two SI robins disappeared, one male (M/BR) and one female (M/OK). M/BR was last seen with his partner feeding a single fledgling on the 14 November. The nest fledged two chicks on 13 November, though by 14 November only one fledgling was found with the parents. By 25 November the female had re-nested with a new partner. There were no checks made between 14 and 29 November, and male M/BR was not seen again. The female M/OK disappeared while incubating, three days before the expected hatch date and was last seen on 6 November 2013. Her partner re-nested with another female only 13 days after her disappearance.

The 2014 1080 pest control operation also had a maximum possible active sodium fluoroacetate period of 35 days. With the drop occurring on 23 November, the active phase lasted until the 28 December 2014, during which no SI robins disappeared.

General mark-recapture results

We compared 23 plausible models for a relationship between SI robin survival and a suite of plausible explanatory variables (**Table 6.2**). All models included recapture probabilities specified by time + sex and follow general CJS mark-recapture models.

The models provide compelling support only for a quadratic relationship between rat abundance and SI robin survival. Models with sex, season, active and stoat as well as rats are within 2 QAIC units of the best model (quadratic rats), but following the arguments of Burnham and Anderson (2002) this does not give any indication that these variables are important. The most parsimonious model contains rat tracking rates as a quadratic variable only, with adult SI robin survival rates negatively affected by rat tracking rates. The fitted model is logit(survival) = 3.676 - 4.826 * tracking rate + 5.544 * trackingrate^2 (**Fig. 6.6**).

Table 6.2: Plausible models of the relationship between adult SI robin survival and a suite of possible explanatory variables. All models included recapture probabilities specified by time + sex.

model	npar	QAICc	DeltaQAICc	weight	QDeviance
Phi(~rat + rat^2)	40	1483.82	0.00	0.13	1248.10
Phi(~rat)	39	1484.39	0.57	0.09	1250.85
Phi(~rat + rat^2 + sex)	41	1484.55	0.74	0.09	1246.66
Phi(~rat + rat^2 + season)	41	1485.15	1.33	0.06	1247.26
Phi(~1)	38	1485.28	1.47	0.06	1253.91
Phi(~rat + rat^2 + active)	41	1485.38	1.57	0.06	1247.49
Phi(~rat + sex)	40	1485.42	1.60	0.06	1249.70
Phi(~rat + rat^2 + stoat)	41	1485.90	2.08	0.04	1248.00
Phi(~since1080)	39	1485.94	2.12	0.04	1252.40
Phi(~rat + rat^2 + sex + season)	42	1485.95	2.13	0.04	1245.88
Phi(~season)	39	1486.21	2.40	0.04	1252.67
Phi(~sincecat)	41	1486.69	2.87	0.03	1248.80
Phi(~sincecat + season)	42	1487.20	3.38	0.02	1247.12
Phi(~site)	39	1487.23	3.41	0.02	1253.68
Phi(~rat + rat^2 + stoat + stoat^2)	42	1487.40	3.59	0.02	1247.33
Phi(~season + sex)	40	1487.47	3.65	0.02	1251.76
Phi(~sincecat + sex)	42	1487.78	3.96	0.02	1247.71
Phi(~since1080 + since1080^2)	40	1488.01	4.20	0.02	1252.30
Phi(~sincecat + sex + season)	43	1488.32	4.50	0.01	1246.06
Phi(~site + sex)	40	1488.49	4.67	0.01	1252.77
Phi(~site + sex)	40	1488.49	4.67	0.01	1252.77
Phi(~site * sex + active)	42	1490.32	6.51	0.00	1250.25

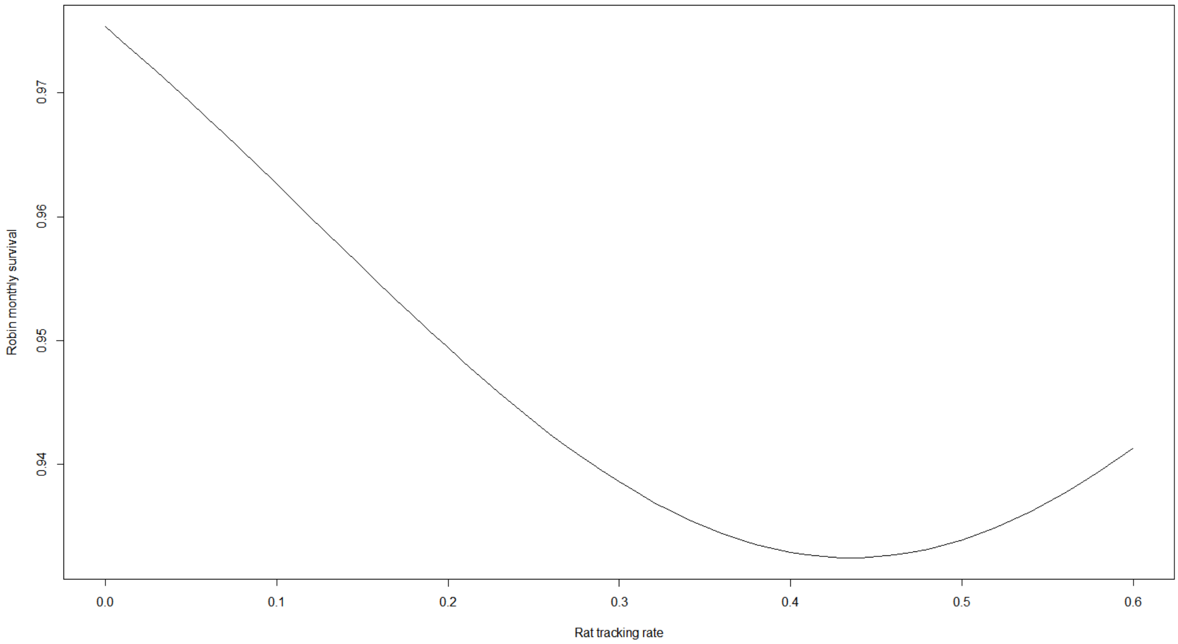


Fig. 6.2: The most parsimonious model using rat tracking rates as a variable to predict South Island robin survival rates: Logit(survival) = 3.676 - 4.826 * tracking rate + 5.544 * trackingrate^2.

6.5 Discussion

Adult survival was analysed over the study period (2012/13 - 2016/17) using monthly survival captures for two samples of 67 robins each, one in the treatment area and one in the non-treatment area of Tennyson Inlet, Marlborough Sounds. We ran competing models in R-MARK accounting for possible explanatory variables such as robin sex and possible by-kill rates after a 1080 pest control effort.

Additionally, we explored variables accounting for predator pressure, in the forms of site, time since a pest control effort and rat tracking rates. Of all the plausible explanatory variables rat abundance in the form of tracking rates was the only variable that showed compelling evidence of having an effect on adults SI robin survival rates. According to our data and models, rats have a negative effect on adult survival rates, with the most parsimonious model contained rat tracking rates as a quadratic variable (logit(survival) = 3.676 - 4.826 * tracking rate + 5.544 * trackingrate^2). This suggests that both rat plagues and predator control affect SI robin survival, as these are the main drivers of rat abundance.

The lack of support for a model containing variables for the active 1080 period or the variables 'since1080' and 'sincecat' are likely in part masked by the noise in our data. In regards to adult survival through 1080 operations, pest control efforts in the 1980s and 1990s used high sowing rates of 10 - 15 kg/ha of carrot baits. These are very high sowing rates coupled with carrot baits which are reported to produce 0.2 - 23.1% chaff easily ingested by birds and in turn caused high by-kill rates (Powlesland et al., 1999). Not only have baits been modified since to reduce chaff, sowing rates have also been reduced to 1 - 3 kg/ha. Our study, like other recent research found very low or un-detectable levels of 1080 by-kill in native birds (Jones, 2016). We found that even though two birds were lost during the active 1080 period of 2013 and 2014 combined this was no different to the loss reported in the non-treatment area during the same time period.

Pest control efforts artificially produce time periods with low rat numbers, and these are therefore expected to have high adult survival based on our most parsimonious model. However, there is no evidence from our models that the variables associated with the pest control efforts and time since a pest control effort (since1080 and sincecat) are useful predictors of SI robin survival rates. This is likely due to the amount of noise in the data. Our null model where survival was taken to be constant ranked fifth out of the 23 competing models assessed. Additionally, the null model is within 2 QAICc of the most parsimonious model. This implies that the even if other variables assessed had an effect on adult survival, the noise in our data may be masking their effect.

The noise in our data may come from a range of sources. Firstly, the relief gained from the pest control efforts was different between the two pest control efforts, though for analysis purposes the two pest control efforts were treated the same. Rats recovered very quickly after the 2013 effort due to the beech mast occurring immediately after the pest control effort (pest control effort in November and seed fall commencing in March). Similarly, after the 2014 pest control effort rats recovered quickly even in the absence of another beech mast. The reasons for this fast recovery are up for discussion, though one of the key drivers is likely the low kill-rate achieved by the pest control effort due to the extremely high rat numbers following the beech mast (**see chapter three**) and possible immigration due to the small size of the treatment area.

Secondly, over the course of the study period, the beech mast and all the flow-on effects from the beech mast, depressed to some degree the long-term gains from pest control operations. The average rat tracking rates above 500 m were very similar between the treatment and non-treatment area, with 19% in the treatment and 22% in the non-treatment area. This is a possible explanation to why time since the pest control effort and site were not reliable predictor variables for SI robin survival rates.

Similarly, there is no compelling evidence that the ‘active’ period is of importance in predicting adult survival rates. This, however, does not mean that there is no robin mortality caused by 1080, but simply that any such mortality is obscured by increased survivorship associated with reduced rat abundance and likely by the large amount of unexplained variation in our models, i.e. noise.

The active periods set after each 1080 pest control effort were set to incorporate a period of 35 days in which by-kill from the pest control effort were possible. However, with the timing of the two pest control efforts being different (2 November and 23 November) a clearly defined 35-day period was not possible within a dataset of monthly survival captures; therefore, the active periods are in part ambiguous. In a perfect setting more time would have been allocated to extract observational data on robin survival within a shorter time period specifically designed for the timing of each pest control effort.

Our study found seasonality (breeding or non-breeding season) and SI robin sex had no detectable effect on survival rates. Other research on robins also found that survival rates did not greatly vary between the sexes (Powlesland, 1983, Armstrong and Ewen, 2002, Drummond, 2017). We had expected to find that sex would be an important variable to include in an adult survival model as between 5.7% and 41.6% females die during the breeding season (Powlesland et al., 2000, Brown, 1997). Our data from nest observations found that in 4.35% of 138 nest attempts the females are killed by rats during a nest predation event (**Chapter four**). In our adult survival models the 4.35% loss of females during the breeding season was not high enough to support an interaction between sex and seasonality, nor for sex alone. Our findings agree in part with research from Tawharanui Regional Park which found no evidence of sex differences in survival rates (Drummond, 2017). Powlesland (1983) found an interaction between seasonality and sex, with similar survival rates during the summer but different survival rates during the winter.

In conclusion, there is no evidence from our models that the active 1080 period negatively affected adult SI robin survival rates. However, survival rates are depressed by rat abundance. It is important to remember, that adult survival is only one variable that contributes to the health and longevity of a population. Juvenile recruitment, nest survival, immigration and emigration are all vital aspects of population models.

Chapter Seven

General Discussion & Conclusions

Discussion

Introduced mammalian predators are known to cause severe impacts on native New Zealand bird species especially during the nesting season, taking eggs, nestlings and even adults (Graham and Veitch, 2002, Brown et al., 1998, Clout and Craig, 1995). Our research investigated the impact of rodents, mustelids and possums on SI robins over the course of five years in Tennyson Inlet, New Zealand. We used tracking rates to monitor rodents and mustelids and wax tags to monitor possum densities in response to pest control efforts (aerial 1080 applications) and in response to a naturally occurring beech mast event in 2014.

Over the course of the study, two separate pest control efforts were conducted using aerially distributed sodium fluoroacetate (1080) pellets at a 1 kg/ha sowing rate. The first 1080 operation successfully depleted the rat population to below 5% tracking rate. The second was conducted during the 2014 beech mast with rat tracking rates of 94% before pest control and 12% after, failing to suppress rats to below 5%. It is possible that after a beech mast event, sowing rates of 1 kg/ha are not high enough to provided bait for all rodents present (Tinnemans et al., 2018, Elliott and Kemp, 2016).

Following the 2014 pest control effort the treatment area experienced fast rodent recovery rates, with rat tracking rates reaching 23% four months and 37% nine months after the 1080 drop. It is unknown whether the fast rat recovery rate observed in the treatment area is due to the number of rats still present after the pest control effort or if the fast recovery rate observed would still have been present if a < 5% tracking rate had been achieved. From other research it appears that small sites recolonise very quickly. Though much smaller, a 12 ha native block near Palmerston North (Keeble's Bush) was recolonised by rats just two months after treatment (Innes and Skipworth, 1983). The Mount Stanley treatment area is a larger area at 34000 ha, though it is a peninsula with a buffer zone along the coast, private land, huts and public walking tracks within the site, thus the distance is likely less than 5 km (< 2 km altitudinally) from any point to the edge of the treated area. This is not a vast distance for a rat to disperse from, especially after a pest control effort where neighbouring territories have become vacant. Stoat and possum dispersal, especially juvenile dispersal, can cover many km. Juvenile possum dispersal is 3.5 - 5.5 km depending on sex (Efford, 1998) and juvenile stoats have been recorded to disperse up to 20 km (King and McMillan, 1982b).

In addition to a low kill rate, the remaining rodent population is equipped with a good food supply and likely highly productive. (McQueen and Lawrence, 2008) found that rat diets contained beech seeds well after the time of germination and speculated that rats cashed seeds securing a food supply. In Kohara, a native forest block in the central North Island, a January sample of rats revealed that 32% of females were pregnant (Brown et al., 1996b). With 1/3 of the female population pregnant at one time, a gestation period of 21 -

23 days and 6 - 12 pups per litter, a rat population could increase 10-fold in just 1 - 2 months. King and Moller (1997) found an increase in overwinter survival rates of adult rats after a beech mast, fuelling population growth, likely providing an explanation to the observed population increase in the treatment area.

In regards to the SI robins, the failure to suppress rats to below 5% and maintain a low rat density means a decreased adult survival rate and lower productivity. We found that all rat tracking rates above 10% had a negative effect on adult robin survival rates. Nest success was also negatively affected with higher rat tracking rates. We found that tracking rates had a quadratic relationship to nest failure rates. During periods of high rat tracking fewer than expected nests were lost due to rat predations. We did not explore how or why SI robin nest predations were not related to nest predations at higher rat densities.

A higher adult and nest survival rate was observed in the treatment area than the non-treatment area implying that the two pest control efforts were of overall benefit to the SI robin population. When the study ceased rodent and mustelid populations had not stabilised to pre-treatment proportions or densities, implying that the full effect on predator population dynamics, from the pest control effort had not yet taken place. We observed by identifying nest predators that nest success figures alone were misleading, as different predators had different predatory behaviour and in turn different effects on the nest success and survival rates of SI robin females.

We observed 54% of nest failure events were attributed to predators, of which 30.6% could be identified as rat predation events from camera footage. If the nest predator was a rat, the female had a 17.8% chance of being injured or killed. From 187 nests, 67% nests were found in 'low risk' sites (branch, fork, ledge or top) and 33% in 'high risk' sites (twigs, nook, vegetation or hollow). Females nesting in a 'low risk' site had a 0.8% chance of injury during a rat nest predation event, in comparison females nesting in 'high risk' sites, had a 11.5% chance of injury during a rat nest predation event. This demonstrates that females choosing protected nest locations are at higher personal risk, during a rat nest predation event. No other predators appeared to harm the incubating or brooding females on the nest over the course of our study period. Boulton et al. (2008) observed that nests had a better daily nest survival rate the older they were, implying that there is variability in nest locations as easy access nests likely fail first. We found some correlation with this idea.

SI robin daily nest survival rate varied between the years. During the rat plague in 2014 both the treatment and non-treatment area experienced extremely low daily survival rates as low as 8% and 3% respectively. This means that re-nests were often missed by staff as the chance of detecting a nest was rather low with whole colony checks occurring every 3 - 10 days. Over the study period the highest number of observed re-nests by a single female was five nests, and this was not uncommon, with a maximum of three nest attempts if nests were successful. With so many nest failures, how many attempts is 'natural' for SI robins? In this current situation where the majority of nest failures are caused by introduced predators. Where is the tipping point that causes harm to the female health due to energy and mineral depletion caused by 'un-naturally' high egg laying?

In 1991 a paper was published on swiftlet energy demands for repeated laying from harvested populations, discussing the available energy based on the estimated daily energy intake and the extra energy used to lay eggs (Kang et al., 1991). They found that both black-nest swiftlet (*Aerodramus maximus*) and white-nest swiftlet (*A. fuciphngus*) females were able to sustain the protein and energy demands, but were not able to replenish lipids. Lipids used for egg laying are assumed to be taken from stored fat within the female's body. The study found that the egg itself did not change, the protein, fat, shell-mass and overall dry-mass and size between first clutches and subsequent re-lays, when nests were harvested. They did find that the re-nesting success rate was significantly reduced for both species if more than one nest was harvested, reducing the overall breeding success.

We did not collect enough data, nor analyse the energy demands and egg make-up within our study. We were therefore not able to explore if a similar effect occurs in years of heavy predator pressures. Such as during a beech mast, when repetitive nest attempts without successful clutches is common. It can be assumed that laying repeated clutches in close succession would impact the female's heath, decreasing her ability to successfully raise a full clutch, undergo post breeding moult and possibly decrease her winter survival rates. This is unlikely to be observed from a small sample size and if explored would need to take into account food availability between seasons and stochastic weather events such as an early winter or frequency of storm events. What we did observe was an increased adult survival rate over the winter months. Additionally, in the later part of the breeding season a decreased interval between nest attempts and a higher nest success rate. Implying that repeated laying had no noticeable negative effect on the female's short-term health or the productivity as was observed with the swiftlets.

Boulton et al. (2008) found that food availability, though paling in comparison to pest control, was a strong driver in NI robin nest success. The publication suggests that research should be conducted giving birds supplementary food to see if productivity is affected. In our study the year of the beech mast, with increased predators (rodents), increases competition for insects, berries and seeds, while the year before the beech mast (2013), when beech is flowering, likely being a year with elevated food supplies (*personal observations*). Though not specifically tested for, we did observe our lowest number of fledglings per successful nest (1.4 fledglings) in 2014 in the presence of elevated rat numbers (food competition). Though the year of the beech flowering (2013) with expected elevated insect life, did not produce the highest number of fledglings per successful nest. Our dataset is insufficient to give these observations credibility, though it may be an interesting avenue for further research.

With most studies on SI and NI robins having been conducted in easy to reach lowland and island habitats, this study widens SI robin nest success and adult survival data for higher altitudinal areas (above 500m). With peaks in the Tennyson Inlet reaching just over 1000m SI robins were found to be clustered around higher areas, generally above 500m altitudinally. The nestling stage appeared to be longer in the populations monitored, with a nest attempt adding to 42 days from first egg to laying rather than the commonly used 38 days (, 1983). This may reflect a poorer diet at higher altitudes. However, with higher rodent numbers in lowland areas, surviving SI robins are likely becoming increasingly altitudinally restricted, to areas with lower predation threat.

This opens many questions for further research. Are there differences in the nestling stages depending on food resources, habitat and altitude in SI robins? Similarly, is there enough genetic mixing in populations of SI robins that are likely geographically restricted to relatively isolated pockets around peaks within a continuous forest system? In regards to pest control and future management of pest control efforts. Further research should be conducted to assess if 1080 sowing rates should be lifted again. Especially when applications are conducted in areas with very high rodent numbers, such as after a beech mast. Additionally, other treatment options should be explored in attempt to dampen or even remove the fast recovery rates of rodents in small treatment blocks after a 1080 pest control effort. I may also be worth comparing different forest systems and treatment blocks nationally to find out if there is minimum size for a treatment block to be effective. It is also possible that the suppression of rodents to below the desired 5% tracking rate is not possible in some systems after a beech mast. Such as coastal forest systems with high edge effect where beech forests are adjacent to mixed forests such as in Tennyson Inlet.

Conclusions

Pest control is vital for the longevity of native New Zealand birds sensitive to introduced predators. Mammalian predators such as possums, rats and stoats are generally regarded as the introduced predators causing the most damage. Sodium fluoroacetate (1080) is the pest control method of choice, with increasingly larger areas being treated. Here, we assessed sowing rates of 1 kg/ha of 1080 pellets in a non-beech-mast year (2013) compared to a mast year (2014) in Tennyson Inlet, New Zealand, in regards to South Island robin (*Petroica australis*) nest success and adult survival.

The use of cameras has not only provided predator identification in 84% of nest predation events (N=138), but has also given an insight into behavioural patterns of both SI robins and the nest predators; stoats (*Mustela erminea*), rats (*Rattus rattus*) and possums (*Trichosurus vulpecula*). Stoats were observed to take SI robin nests exclusively by day while rats and possum hunted by night. Possums appear to accidentally find nests, passing within cm of a nest and appearing to be unaware of the nests' presence. Rats and stoats appear to seek out nests and always destroyed the nest contents. The hunting behaviour of rats, possibly due to stealth and night cover resulted in six SI robin female deaths during nest attacks. Thirteen percent of females attacked by a rat were killed and 4% escaped with only injuries. Overall, 31% of nests were attacked by rats, 21% by stoats, 16% by possums and 3% by the native predator ruru (*Ninox novaeseelandiae*). We did observe a possible nest failure event due to long-tailed cuckoo (*Urodynamis taitensis*) and mouse (*Mus musculus*) though these were no confirmed.

The cameras provided exact data that would otherwise not have been obtained such as hatching and fledging dates. This allowed us to report that SI robins in Tennyson Inlet have a nesting period of 42 days, including three days for laying, 18 days of incubation and 22 days of chick nest days, with incubation starting as the last egg is laid. Females started laying in late August each year and continued up until mid-January. Up to five nest attempts were recorded per season per female and up to three successful clutches. One individual female managed to raise eight chicks to fledgling in one breeding season. However, on average, females raised 1.93 fledglings per successful nest attempt (N=63). On average the interval of time between nest attempts for females was 8.5 days. The re-nesting period was longer at the start of the breeding season than at the end of the breeding season and was affected by nest outcome, with the re-nesting period being 15.3 days if the previous nest was successful and 6.9 days if it had failed.

Predator pressure varied between years, with the mast year of 2014 increasing rat numbers to very high tracking rates (94%) even with a 1080 pest control the year before in the treatment area (2013), and similarly stoats the year after (2015). The pest control effort in 2013 was more effective as it had a higher kill rate, reducing rats to below 5% tracking rate. Possums were unable to recover to the pre-pest-control densities by 2017. Stoats increased the year after the beech mast in the treatment area, but did not become the predominant threat to SI robin nest success as in the non-treatment area.

Nest success before pest control was 34% in the treatment and 17% in the non-treatment are in 2013. In 2014 the ENS was 8% in the treatment and 4% in the non-treatment before the 2014 1080 drop. Introduced predators combined caused 80% of all SI robin nest failures observed (2012/13 – 2014/15). After the 2013 1080 treatment the ENS was lifted from 34% to 77% and in 2014 from 8% to 55% after treatment.

Adult survival was lower in the winter months than during the breeding season, and no difference was found between the sexes during the summer months. The 4.4% of females killed by rats did not appear to be proportionally different from the number of males that disappeared during the breeding season. It is not known for what reasons males disappeared during the breeding season. Overall, adult survival did not appear to be correlated with rat numbers or the mast year, though rats were the cause of at least eight female deaths over the course of the study period.

In conclusion, the one-off 1 kg/ha aerial 1080 pest control effort in response to the beech mast did not manage to suppress rats to below 5% tracking rate. Adult survival and nest success were higher in the treatment than the non-treatment area. Two years after the beech mast aerial 1080 operation, predator proportions were still changing and had not settled back to pre pest control proportions or densities. Nest success alone is unable to give clarity to the effectiveness of a pest control effort in the long term. We believe that pest control efforts can be further improved and tailored to target one species over another between different years by better understanding how key predators interact and how food shortages or availability affect dispersal and productivity, and in turn nest predation events of native birds.

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