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Reintroduction of North Island robins to Paengaroa Scenic Reserve: factors limiting survival, nest success, and population viability in a mainland restoration area.

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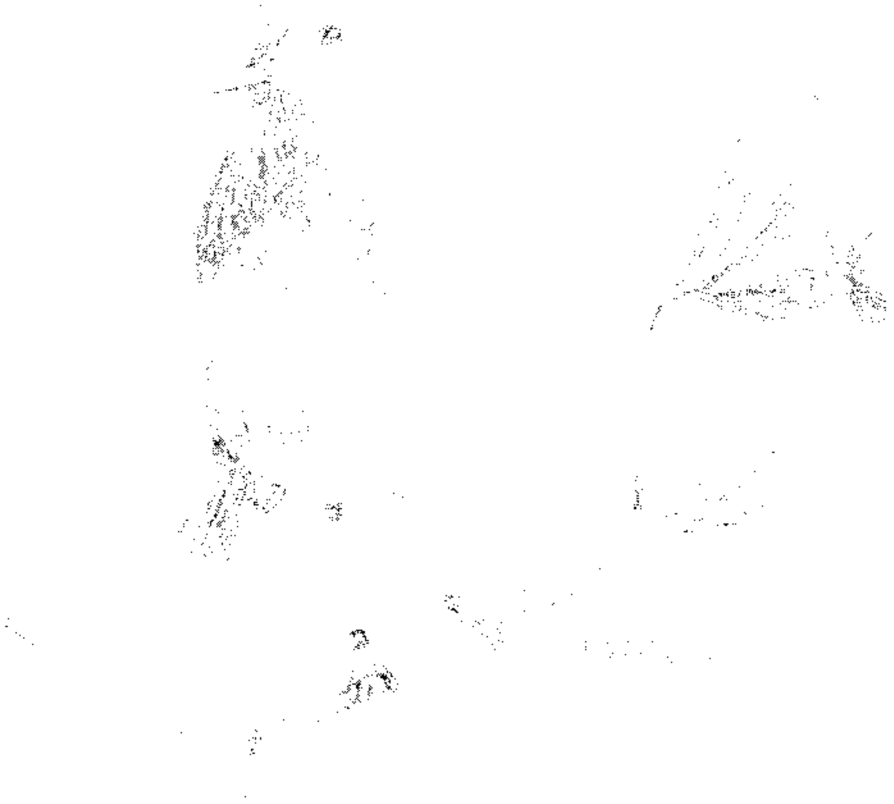

Drawing of North Island robin in Paengaroa Scenic Reserve (by author).

ABSTRACT

Forty North Island robins (*Petroica australis longipes*) were reintroduced to Paengaroa Scenic Reserve in March 1999. I monitored the survival and breeding success of this population for two years post-release. This study aims to assess survival, nest success, and population viability of robins in Paengaroa in an attempt to discover whether habitat in the reserve is likely to support a population of robins.

Survival from the time of release to the start of the first breeding season was lower at Paengaroa than at two other release sites, Boundary Stream Scenic Reserve and Tiritiri Matangi Island. This may be due to higher predator levels at Paengaroa or dispersal out of the reserve.

Methods of estimating nest success were compared, and Stanley's (2000) method was found to have advantages over the traditional and Mayfield methods. Daily survival rates of nests at Paengaroa depended on both the stage in the nesting cycle and stage of the breeding season, with the survival rate lowest for early nests at the incubation stage. Nest success for the first two breeding seasons after translocation was compared to that for the first two seasons after release at Tiritiri Matangi and Boundary Stream. Paengaroa had a similar nest success rate to Tiritiri Matangi (25% and 26% respectively), and both of these sites had lower nest success than Boundary Stream (47%).

Survival at Paengaroa was most affected by whether a bird was recently-translocated, a juvenile, or an adult. Recently-translocated birds and juveniles suffered similarly low survival rates, suggesting that this high mortality may be due to problems faced when finding and establishing a territory. The survival of juveniles from January to September was estimated to be 29%. The annual adult survival rate was also low (59%). Fecundity and survival estimates were used in a stochastic simulation model to predict the viability of the Paengaroa population. Under current conditions, the population was predicted to have a 17% probability of surviving 10 years. However, variation of parameters to lower and upper 95% confidence limits gave survival probabilities of 0% to 100% over 10 years. When data from the first year after translocation were excluded, the population was predicted to have a 100% probability of surviving 100 years. These results demonstrate the large uncertainty associated with small sample sizes and short-term studies.

To assess whether habitat quality is likely to account for the poor overall viability predicted at Paengaroa, the habitat quality at Paengaroa was compared to that at Waimarino forest, where robins still persist. Food supply and predator levels were used to assess habitat quality, as these are obvious factors that may limit viability. Data on food and predator levels provided no indication of why robins may be non-viable at Paengaroa. The power of statistical tests was low due to small sample size, but results suggest Paengaroa has more food as well as fewer rats and stoats than Waimarino. There is a need for further research to improve our understanding of why robins are present and common in some mainland areas but have disappeared from others without any obvious difference in habitat quality. Continued research is also required to reduce the uncertainty regarding population viability at Paengaroa and to determine whether improved management is needed.

CHAPTER ONE

General Introduction

New Zealand is a continental remnant that split off from Gondwanaland in the late Cretaceous period. Eighty million years of isolation has led to a distinctive biota with a high level of endemism, and no terrestrial mammals other than bats and seals reached New Zealand until humans arrived around 1000 years ago (King, 1995). The absence of mammalian predators has allowed traits such as gigantism, flightlessness, and loss of defensive behaviours to arrive and persist. The presence of these traits caused the rapid decline of endemic species after the arrival of mammals, which has had devastating effects on New Zealand's native flora and fauna (King, 1984; Daugherty *et al.*, 1993).

Polynesians settled in New Zealand around 850-950 A.D., bringing with them dogs and kiore (*Rattus exulans*), both of which successfully adapted to the New Zealand environment. Polynesian settlement led to a reduction in native forest area, which was reduced further following mass immigration of Europeans after 1840. This habitat loss has resulted in a highly fragmented forest cover. Europeans also brought with them 82 species of mammals, birds, and fishes (Atkinson & Cameron, 1993). Since human colonisation, at least 40% of New Zealand's landbird fauna have become extinct (Atkinson & Cameron, 1993). This history of habitat loss, fragmentation and extinctions led to the initiation of conservation programs (Saunders, 2000).

The first reserves were established in New Zealand over 100 years ago. However, these were generally focused towards scenic and historic features rather than ecological significance. In more recent years, the necessity of protecting areas for their ecological value has been recognised. In the early 1980s, the Protected Natural Areas Programme was started, aimed at preserving fragments of natural ecosystems on all land tenures (Saunders, 2000). Thirty percent of New Zealand's land area is now formally protected, as are most native vertebrates and remaining native forest areas (Towns & Ballantine, 1993).

Offshore islands have played an important role in New Zealand's conservation history. New Zealand has the highest proportion of threatened avifauna in the world (Clout, 1997). Early conservation efforts were made to move species (including hihi *Notiomystis cincta* and saddleback *Philesturnus carunculatus*) to predator-free islands where it was hoped they would survive. The increasing ability to remove pest mammals from offshore islands has meant that many islands are now used as sanctuaries for species that are continuing to decline on the mainland, and has also enabled the focus of management to change from preservation to restoration (Towns & Ballantyne, 1993). Over 120 successful pest eradications had been successfully carried out on New Zealand islands by the early 1990s (Veitch *et al.*, 1992), and these programs have continued.

Despite the important role of offshore islands in providing safe habitat for threatened species, they only represent a small proportion of New Zealand's environment (Saunders, 2000). Thus, there has been recent emphasis on the management of mainland areas. In 1995, the Department of Conservation allocated funding for 'mainland restoration' projects as part of the threatened species management scheme. Six mainland restoration areas were chosen using specified criteria (see Saunders, 2000). One of the main features of these restoration projects is that they have multi-pest control programs. A major focus of these programs is the intensive control of mammalian pests.

Mammalian predators are responsible for the widespread declines of many indigenous species. The four species of rodent in New Zealand (in order of arrival) are kiore, Norway rat (*Rattus norvegicus*), house mouse (*Mus musculus*), and ship rat (*Rattus rattus*). Possums (*Trichosurus vulpecula*) and three mustelid species, stoats (*Mustela erminea*), weasels (*Mustela nivalis*), and ferrets (*Mustela furo*) are also widespread over New Zealand's mainland. The problem with controlling pest mammals on mainland areas is that reserves are susceptible to constant re-invasion from the surrounding landscape, so eradication is generally not feasible. Much of the predator control on the mainland has involved poison operations, and recent successes can be largely attributed to the development of anticoagulant poisons such as Brodifacoum (Taylor, 1992). Brodifacoum is routinely used for controlling possums, rodents and mustelids (Innes & Barker, 1999). Second generation poisons such as this are persistent and slow-acting, so not only can a lethal dose be consumed in a single feed, but secondary poisoning may be achieved by predators scavenging on poisoned carcasses (Mollër & Alterio, 1998).

Restoring an area's biodiversity involves the intensive control of predators, and may also include the reintroduction of species that have disappeared from that area prior to the start of management. It is very important when planning a reintroduction that the factors responsible for the original extinction are identified and reversed (Veitch, 1995; Kleiman, 1989; IUCN, 1998). Extinction is often associated with a range of simultaneous changes (e.g. habitat loss and introduction of mammals) so it can be difficult to know whether the factors responsible have been correctly identified. There is a large amount of evidence that predators were the primary cause of population declines of endemic species across the country (e.g. Bell, 1978; Saunders, 1992; King, 1995; Brown, 1997a; Brown, 1997b; Clout, 1997; Spurr, 1999; Spurr & O'Connor, 1999; Thomas, 1999). As a consequence, predator abundance is generally the main factor considered prior to a reintroduction in New Zealand.

Translocations have been defined as the intentional release of plants or animals to the wild in an attempt to establish, reestablish, or augment a population (IUCN,1987). A more recent variation on this definition considers translocations to be a "deliberate and mediated movement of wild individuals to an existing population of conspecifics" (IUCN, 1998). The first translocations in New Zealand took place as early as the 1890s when Richard Henry attempted to relocate several bird species to mustelid-free islands (Towns & Ballantyne, 1993). Since this time there have been over 400 translocations mainly involving birds (Armstrong & McLean, 1995), and reintroduction appears to have become an integral part of New Zealand's modern conservation effort.

It has become clear that translocation should be considered long before it becomes a last resort for a species. This is not only because translocations have a low chance of success with small numbers of individuals (Griffiths, 1989), but also because critically endangered species require disproportionately large amounts of resources to prevent extinction. Hence, there has been growing awareness about the sensibility of putting recovery programs in place even if a species is not yet classed as endangered. The increased emphasis on ecological restoration has meant that relatively common species can be reintroduced to areas with a restoration program before their numbers are reduced to levels that are considered critical in terms of conservation status.

The New Zealand robin (*Petroica australis*) is a small (~10cm tall, 35g) insectiverous forest-dwelling passerine. There are three subspecies of New Zealand robin – the North Island robin (*Petroica australis longipes*), the South Island robin (*Petroica australis australis*), and the Stewart Island robin (*Petroica australis rakiura*). They all have grey to black plumage with a white breast, long slender legs, and an upright stance. Robins are generally monogamous, and remain in their territory throughout the year (Flack, 1979). New Zealand robins are still found on the mainland, but have disappeared from much of their former range since European settlement. This is primarily due to habitat loss and predation by introduced mammalian predators (Powlesland, 1997). Robins are susceptible to predation by Norway rats, ship rats, stoats and weasels (Flack & Lloyd, 1978; Brown, 1997a). The species is currently considered to be "regionally threatened" (Bell, 1986), and has become an important part of restoration programs. Robins have been the first species released into some restored habitats due to the relatively low risk associated with their conservation status. Robins have also managed to survive in habitat that other bird species have disappeared from, and are thus useful for predicting the outcome of other proposed translocations to the target area.

The black robin (*Petroica traversi*) is in the same family as the New Zealand robin (Eopsaltriidae), and was considered "the world's most endangered bird" (Butler & Merton, 1992). A translocation was planned for the black robin population from Little Mangere Island to Mangere Island. In 1972 and 1973, South Island robins were used to trial transfer techniques to be used in the black robin translocation, and to evaluate the success of releasing small numbers of birds. Populations were successfully established from five birds on Motuara and Allports Islands, both of which were free from robin predators.

Since these initial translocations, New Zealand robin reintroductions have been carried out as part of restoration projects. During the 1980s, translocations to offshore islands were carried out with varying success. These releases were made into naturally regenerating habitat, and reasons for success or failure are unclear (Armstrong, 2000). During the 1990s, the first reintroductions were done in the context of intensive ecological restoration (including extensive re-vegetation, weed control and pest control). Robin populations were successfully established on Tiritiri

Matangi Island in 1992 and Mana Island in 1996, neither of which had predators likely to threaten robins. In accordance with the increasing trend of mainland restoration, the first mainland robin reintroduction occurred in 1994 to Hinewai Reserve. Robins disappeared within 6 months of release. The reserve had no predator control, but it is not apparent whether this was the cause of failure (Armstrong, 2000). Predator control has been in place for all sites used for subsequent mainland reintroductions (Table1.1).

Table 1.1. North Island robin reintroductions to mainland sites.

Release site	Year
Trounsen Kauri Park, Northland	1998
Boundary Stream Scenic Reserve, Hawkes Bay	1998
Paengaroa Scenic Reserve, Taihape	1999
Wenderholm Regional Park, Nth Auckland	1999
Kakepuku Mountain, Sth Waikato	1999
Mangaokewa Reserve, Te Kuiti	2001
Barnett Reserve & Stephenson covenant, Waikato	2001
Hunua Ranges, SE Auckland	2001
Karori Wildlife Sanctuary, Wellington	2001
Bushy Park Scenic Reserve, Wanganui	2001

Paengaroa Scenic Reserve is a 101 hectare, predominantly broad-leaf/podocarp forest remnant selected by the Department of Conservation to be one of the six original mainland restoration areas. The reserve was chosen as it has a unique botanical assemblage, characterised particularly by the wide diversity of divaricating shrubs. One of the primary goals of the project is to "restore and enhance the outstanding biodiversity" of the reserve (Barkla, 1996). North Island robins were named as a target species for reintroduction in the management strategy for Paengaroa. In March 1999, 40 robins were released into Paengaroa Scenic Reserve.

North Island robins are the first species being reintroduced to many mainland habitats. However, there is little information available about what mainland habitat features are required to establish a viable population of robins. Monitoring the outcome of the translocation to Paengaroa was therefore important, as this was the first reintroduction to a site covering a relatively small area with low-intensity predator control. If a robin population could be successfully established under the current management regime, it would suggest that future robin reintroductions to sites of similar size and habitat quality are likely to be successful.

The aims of this study are: (1) monitor the survival and nest success of a recently translocated population of North Island robins; (2) determine which habitat features are most likely to be affecting survival and nest success rates; and (3) assess whether the habitat quality at Paengaroa is sufficient to establish a self-sustaining population of robins.

This study is presented in four chapters and the conclusions are summarised in the final chapter (Chapter 6). Chapter 2 explains the background to the translocation, and methods used to capture and transport the robins. Survival of robins during the establishment phase at Paengaroa is compared to survival after translocation at Boundary Stream and Tiritiri Matangi Island. Possible reasons for any differences are discussed. Chapter 3 assesses methods used to estimate nest success. A modelling approach is used to compare nest success estimates at Paengaroa to estimates calculated for Boundary Stream and Tiritiri Matangi over the first two breeding seasons after reintroduction. Likely reasons for any differences are explored. Chapter 4 uses a modelling approach to obtain survival estimates for robins in Paengaroa. Parameter estimates are used in a computer simulation program to predict the viability of the Paengaroa robin population. Chapter 5 looks at the foraging behaviour of North Island robins, and explores possible reasons for the low viability found at Paengaroa. Aspects of habitat quality thought to be important (food supply and predator levels) are measured at Paengaroa and the source site (Waimarino) where robins are known to persist. The possible impacts of these factors on robins at Paengaroa are discussed.

CHAPTER TWO

Reintroduction of North Island robins to Paengaroa Scenic Reserve

North Island robin reintroduced to Paengaroa Scenic Reserve.

2.1 INTRODUCTION

One of the most important decisions to be made when planning a reintroduction is whether a particular site currently has habitat suitable for the target species (Veitch, 1995; Wolf *et al.*, 1996). The factors that caused the original extinction must be identified and eliminated, or reduced to a sufficient level (IUCN, 1998). The identification of these factors often proves to be difficult. In many cases there are multiple habitat changes that are associated with population extirpation.

Predation is thought to be the main cause of the population declines seen in New Zealand bird species (Brown, 1994), and has been found to significantly reduce reproductive success in North Island robins (*Petroica australis longipes*) (Brown 1997b; Powlesland *et al.*, 1999). Past North Island robin reintroductions have involved the transfer of birds to sites without predators, or with intensive predator control in place. However, the question of how much predator control is enough to be a "sufficient level" enabling the establishment of a viable population of robins remains unanswered.

In 1998 a proposal was made to reintroduce North Island robins to Paengaroa Scenic Reserve. Paengaroa has been classed as a "mainland restoration project" since 1995, and has low-intensity predator control that is targeted at possums. The North Island robin was noted as a potential candidate for reintroduction to the reserve in the management strategy put out by the Department of Conservation (Barkla, 1996). Paengaroa is within the robins' natural range. Robins are still found in patches of bush in the Rangitikei ecological district, and unsubstantiated accounts from local sources suggest robins may have been in the reserve as recently as seven years ago.

In 1998 the Regional General Manager (Central Region) of the Department of Conservation approved a robin transfer from Waimarino pine forest to Paengaroa Scenic Reserve for the purpose of testing whether a robin population could be established under the current regime of predator control. This project was put under the jurisdiction of both the Wanganui Conservancy and the Palmerston North Area Office. Approval for the transfer was required from local iwi. The Department of Conservation liased with Tamahaki and Ati Hau (Waimarino), and Ngati Rangi

(Paengaroa), and approval was given on the condition that predator levels at Paengaroa were shown to be lower than those at Waimarino.

There have been a number of North Island robin reintroductions in recent years (Table 1.1). In addition to Paengaroa, two of these (Tiritiri Matangi Island, 1992 and Boundary Stream Scenic Reserve, 1998) have involved ongoing monitoring to assess the outcome of each translocation. Similar methods of transfer were used for each project, so the survival rates after these reintroductions can be easily compared and reasons for any differences can be examined. This chapter looks at the predator levels found at these three sites and associated survival rates for the first two months post-release in an effort to understand factors affecting mortality during the establishment phase after a translocation.

2.2 METHODS

2.2.1 *Release site*

Paengaroa Scenic Reserve (Figures 2.2.1) is a 101 hectare bush remnant in the Rangitikei Ecological District. It is situated next to the Hautapu River, near the small township of Mataroa (approximately 10 km north-west of Taihape) (Figure 2.2.2).

The landforms in the reserve include a narrow river terrace and a predominantly southern facing hillslope with an altitude ranging from 520 to 706 metres. The river terrace has a podocarp/broadleaf canopy over an understorey of mainly divaricating shrubs, whereas the hillslope is covered by less complex forest dominated by *Melicytus ramiflorus* and *Coprosma australis* (Barkla, 1996). The reserve is surrounded by farmland, but there are some small forest remnants nearby.

Paengaroa has a unique botanical make-up as well as a range of threatened plant species, and as a consequence it was made a reserve in 1913. It has since been recognised as a priority area for restoration, which led to the reserve being funded by

Figure 2.2.1. Paengaroa Scenic Reserve.

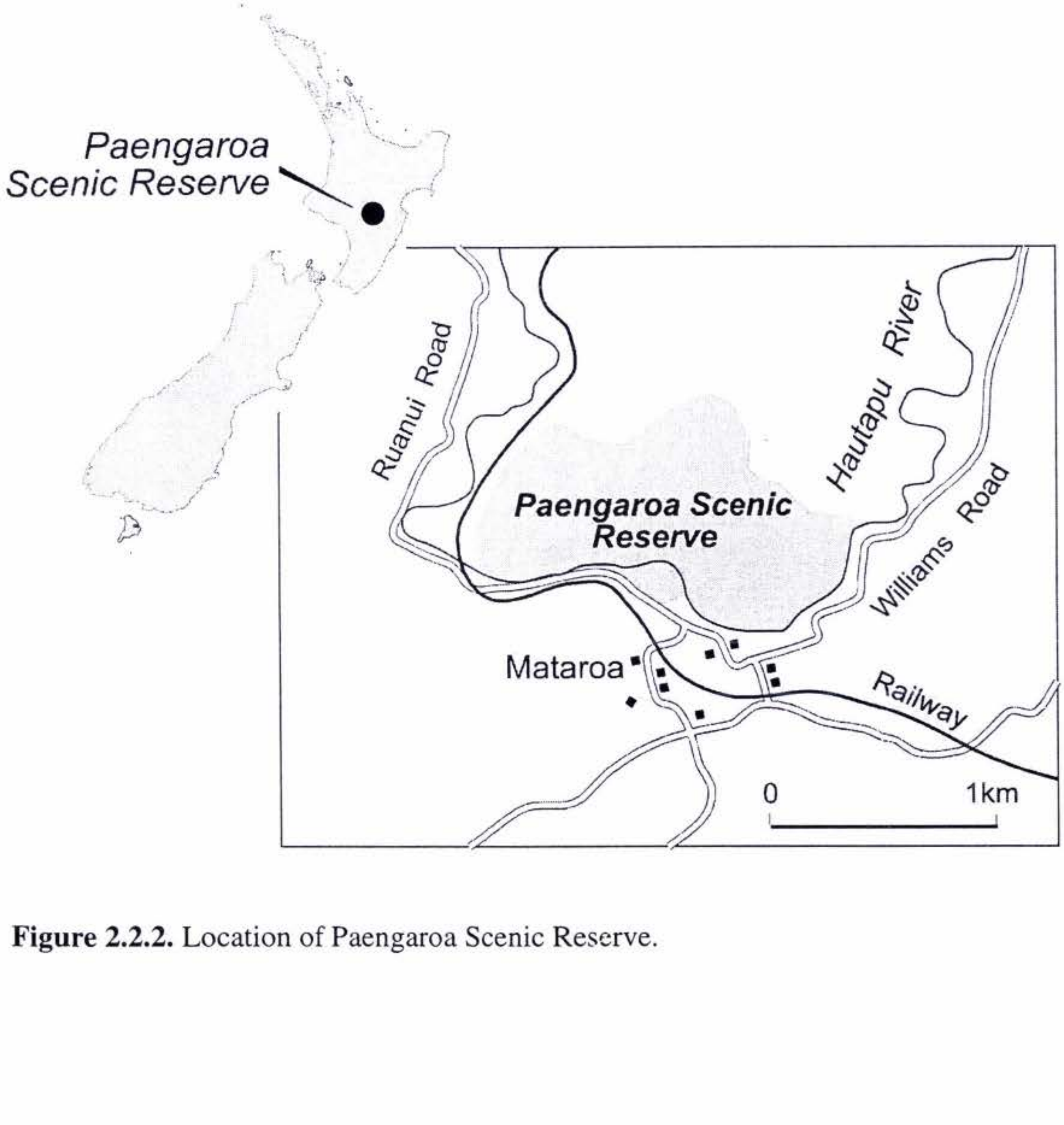

Figure 2.2.2. Location of Paengaroa Scenic Reserve.

the Department of Conservation as a mainland restoration area (previously "mainland island") in 1996 (Barkla, 1996).

Possom control started in the reserve in 1992 with an aerial 1080 drop. Permanent bait stations were then put into the reserve at an average density of 0.96 per hectare, and were filled every 4 weeks with 250g of brodifacoum 'Talon' cereal baits. This predator control regime has continued to the present time except between December 1999 and May 2000 when stations were not baited due to Departmental concerns over secondary poisoning. Brodifacoum is an anticoagulant which acts slowly and is persistent, meaning that predators (such as stoats *Mustela erminea*) can receive a lethal dose by feeding on dying prey or scavenging on poisoned carcasses (Mollër & Alterio, 1998). Brodifacoum poisoning operations are also known to be effective in killing ship rats (*Rattus rattus*) (Brown, 1997a).

2.2.2 *Source site*

Kapiti Island was initially suggested as a source site for the translocation. However, it is unknown whether significant genetic differences have arisen between Kapiti and mainland robins since geographical separation. It was also suggested that Kapiti birds would have lesser predator avoidance skills. Maloney & McLean (1995) and McLean *et al.* (1999) found on an island where stoats were absent, robins responded less strongly to a model stoat than mainland robins. Hence, a mainland source population was required.

Waimarino pine forest is located approximately 20 km west of Ohakune, and covers 13,893 hectares with pine trees of varying ages. The forest is run by Winstone Pulp International Limited. North Island robins present in Waimarino pine forest were chosen as the source population due to the desirability of taking birds from exotic forest rather than native forest. The areas targeted were due to be felled within the next few years, meaning the birds may have benefited from removal out of Waimarino prior to their habitat being destroyed. Waimarino is only 53km from Paengaroa, and both are within the Rangitikei Ecological District.

Within Waimarino pine forest is Anini Native Reserve, which is 1008 hectares and appears to have a high density of North Island robins. Kaitara Road runs around one side of this reserve, and on the other side of the road is mature pine forest from which the birds were captured . There are bait stations located only at the perimeter of the pine forest. They are placed at 100m intervals along Kaitara Road, and are filled with 'Pest Off' brodifacoum bait every four weeks as a general maintenance operation.

2.2.3 ***Previous release sites used for comparison***

Tiritiri Matangi Island

Tiritiri Matangi Island is located in the Hauraki Gulf, 3.5 km offshore from the Whangaparoa peninsula. It is 220 hectares, and there are several small forest remnants remaining (0.15-3.8 ha) despite being used for grazing from around 1955 until 1971. Since 1983 Tiritiri Matangi has undergone revegetation as part of a restoration program. Forty-four robins were released on Tiritiri Matangi in April 1992 (Armstrong, 1995). Kiore (Pacific rats, *Rattus exulans*) were present on the island at this time, but were eradicated 18 months later. A further 14 robins, mainly females, were released in June 1993 due to the small proportion of females in the initial population.

Boundary Stream Scenic Reserve

Boundary Stream covers 800 hectares, and is located 45 km north of Napier. It has predominantly broadleaf and podocarp forest, and is also a mainland restoration area. Since 1996 an intensive predator control program has been in place. Poison bait stations (filled with brodifacoum or pindone) to control rats and possums are situated throughout the reserve at a density of approximately 0.44 per hectare. Cats and mustelids are controlled by trapping. Twenty-eight robins were captured in the Tarawera Conservation Area and released into Boundary Stream from March to May 1998 (Howard, 2000).

2.2.4 *Predator indexing at Paengaroa Scenic Reserve*

Major predators of robin nests are thought to be introduced mammals such as stoats, rats and brushtail possums (*Trichosaurus vulpecula*) (Brown, 1997a). Studies have shown ship rats to be a very significant predator of robins (Flack & Lloyd, 1978; Brown, 1997b; Powlesland *et al.*, 1999), but no evidence has been found that mice (*Mus musculus*) prey upon robin nests (Flack & Lloyd, 1978).

Footprint tracking tunnels rates are a frequently used method of indexing the relative abundance of small mammals in New Zealand (Innes *et al.*, 1995; Brown *et al.*, 1996). Tracking tunnels were used to obtain levels of predator abundance at Waimarino and Paengaroa.

Tracking tunnels

Predator tracking tunnels were laid at Paengaroa, and in both the pine and native forest at Waimarino during December 1998.

Paengaroa has two permanent lines (Line A and Line B) of 25 tracking tunnels placed over a range of terrains and habitats. Waimarino's pine and native forest had one line of 25 each. The tracking tunnels are spaced at 50m intervals. They are made up of a wooden base with a blue corflute covering, and contain a clear plastic tray divided into three sections (Figure 2.2.3). The two outer sections of the tray contain tracking papers, and the central section contains a sponge soaked with red food-colouring. Every time the tunnels were 'run' (approximately monthly), each tracking tunnel was baited with peanut butter to attract rodents. The following day the tracking papers were collected, and the tracking tunnels were then baited with meat to attract mustelids. In this case, only every second tunnel was baited (every 100m) due to the large home ranges of stoats (27 to 453 hectares [Brown & Miller, 1998]), and the tunnels were left for two nights before collecting the papers. The tracking tunnels in Paengaroa were run by DoC contracted workers, whereas I carried out the monitoring in Waimarino.

The tracking tunnels baited for rodents (peanut butter) were run three times prior to the transfer (December 1998-February 1999) (Figure 2.3.1). There were no data collected for February in the pine forest, as most of the tracking papers were removed by possums. The tunnels baited for mustelids (meat) were also run three times (Figure 2.3.2). This enabled the relative predator densitites at each of these sites over three months to be estimated

Mice (Fitzgerald *et al.*, 1981), Norway rats (Moors, 1995), ship rats (Hooker & Innes, 1995), and stoats (Brown & Miller, 1998) all have a travelling range larger than the spacing between tracking tunnels. The tunnels could herefore not be considered independent replicates, so statistical analysis could not be carried out on these data.

Predator indexes were calculated as the percentage of undisturbed tracking tunnels with mouse, rat and mustelid tracks respectively.

Cat boards

Due to the close proximity of Paengaroa to the settlement of Mataroa, it was thought that domestic cats moving into the reserve from the township may prove to be a threat to robins. Ten cat boards were placed at Paengaroa and ten at Waimarino during December 1998 in an attempt to identify cat presence. The boards were made from hardboard, and were 50cm x 50cm in size. Each board was placed on the ground at the base of a tree, and sprayed with a mixture of chalk and methyl alcohol. The alcohol evaporated off, leaving a fine layer of chalk dust on the board. A piece of rabbit meat was nailed to the tree approximately 30 cm above the board. The boards were sprayed at dusk then left overnight, and checked at 6:00 am the next day. Any prints found in the chalk were examined. None of the bait was taken, and all prints were identified as possum. No cat sign was found at either site, leading to DoC instructing that this method of monitoring be abandoned at both sites after one night.

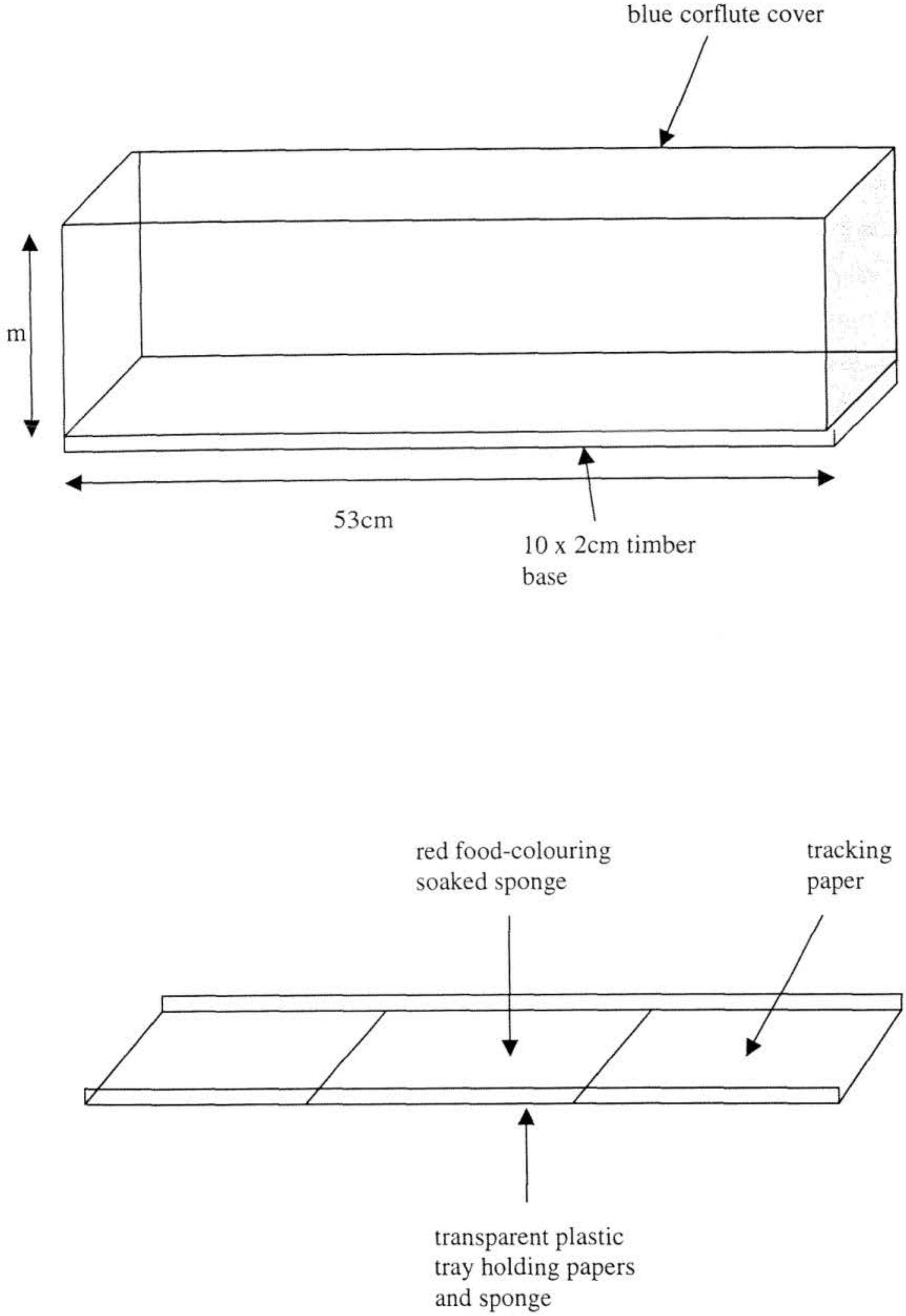

Figure 2.2.3. Tracking tunnel and tray used to monitor rodents and mustelids at Paengaroa Scenic Reserve, and Waimarino pine and native forests.

2.2.5 *Predator indexing at Boundary Stream Scenic Reserve and Tiritiri Matangi Island*

At Boundary Stream rodents are tracked using very similar methods to Paengaroa. Tunnels are spaced at 50m intervals. There are 25 tunnels in each line, and there are 5 lines in the core treatment area. Tunnels are baited with peanut butter, and papers are left out for one night (T. Ward-Smith, pers. comm.). I received tracking tunnel data from Boundary Stream for rodent monitoring prior to release, but mustelid monitoring was not yet in place. Tiritiri Matangi Island did not have any tracking tunnels to index kiore abundance before the robin reintroduction.

2.2.6 *Testing for disease*

Faecal samples were taken from 10 birds in Waimarino pine forest, and sent to a laboratory for worm egg counts and coccidia testing two weeks prior to release. Two veterinarians from Massey University, Maurice Alley and Caroline Twentyman, were present at Waimarino throughout the capture period. A sample of birds were checked for signs of diarrhoea and avian pox. All birds examined were considered to be in good health.

2.2.7 *Robin capture and release*

Birds in the capture area were visited once every couple of weeks, and were trained to eat mealworms (*Tenebrio molitor*) at specific locations within their territories over a period of 3 months prior to the translocation. Some untrained birds were also caught during the capture period (25 & 29-31 March 1999). Forty North Island robins were captured from Waimarino pine forest and released into Paengaroa Scenic Reserve. The sexes and ages of most of these birds were unknown. If a bird could not be located, it was attracted using recordings of a male territorial call. Birds were caught using electronically and cable-triggered clap traps baited with mealworms. All captured birds were banded with a single metal band, and an individual combination of colour bands. The banded robins were then placed in modified cat boxes with a perch, a supply of mealworms and water (Figure 2.2.4), and were held in the shade. Catching took place from 8:00 to 13:30 each day and birds were then driven to Paengaroa and released. All birds survived the translocation and flew away on release, although some showed signs of stress such as beak gape and ruffled feathers while in the transfer boxes.

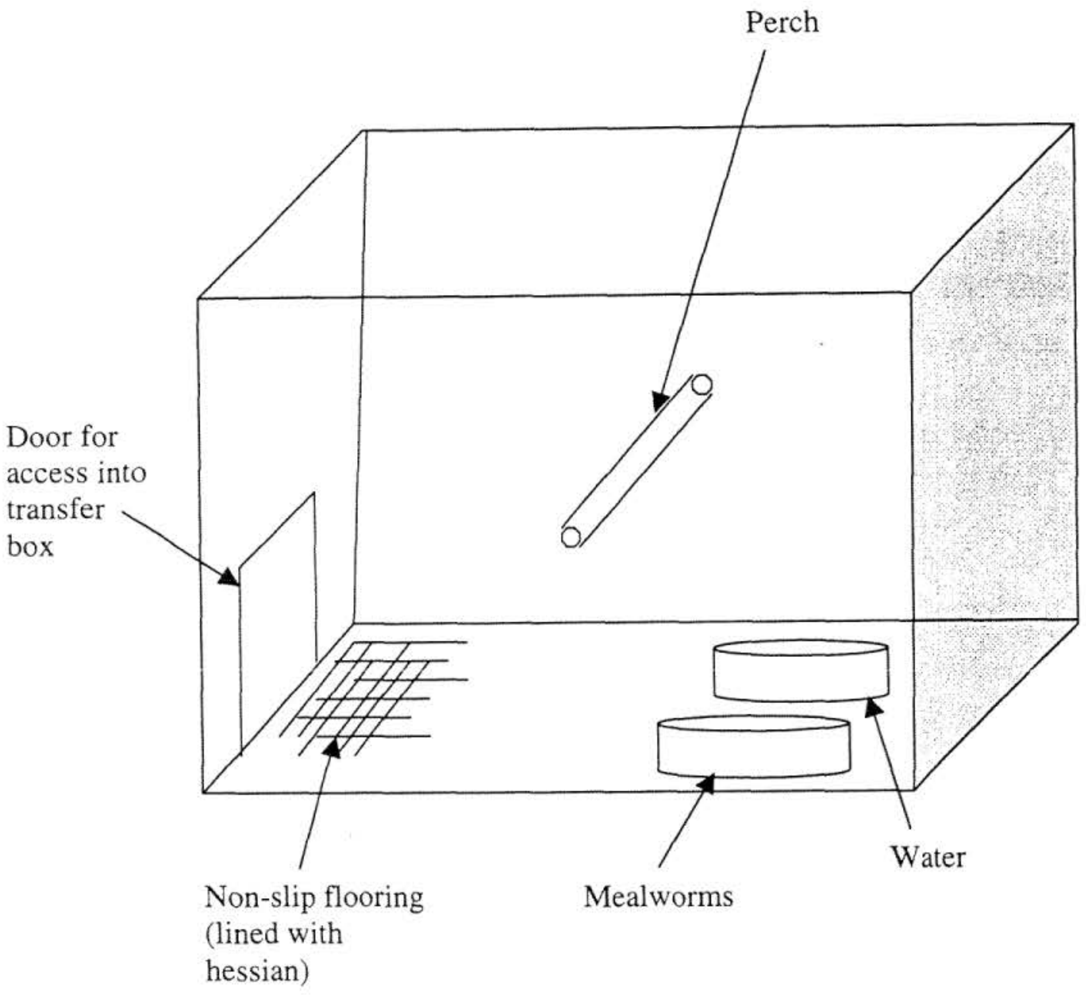

Figure 2.2.4. Construction of transfer boxes used to transport North Island robins from Waimarino pine forest to Paengaroa Scenic Reserve.

2.2.8 *Post-release survival*

Paengaroa Scenic Reserve

I monitored the survival of robins at Paengaroa by performing searches of the reserve and nearby bush remnants. The surveys were designed as a mark-recapture experiment so that survival and resighting probabilities could be estimated (see Chapter 4). Consequently, each survey consisted of intensive searches of the whole reserve over 1-2 days. The two main remnants are located across the Hautapu river from the reserve, and are also Department of Conservation land. The reserve was divided into a grid, where every 100m a robin territorial call was played, and the presence or absence of robins at that site was noted. If a bird was found, its band combination and approximate location was recorded. The first search was carried out two months (May 1999) after the birds were released. A second search took place at the start of September 1999 to identify the minimum number of birds alive at the beginning of the breeding season.

Other release sites

I obtained data on survival of robins at Boundary Stream from Howard (2000). Survival of robins at Boundary Stream was monitored by searching for birds following bait station lines (Howard, 1998). Birds were searched for at approximately 300m intervals, creating a less intense search pattern than at Paengaroa, and different parts of the reserve were searched at different times. It was therefore impossible to estimate survival from these data, and was only possible to note the minimum number known to be alive at any time. Survival of robins on Tiritiri Matangi Island was obtained from D. Armstrong. Surveys on Tiritiri Matangi Island were designed as a mark-recapture experiment, and involved searching remnant patches of bush. One transect line was walked in the smaller patches, and several transects were walked in the larger patches. Searches were carried out in late April (2 weeks after release), early May (4 weeks after release), July (3 months after release) and in September 1992. Survival was estimated for the first two months after release. At both sites, a territorial call was used to attract robins, and the band combinations of individual birds were recorded.

2.3 **RESULTS**

2.3.1 ***Tracking tunnels***

Paengaroa Scenic Reserve

Mouse levels were much higher at Paengaroa than Waimarino, which had a zero tracking rate (Figures 2.3.1 & 2.3.2). Rat levels were much higher at Waimarino, however, with as many as 83% of tunnels in the native forest showing rat prints. The incomplete data on the abundance of predators in the pine forest shows fewer rats overall than the native bush, but still a higher abundance than in Paengaroa.

Mouse and rat levels were higher in tunnels baited with meat than in tunnels baited with peanut butter. This suggests that meat attracts rodents as well as mustelids. The higher tracking rate with meat does not necessarily mean that meat is a better attractant because meat-baited tunnels were left out for two nights instead of one night.

Mustelid tracks were found mainly in tunnels baited with meat. Some mustelid prints were found in tunnels baited with peanut butter at both Paengaroa and Waimarino (Figure 2.3.1), but the levels were very low (0-5%) at both sites. Paengaroa had a maximum mustelid tracking rate of 9% in meat-baited tunnels, whereas the native and pine forests in Waimarino both averaged around 25% (Figure 2.3.2). This suggests that Waimarino had higher levels of mustelids than Paengaroa.

These data were presented to both Winstone Pulp and the Tamahaki and Ati Hau, and approval for the transfer was given on the basis of this information.

Other release sites

Boundary Stream had 0% tracking rates for both rats and mice during the monitoring carried out in October 1996 and February 1997 prior to release (results received from T. Ward Smith).

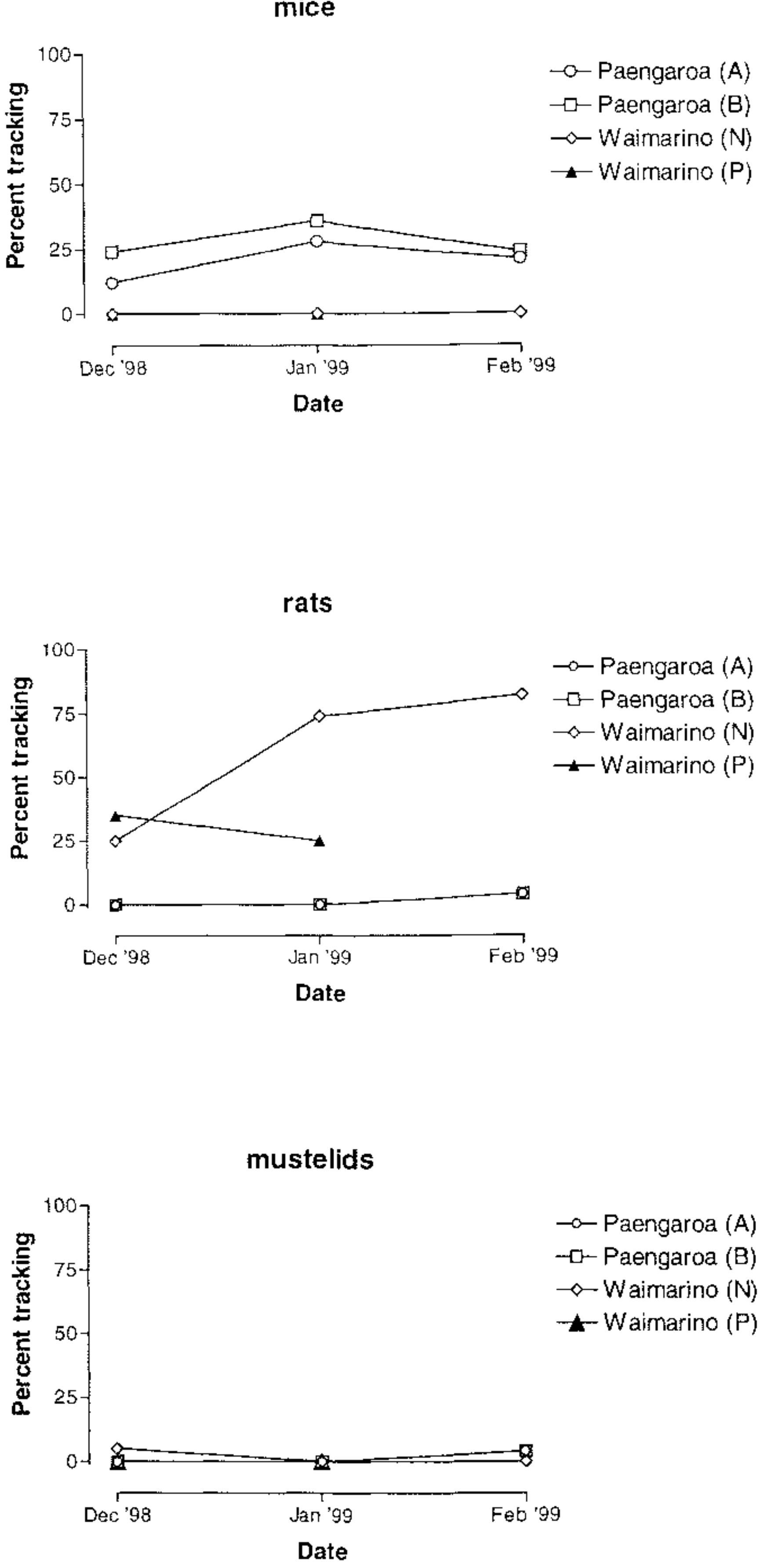

Figure 2.3.1. Predator indices using tunnels baited with peanut butter in Paengaroa (Line A & B) and Waimarino (native forest = Line N; pine forest = Line P).

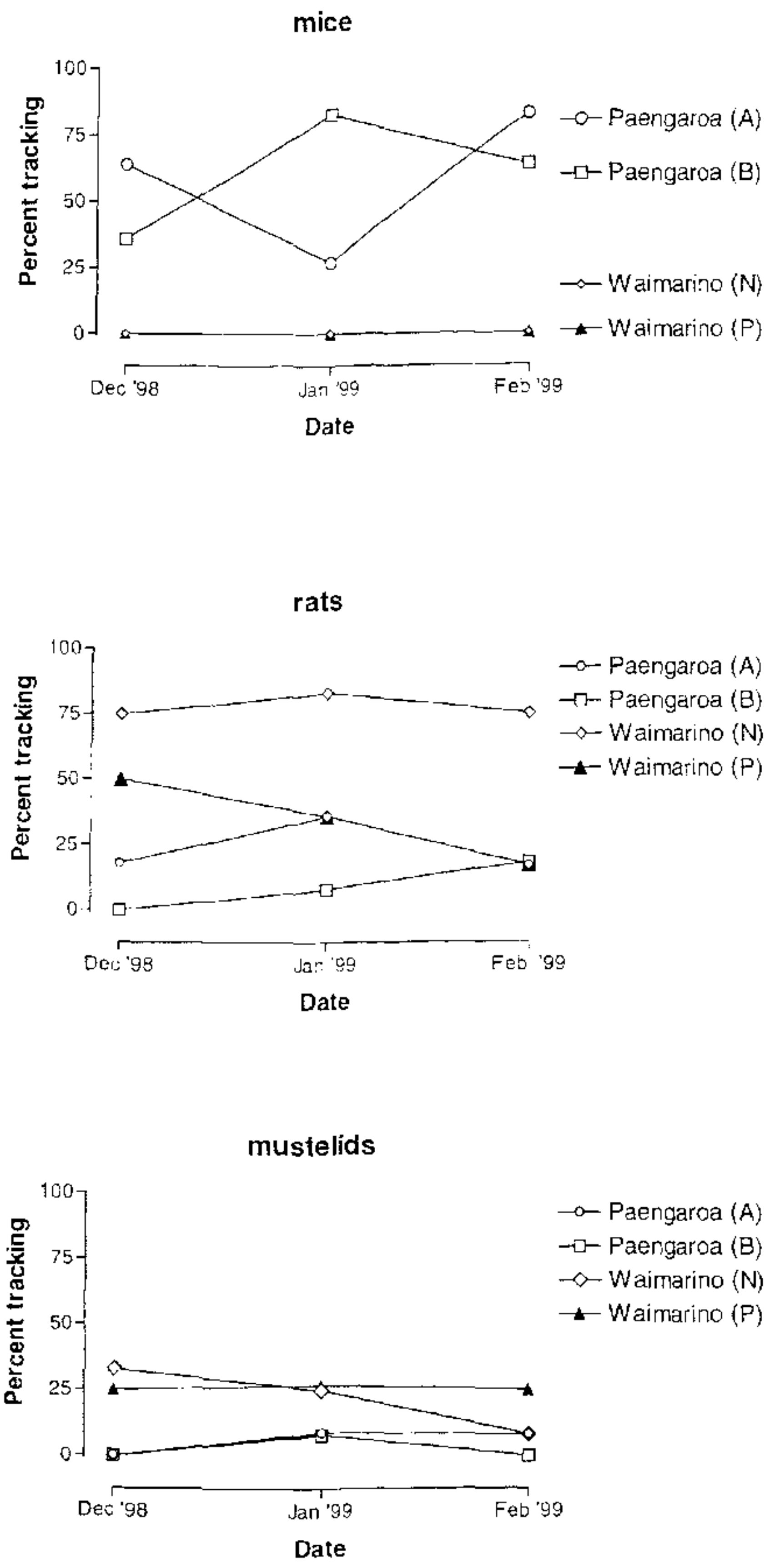

Figure 2.3.2. Predator indices using tunnels baited with meat in Paengaroa (Line A & B) and Waimarino (native forest = Line N; pine forest = Line P)

2.3.2 *Post-release survival*

Paengaroa Scenic Reserve

Of the 40 North Island robins released into Paengaroa, a total of 22 were seen in the survey 2 months after release. Two of these birds were found in a small bush remnant across the river from the main reserve (Figure 2.2.2) but this was considered to be part of Paengaroa. The estimated survival probability over this 2 month period was 55% with a 95% confidence interval of 38%-69%. The search in September found three more adults had disappeared during the 3.5 month period since the initial search, giving an estimated survival rate of 81% (95% confidence interval of 57%-93%) from May to September.

Other release sites

Howard (2000) reported that two birds were never seen post-release, and that a further four birds had since gone missing. Therefore, at least 22 of the 28 birds survived the first 2 months after release. This gives a minimum survival rate of 79% over that time period.

Tiritiri Matangi Island did not have a survey 2 months after release. However, I could construct alternative models of changes in survival from the surveys available, and use these to estimate survival for 2 months (see Chapter 4 for modelling and estimation of survival). Based on the best model, Tiritiri Matangi had 77% survival (95% confidence interval of 64%-91%) over the first two months after release, and subsequently had 95% survival (95% confidence interval of 89%-98%) from June to September. The alternative models that were considered produced similar estimates.

Paengaroa appears to have a lower post-translocation survival rate than both Boundary Stream and Tiritiri Matangi Island. However, 95% confidence limits for survival probabilities are large due to small sample sizes, and this should be taken into consideration when comparing the two sites.

2.4 DISCUSSION

The capture and transport involved in a translocation causes stress (Siegel, 1980) which is indicated by increased levels of glucocorticosteroids such as corticosterone. Stress has been found to increase dispersal (Wingfield *et al.*, 1997) and lower resistence to disease (Viggers *et al.*, 1993). Translocated populations that have been closely monitored often have a high mortality period after release (Slough, 1989; Jones & Witham, 1990; Lovegrove, 1992; Musil *et al.*, 1993; Van Vuren *et al.*, 1997). The low survival rate of robins found in Paengaroa for the first two months after release is in accordance with this finding. Results from Chapter 4 indicate that the survival after a translocation is similar to survival rates for juveniles. This suggests that the higher mortality is not necessarily due to the stress associated with a transfer; rather it may be the normal mortality rate during dispersal and territory establishment.

Paengaroa does appear to have a much lower post-translocation survival rate than other two reintroduced North Island robin populations. Stress is likely to have been a factor in all three translocations as similar methods of capture and transport were used. Hence the differing survival rates must be related to factors acting within each site.

Past studies have shown predation to be the main cause of mortality after translocation (Kurzejeski & Root, 1988; Wilson *et al.*, 1992; Van Vuren *et al.*, 1997). As there is a low survival rate found shortly after translocation in many studies and predation is often thought to be the main cause of these deaths, this suggests that animals may be especially vulnerable to predation directly after transfer. This may be due to the individual's lack of familiarity with the environment (Ambrose, 1972; Gaines & McClenaghan, 1980). O'Bryan and McCullough (1985) noted an initial period where translocated individuals did not actively avoid potential hazards such as predation. Thus, the comparatively low post-translocation survival in Paengaroa may be caused by the higher levels of predators found at this site.

Chronic high levels of corticosteroids may cause behavioural patterns that force birds to leave the reserve and travel over a large area (Wingfield, 1994). Dispersal out of the reserve would lead to an underestimate of the survival of translocated individuals. Past translocations have found dispersal to be common after release (Musil *et al.*, 1993; Van Vuren *et al.*, 1997). Initial studies on the behaviour of New Zealand robins suggest that they are highly sedentary, and only rarely cross open areas (Flack, 1979). However, more recent reports on translocated robins have found that the birds have a much higher capacity for movement than originally thought. Two birds released into Wenderholm Regional Park have been sighted approximately two kilometres outside the managed area (T. Lovegrove, pers. comm.), and five birds released to Ulva Island have flown at least 18 kilometres back to the source site (B. Beaven, pers. comm.). A juvenile robin fledged in Paengaroa was located in Taihape Scenic Reserve, approximately 7 km from Paengaroa (see Chapter 4). These findings suggest it is likely that some of the birds released into Paengaroa may have dispersed out of the reserve to bush patches in the surrounding farmland. Searches were carried out in some patches and no birds were found. However, access to many is restricted and some remnants may contain translocated robins. The likelihood of dispersal is reduced at the other two sites because Boundary Stream Scenic Reserve covers a much larger area (800ha), and Tiritiri Matangi is surrounded by water, with the nearest land 3.5 km from the island.

Another possible reason for the low survival rate found at Paengaroa is the occurrence of disease. Suppression of the immune system occurs when individuals experience high levels of glucocorticosteroids which are induced by stress (Wingfield *et al.*, 1997; Hofer & East, 1998). The stress associated with a reintroduction may increase the pathological effect of infectious agents, and cause further problems for animals already affected by the release process. Even though disease has been a factor in the decline or extinction of some small populations of different species, its importance has been overlooked in the planning of many reintroduction programs (Viggers, *et al.*, 1993). Despite the examinations performed by veterinarians, it is possible the robins released into Paengaroa were "immunologically naiive" to a disease which may have been carried by a closely related species (e.g. tomtits *Petroica macrocephala*) found in Paengaroa. Cunningham (1996) suggests that the translocation of animals to areas without any related species reduces this risk of interspecific transmission of disease.

Diseases can also potentially cause an increased susceptibility to predation (Cunningham, 1996). As no robins have shown any obvious signs of sickness prior to their disappearance, the effect of disease on these populations is impossible to assess. Stress and disease are possible factors in all three translocations. Disease may have been an underlying cause of the poor performance found in the Paengaroa robins, but it is more likely that an interaction between stress, disease, and predation has caused the comparatively low survival in Paengaroa. The most obvious link to survival appears to predator abundance, with Paengaroa having both the lowest post-translocation survival and the highest predator indices.

CHAPTER THREE

Estimation and analysis of nest success

Nesting female robin in Paengaroa Scenic Reserve.

3.1 INTRODUCTION

Studies of nest success are common when assessing the quality of habitat for bird species (Miller & Johnson, 1978). Predation by introduced mammals is one of the main causes of nest failure in New Zealand (Moors, 1983). Hence, it is important to investigate the effects of conservation management (especially predator control) on the nest success of target species.

Nest success is defined as the probability of a nest surviving to result in at least one offspring fledging (Stanley, 2000). The traditional method for estimating nest success was to calculate the proportion of observed nests that are successful. Mayfield (1961) noted that this "apparent success" will be positively biased because unsuccessful nests that fail shortly after initiation are less likely to be observed than successful nests, which persist for the entire nesting cycle.

Mayfield (1961) developed an alternative method for estimating nest success whereby daily survival probabilities ($\hat{p}$) are calculated. The analysis is based on the number of days a nest is under observation. These are termed "nest days". For example, 3 nests observed for 8 days gives 24 nest days, as does 4 nests observed over 6 days. The nest does not have to be visited on all of the days over the observation period. Dividing the total number of losses by the total number of nest days gives the daily mortality rate, and hence the daily survival rate is $\hat{p} = 1 -$ (losses/nest days). When nest checks are made periodically it is often unknown how many nest days there were before a failure occurred. Mayfield (1961, 1975) assumed the failure happened midway between the last two checks. Miller and Johnson (1978) noted that this midpoint assumption gave too much exposure to failed nests, and suggested that the exposure should be 40% rather than 50% for extended intervals between visits (longer than about 15 days). Johnson (1979) showed that the Mayfield estimate was the maximum-likelihood estimator of nest success, and also presented a model which improved on the Mayfield method as it allowed for the date of failure to be unknown.

The Mayfield method can be further improved by calculating separate survival probabilities for each stage, as nest failure rates may be different at the incubation and nestling stages. Mayfield (1961, 1975) and Johnson (1979) suggest calculating stage-specific survival. However, this is problematic unless the exact date of transition from one stage to the next is known and it is known whether the failure occurred before or after hatching. This information cannot be obtained in studies that do not have daily nest checks. Stanley (2000) developed an iterative method that estimates stage-specific daily survival probabilities even if both the date of transition between stages and the date of nest failure are unknown. Stanley's method also allows alternative models to be constructed to assess which factors most affect nest success.

This chapter aims to: (1) assess nest success estimates obtained from Stanley's (2000) method versus the traditional and Mayfield methods using data from the North Island robin (*Petroica australis longipes)* population at Paengaroa Scenic Reserve during its first two years after reintroduction; (2) discover the effects of year, nesting stage, and laying date on daily survival probabilities of nests at Paengaroa by selecting the best model from Stanley's program; (3) compare nest success estimates at Paengaroa to those from an essentially predator-free island (Tiritiri Matangi) and a larger restoration area (Boundary Stream Scenic Reserve) during the first two years after reintroduction to those sites; and (4) compare predator levels at the three sites as a possible explanation for any differences in nest success.

3.2 METHODS

3.2.1 *Monitoring nesting*

I monitored nesting over the 1999/00 and 2000/01 breeding seasons at Paengaroa for the first two years after reintroduction in March 1999. A grid search of Paengaroa was performed at the start of both breeding seasons (early September) to locate individual birds and breeding pairs. Nine pairs were located at the start of the 1999/00 breeding season and 6 pairs at the start of the 2000/01 season. Nesting in the reserve was monitored continuously throughout the first breeding season, but did not start until 11 October 2000 in the second season. I checked every pair at least once a week until the beginning of March to locate nests and determine their success or failure. I fed mealworms to the male of each pair, who would then call the female away from the nest to feed her (the male would occasionally feed her while she remained on the nest). By following the female after the food exchange, the nest could be located. If the female was watched continuously for at least 30 minutes without going to a nest, I assumed that she did not have one. Previous longer-term studies have shown that female robins always return to active nests within 20 minutes (Armstrong *et al.*, 2000; Powlesland *et al.*, 2000).

Due to the nest site selection of most pairs, many nests could not be reached even with a ladder. This meant that the nesting stage had to be deduced from parental behaviour. If no food was being taken up to the nest it was assumed to be in the incubation stage. Conversely, if food was being carried up to the nest it was assumed to be at the nestling stage (Powlesland, 1997). Nest failure was indicated if the female was no longer incubating and neither parent was taking food to young.

If a nest was known to have nestlings, then once the parents no longer showed interest in the nest site, they were followed to see whether there were fledglings present in the territory. The only nests used in the analysis were those that were actually seen (as opposed to those where the parent's behaviour suggested they were nesting, but the actual location of the nest was never found). It is unlikely that any successful nesting attempts were missed given the regularity of checks.

3.2.2 *Estimating nest success*

Nest success was estimated using the traditional method, Mayfield's (1961) method, and Stanley's (2000) method.

Traditional method

The "apparent success" of nests was calculated by dividing the number of nests that produced at least one fledgling by the total number of nests found.

Mayfield method

The total number of losses and nest days were calculated for the whole observation period. Under the Mayfield method,

$\hat{p}$ = 1 - (losses/exposure)

where $\hat{p}$ is the daily survival rate, losses is the number of failed nests, and exposure is the total nest days. I assumed that all failures occurred at the mid-point between checks, as the interval between consecutive checks was never more than 15 days.

Nest success is then estimated as $\hat{p}^t$, where t is the average number of days in the nesting period (40 for North Island robins [Powlesland, 1997]).

Johnson (1979) noted that the standard error of daily survival probabilities obtained by the Mayfield method was given by:

$$se(\hat{p}) = \sqrt{\frac{(exposure - losses).losses}{exposure^3}}$$

I calculated lower and upper 95% confidence limits using Burnham *et al.*'s (1987:214) method. This method (shown below) uses a log-odds transformation on $\hat{p}$, and assumes the transformed variable is normally distributed. The lower and upper confidence limits are given by

$$\hat{p}_L = \frac{\hat{p}}{\hat{p} + (1 - \hat{p})C}$$

and

$$\hat{p}_U = \frac{\hat{p}}{\hat{p} + (1 - \hat{p})/C}$$

where

$$C = \exp\left[\frac{2se(\hat{p})}{\hat{p}(1-\hat{p})}\right]$$

Stanley's method

The program for Stanley's (2000) method can be downloaded from Ecological Archives at http://esa.sdsc.edu/Archive/E081-021. Each interval between nest checks is coded according to its duration (number of days); the stage of the nesting cycle at the two checks (i.e. incubation or nestling); and whether the nest survived the time between checks (coded as 1 0 for survived or 0 1 for failed).

Stanley recognised 5 "nest types" based on the stage(s) of the nesting cycle at the two checks. Type A nests are those found and rechecked during egg-laying. Type B nests are those found during the egg-laying stage, and rechecked during incubation. Type C are those found and rechecked during incubation. Type D nests are those found during the incubation stage, and rechecked after hatching (the nestling stage). Type E nests are those found and rechecked during the nestling stage. Only nest Types C-E were considered as robin nests were rarely found in the egg laying phase.

Each nest was looked at individually, with some nests contributing just one interval to the data set and others contributing several. For example, a nest could be found in the incubation stage, and then fail 3 days later while still in the incubation stage. This would be coded 0 1 3 C. Another nest may have been found in the incubation stage, and when checked 10 days later be found with nestlings, still have live chicks when checked 7 days later, and found to have failed 4 days later. This nest would be coded 1 0 10 D 1 0 7 E 0 1 4 E.

For failed nests, it is necessary to estimate the stage at the last check (i.e. whether it would have hatched by that day). I determined the earliest and latest possible laying dates, and assumed that hatching would have occurred after 19 days (Powlesland, 1997). The stage was still ambiguous for some nests, and I assumed that egg-laying occurred at the midpoint between the possible dates in these cases. I explored the consequences of incorrect stage assessment by switching interval Type C to Type D, or vice versa, and found this to have little effect on the results.

The analysis also requires that fledgling dates are known for successful nests so the amount of days in the last interval only includes the number up to the time of fledging. Once again, these dates were not always known. For some fledglings, it was obvious that they had only been out of the nest for about a day, but in other cases the fledgling dates had to be estimated. This was done by assuming 21 days are spent in the nest after hatching (Powlesland, 1997).

Once nests checks for each nest have been encoded alpha-numerically as described above, the data can replace the sample data included in Stanley's program. The program uses the NLIN procedure in SAS (SAS Institute, 1989). The program (including the data) was pasted into the SAS editor. SAS Version 8 was used.

The SAS output gives estimates of the daily survival probability for each stage, $\hat{p}$, and the associated standard error, $se(\hat{p})$. The confidence intervals shown are calculated with incorrect degrees of freedom (Stanley, 2000), and are based on a t-distribution which is not correct for survival rates, and hence should not be used. Confidence intervals were calculated using Burnham *et al.*'s method (1987:214) (shown above) as recommended by Stanley (2000).

As the data only contained information on the incubation and nestling stage, there were no intervals of Type A or B. Hence, the lines of the program for calculating a survival rate for the egg-laying stage were removed. This meant that daily survival probabilities were obtained for the incubation stage $(\hat{p}_1)$ and the nestling stage $(\hat{p}_2)$ only. The probability of a nest surviving from egg-laying to hatching can be calculated if the duration of the incubation stage (t_1) and nestling stage (t_2) are known. The estimated probability is:

$$\hat{p}_1^{t_1}\hat{p}_2^{t_2}$$

I assumed an incubation period of 19 days and a nestling period of 21 days (Powlesland, 1997). The approximate standard error for this estimate (needed to calculate the confidence interval) can be calculated using the delta method (Seber, 1982: 7-9) as:

$$\sqrt{\left(se(\hat{p}_1)\frac{\partial\left(\hat{p}_1^{t_1}\hat{p}_2^{t_2}\right)}{\partial\hat{p}_1}\right)^2+\left(se(\hat{p}_2)\frac{\partial\left(\hat{p}_1^{t_1}\hat{p}_2^{t_2}\right)}{\partial\hat{p}_2}\right)^2+2\operatorname{cov}[\hat{p}_1\hat{p}_2]\left(\frac{\partial\left(\hat{p}_1^{t_1}\hat{p}_2^{t_2}\right)}{\partial\hat{p}_1}\right)^2\left(\frac{\partial\left(\hat{p}_1^{t_1}\hat{p}_2^{t_2}\right)}{\partial\hat{p}_2}\right)^2}$$

where

$$\frac{\partial\left(\hat{p}_1^{t_1}\hat{p}_2^{t_2}\right)}{\partial\hat{p}_1}=t_1\hat{p}_1^{t_1-1}\hat{p}_2^{t_2}$$

and

$$\frac{\partial\left(\hat{p}_1^{t_1}\hat{p}_2^{t_2}\right)}{\partial\hat{p}_2}=t_2\hat{p}_2^{t_2-1}\hat{p}_1^{t_1}$$

and

$$\operatorname{cov}[\hat{p}_1\hat{p}_2]=r.se(\hat{p}_1).se(\hat{p}_2)$$

The correlation coefficient (*r*) is given in the SAS output.

3.2.3 *Determining factors affecting nest success*

Stanley's (2000) method allows alternative models to be constructed in order to identify what factors affect nest success. Parameters were included in the models based on my knowledge of factors likely to affect nest success. I decided that stage (incubation/nestling), year (1999/00 or 2000/01 breeding season) and season (whether the nest was initiated in the first or second half of the breeding season) could all be potentially affecting nest success at Paengaroa.

It is useful to create different models for survival rates, and determine which is the best. For example, if the daily survival rate is similar during the incubation and nestling stages, it will be beneficial to obtain a combined daily survival rate rather than stage-specific daily survival rates. In order to do this in Stanley's (2000) program, all interval Types (C-E) were made uniform (i.e. all Type C).

In addition to stage-specific survival, separate survival rates were also calculated for both year and season type (early/late). The {year} model calculated a separate survival rate for each year that monitoring took place. The model {season} considered the effect of time of season on nesting success. A nest was classed "early" if it was initiated before 16 October (the median date for the 1999/2000 season), or "late" if initiated after this date. This model estimated the probability of a nest surviving if it was initiated in each of these two season types. Survival was constrained to be constant across stages, years and seasons for the model {constant}.

Increasingly more complex models were made by incorporating more parameters. Under the most complex model {year*season*stage} daily survival probabilities were obtained for each season type at each stage in each year. Eight different models were considered, which consisted of all possible models with the three variables (Table 3.3.1). The models included all interactions among variables as it is not currently possible to include parallel effects without interactions.

The output from the eight models was used to determine which was the best model based on Akaike's Information Criterion, or AIC (Burnham & Anderson, 1998). This value measures the compromise between how well a model fits the data (higher in a more complex model) and the simplicity (fewer parameters) of that model. The lowest AIC value indicates the best model.

$$\mathrm{AIC} = -2\log_e(L) + 2\mathrm{k}$$

where L is the likelihood of the model and k is the number of parameters. The SAS output includes the "Objective", which is the same as the negative log-likelihood of the model ($-\log_e(L)$). For my analysis: k=1 for the model with a combined (constant) survival rate; k=2 for the {stage} model considering the different survival rates in the incubation and nestling stages, but with survival constant across season type (early/late) and year; k is also 2 for both the {season} and {year}models; k=4 for the more complex models that consider both nesting stage and season type {stage*season}, year and season type {year*season}, and nesting stage and year {stage*year}; k=8 for the most complex model {year*stage*season}.

3.2.4 *Comparison of nest success at Paengaroa to Boundary Stream and Tiritiri Matangi*

Raw data were received from nest success studies at Boundary Stream Scenic Reserve and Tiritiri Matangi Island. This analysis focused on the first two breeding seasons post-release for these sites, as the Paengaroa data set only contained information over this time-frame. The breeding population consisted of 6 pairs (1998/99) and 8 pairs (1999/00) in the first two years at Boundary Stream, and 7 pairs (1992/93) and 10 pairs (1993/94) for the first two years at Tiritiri Matangi. The monitoring regime varied between sites. Similarly to Paengaroa, Boundary Stream had weekly checks of all pairs (Howard & Christensen, 2001). Monitoring on Tiritiri Matangi was designed to measure the number of fledglings produced per pair, hence checks were not as frequent during the incubation stage (Armstrong *et al.*, in press).

The methods of analysis described above for Paengaroa were used to estimate nest success at Tiritiri Matangi Island and Boundary Stream.

3.2.5 *Tracking tunnels*

Tracking tunnel data were examined to see if there was any relationship between nest success and predator indices. For Paengaroa, tracking tunnels were run using the methods described in Chapter 2. I took over the predator monitoring from the DoC contract workers, and ran the tracking tunnels at Paengaroa from February 1999 to February 2000. For Boundary Stream, I was given tracking tunnel data collected by other workers. At Boundary Stream there is a core treatment area where rodents are tracked using very similar methods to Paengaroa. Tunnels are spaced at 50m intervals. There are 25 tunnels in each line, and there are 5 lines in the core treatment area. Tunnels are baited with peanut butter, and papers are left out for one night. Mustelids are monitored using tunnels at 100m intervals. There are 8 lines in the core treatment area, with 5 tunnels in each. Tunnels are baited with meat placed in the centre of the sponge, and papers are left out for 3 nights (T. Ward-Smith, pers. comm.). Predator tracking rates from the beginning and end of the first two breeding seasons at Paengaroa and Boundary Stream were used in the analysis. Tiritiri Matangi Island did not have any predator monitoring in place. The only mammals on Tiritiri Matangi were kiore (*Rattus exulans*), which were eradicated following a poison drop in 1993 (between the first two breeding seasons).

3.3 **RESULTS**

3.3.1 ***Paengaroa Scenic Reserve***

A total of 35 robin nests were found over the 1999/2000 and 2000/2001 breeding seasons at Paengaroa. Fifteen of these nests were successful, producing at least one fledgling. Using the traditional method, this gives an apparent nesting success of 43%.

Under the Mayfield method described, there were 648 exposure days, hence the daily survival probability is 0.969 (1-[20/648]). Using Johnson's (1979) estimator, the standard error for this probability is 0.007. The same values are obtained with Stanley's method if a constant daily survival probability is assumed. Using the log-odds transformation, a confidence interval of 0.952-0.981 is obtained. The estimated nest success over a 40 day nesting period (incubation 19 days, nestling 21 days) is 28%, with a 95% confidence interval of 14%-46%.

Modelling with Stanley's program shows that survival in different years is relatively unimportant, but that it is unlikely that daily survival rate is constant for different stages or across the season (Table 3.3.1). It was found that the daily survival rate is most affected by the time a nest was initiated in the season, but that this model is improved by adding separate survival rates for the incubation and nestling stages.

Adding these aspects to the model substantially lowers the AIC. This model {season*stage} gives an estimated nest success rate of 9% if the nest was laid "early" in the season, and 45% if the nest was laid "late". The main difference is at the incubation stage. Early nests had a daily survival probability of 0.902, whereas later nests had a probability of 1.0. In contrast, nests were more likely to survive the nestling stage if they were laid earlier in the season (Table 3.3.2).

Table 3.3.1. Analysis assessing the effects of year, season and stage on the daily survival probabilities at Paengaroa Scenic Reserve.

[1]Model	[2]k	[3]AIC	[3]ΔAIC
year*season*stage	8	97.732	2.426
year*season	4	103.812	8.506
year*stage	4	108.008	12.702
season*stage	4	95.306	0.000
year	2	105.358	10.052
season	2	100.904	5.598
stage	2	105.188	9.882
constant	1	103.780	8.474

[1] Candidate models for factors affecting daily survival probabilities.

[2] Number of parameters in model.

[3] Akaike's Information Criterion value.

[4] Delta AIC, indicating the difference in AIC value from the best model.

Table 3.3.2. Estimated stage-specific survival probabilities and nest success for nests laid early and late in the season at Paengaroa Scenic Reserve.

	[1]n	[2]$\hat{p}_1$	[3]$\hat{p}_2$	[4]$\hat{p}_1^{t_1}\hat{p}_2^{t_2}$
[5]Early	13	0.90(0.81-0.95)	0.98(0.90-1.00)	0.09(0.02-0.36)
[5]Late	22	1.00	0.96(0.93-0.98)	0.45(0.23-0.68)
[6]Overall				0.25(0.09-0.53)

[1] Number of nests.

[2] Estimated daily survival probability (and 95% Confidence Limits) during incubation stage.

[3] Estimated daily survival probability (and 95% Confidence Limits) during nestling stage.

[4] Estimated probability (and 95% Confidence Limits) of nest surviving from laying to fledging.

[5] from model {season*stage} (Table 3.1)

[6] weighted average over whole season based on estimated number of early and late nests.

As monitoring began late in the 2000/2001 breeding season, only 4 out of the 16 nests found were started before the 16th of October. Despite this, it is likely that all nests laid prior to the start of monitoring would either be still present (and thus located), or have previously failed. It is improbable that any successful nests would have been missed because fledglings from these would have been seen. It is therefore likely that early nests suffered similarly poor success as the previous breeding season.

When calculating overall nest success for Paengaroa, a source of bias is introduced due to a higher proportion of nests being detected in the second half of the season when nest success is high. This causes nest success to be overestimated for this late period. As mentioned above, it is unlikely that any successful nests were missed over the study period (this would give the estimate a negative bias), and due to consistent search methods it can be assumed that the probability of an active nest being found on a given search day is the same for both early and late nests. Therefore, the actual number of early and late nests can be estimated by dividing the number of successful nests by the estimated success rate. This gives 33 early nests (3/0.09) and 27 (12/0.45) late nests, and an overall nest success rate of 25% ((33 x 0.09 + 27 x 0.45)/60). The delta method (Seber, 1982; see 3.2.4) was again used to obtain the approximate standard error for this product necessary to calculate the 95% confidence interval (Table 3.3.2).

3.3.2 *Boundary Stream Scenic Reserve*

Thirty nests were found during the two breeding seasons at Boundary Stream. Eighteen (60%) of these were successful. The daily survival probability estimated by Stanley's method under the constant model is 0.983. The nest success rate over the 40 day nesting cycle was 51% with a 95% confidence interval of 28%-70%. Modelling was done on these data using the same parameters as at Paengaroa. The model {season} had the most support, but {constant}, {year} and {stage} also had reasonable support (Table 3.3.3). The AIC values were very similar for all models, and models with more than one factor had less support. These outcomes are probably the result of the small data set used in this analysis.

Table 3.3.3. Analysis assessing the effects of year, season and stage on the daily survival probabilities at Boundary Stream Scenic Reserve.

Model	k	AIC	ΔAIC
year*season*stage	8	73.18	9.36
year*season	4	66.94	3.13
year*stage	4	68.39	4.57
season*stage	4	66.44	2.62
year	2	64.9	1.09
season	2	63.82	0.00
stage	2	64.97	1.15
constant	1	64.07	0.25

After selecting the model {season}, the daily survival rate is estimated to be 0.874 for early nests (initiated prior to 1 November), and 0.989 for later nests (Table 3.3.4). The success rate for early nests is hence 35%, and 65% for late nests. As described for Paengaroa, calculations were made to obtain an estimate of the actual number of nests at Boundary Stream. These gave 23 (8/0.35) early nests and 15 (10/0.65) late nests, and an overall nest success of 47%.

Table 3.3.4. Estimated stage-specific survival probabilities for nests laid early and late in the season at Boundary Stream Scenic Reserve.

	n	$\hat{p}$	$\hat{p}^{t_1+t_2}$
[1]Early	15	0.874 (0.947-0.987)	0.35 (0.11-0.59)
[1]Late	15	0.989 (0.971-0.996)	0.65 (0.31-0.85)
Overall			0.47 (0.23-0.73)

[1]from model {season} (Table 3.3)

3.3.3 *Tiritiri Matangi Island*

At Tiritiri Matangi, 24 nests were found over the first two breeding seasons. Ten (42%) of these produced at least one fledgling. Modelling with Stanley's method using a constant probability across years gives the best explanation of the data (Table 3.3.5). This is likely to be caused by the very small data set rather than no particular factor (or factors) being important.

Table 3.3.5. Analysis assessing the effects of year, season and stage on the daily survival probabilities on Tiritiri Matangi Island.

Model	k	AIC	ΔAIC
year*season*stage	8	91.50	11.25
year*season	4	85.45	5.20
year*stage	4	85.03	4.78
season*stage	4	84.61	4.36
year	2	81.63	1.38
season	2	82.12	1.87
stage	2	81.90	1.65
constant	1	80.25	0.00

Under {constant}, the daily survival probability is estimated to be 0.967 with a standard error of 0.009 and a 95% confidence interval of 0.943 to 0.981. The overall survival rate is thus 26% with a 95% confidence interval of 10% to 62%. The "apparent" nest success estimate (42%) is almost twice that of this estimate obtained from Stanley's method (Table 3.3.6).

3.3.4 *Comparison of methods used to estimate nest success*

The traditional method of estimating nest survival probabilities greatly overestimated nest success compared to the Mayfield method and Stanley's method (Table 3.3.6). Using the Mayfield method gave an identical survival probability to Stanley's method for Tiritiri Matangi, but produced slightly higher estimates for Paengaroa and Boundary Stream. These differences arose because Stanley's method allowed factors affecting nest survival, such as lay date and stage, to be identified. This suggests that Stanley's method is an improvement on the Mayfield method for obtaining more accurate estimates of nest success.

Table 3.3.6. Estimated probability of nest survival from egg laying to fledging using three different methods of analysis.

Site	*n*	[1]Apparent	[2]Mayfield	[3]Stanley	[4]Best model
Paengaroa	35	43%	28% (14%-46%)	25% (9%-53%)	{season*stage}
Boundary Stream	30	60%	51% (28%-70%)	47%(26%-69%)	{season}
Tiritiri Matangi Is.	24	42%	26% (10%-62%)	26%(10%-62%)	{constant}

[1] Estimated probability of survival obtained using the apparent (traditional) method.

[2] Estimated probability of survival obtained using the Mayfield method.

[3] Estimated probability of survival obtained using Stanley's (2000) method.

[4] Best model to explain the data (from Stanley's method).

Using the nest success estimates given by Stanley's method, Boundary Stream has almost double the nesting success of both Paengaroa and Tiritiri Matangi for the first two breeding seasons after translocation.

3.3.5 ***Tracking tunnels***

The tracking rates showed that Paengaroa had consistently higher rat levels than Boundary Stream (Table 3.3.7). The first season at Paengaroa had a high rat index for both the beginning (44%) and end (30%) of the breeding season, whereas Boundary Stream had negligible tracking (1% and 2% respectively). The lower rat index at Paengaroa for the second breeding season was still at least 7 times higher than that found at Boundary Stream, which again had almost no rat tracks. It is interesting to note that the rat index at the end of the second season in Paengaroa is much lower (9%) than previously found at this site. However, the reasons for this are unclear. Mustelid tracking rates were low for both sites across both seasons, although Boundary Stream did have a relatively higher mustelid index at the end of the second breeding season. Interestingly, no obvious trend in tracking rates was found for season type (early/late), which was shown to be an important factor affecting nest success at these two sites. The mouse tracking rates found at Paengaroa were higher than Boundary Stream for the first breeding season, but were similar over the second season.

Table 3.3.7. Percent of tunnels tracked by predators at the beginning and end of the first two breeding seasons after releasing robins into Paengaroa and Boundary Stream.

			Site	
			Paengaroa	Boundary Stream
1st season after release	Beginning (early)	Mice	44%	13%
		Rats	31%	1%
		Mustelids	0%	0%
	End (late)	Mice	30%	12%
		Rats	32%	2%
		Mustelids	4%	2%
2nd season after release	Beginning (early)	Mice	46%	56%
		Rats	22%	3%
		Mustelids	0%	6%
	End (late)	Mice	29%	22%
		Rats	9%	1%
		Mustelids	0%	10%

3.4 **DISCUSSION**

The bias associated with estimating nest success using the traditional method has long been recognised (Coulson, 1956; Hammond & Forward, 1956). Modelling daily survival rates with Stanley's (2000) program allowed the high degree of bias associated with estimating nest success using the traditional method to be clearly seen. The traditional method greatly overestimated nest success at Paengaroa, Boundary Stream and Tiritiri Matangi. The analysis for these sites showed that the Mayfield method gives the same estimate as Stanley's model if a constant daily survival probability is assumed. Stanley's program has an advantage over the Mayfield method as it enables comparison of candidate models, and hence provides improved understanding of the factors affecting nest success rates for each population.

Stanley's method allowed stage-specific mortality to be modelled. Bias is introduced if there are changes in survival probabilities across the incubation and nestling stages, but these are not taken into account. Most monitoring of nests takes place at the nestling stage as nests are often not found until they are at least part way through the incubation stage. If nest survival is higher during the nestling stage than the incubation stage, an additional positive bias is introduced.

Nests laid in the first half of the season had lower survival than later nests at both Paengaroa and Boundary Stream. At Paengaroa this was due to a high rate of failure at the incubation stage for early nests. None of the late nests ($n = 22$) were known to have failed at the incubation stage. These data suggest that the vulnerability of robin nests may change between the incubation and nestling stages, but that the most vulnerable stage for a given nest depends on when it is laid.

The model {constant} gave the best explanation of the data from Tiritiri Matangi. As mentioned previously, this is probably the result of the small data set available for the analysis. I used the first two years after release onto Tiritiri Matangi for direct comparison to data obtained for the first two years at Paengaroa and Boundary Stream. Armstrong *et al.* (in press) carried out a similar analysis on Tiritiri Matangi data, but used information on the first five years after release. This study found that the best model for this population was {trans*stage}, where daily survival rates were different for recently-translocated females (females in their first year after each of the translocations which took place in 1992 and 1993) and between stages, but were otherwise constant. The model {kiore*stage} also had reasonable support, where survival rates were different only for the first year after reintroduction, before kiore were eradicated. Under the {trans*stage} model, nest success was estimated to be 37% compared to 26% success estimated in my study. Armstrong *et al.*'s study demonstrates the necessity of having a larger data set to get a more accurate and comprehensive understanding of factors affecting nest success. It also suggests that predicting future nest success rates using data from the first two years after a translocation may be misleading.

Models incorporating the effect of recently-translocated females on nest success were unnecessary for Boundary Stream and Paengaroa because there was only one translocation. The model {year} allowed for differences between the first and second year, and therefore enabled me to detect whether nest success was substantially lower for recently-translocated females. Using the model selection procedures in Stanley's program, {year} did not give the best explanation of the data at either site. This suggests that whether or not a female was recently-translocated was found to be relatively unimportant to nest success at both Paengaroa and Boundary Stream. Of the two mainland sites, Boundary Stream had a much higher nest success rate (47%) than Paengaroa (25%). The difference was great across both season types: 9% at Paengaroa *vs* 35% at Boundary Stream for early nests, and 45% at Paengaroa *vs* 65% at Boundary Stream for late nests. These results suggest that there is higher predation pressure at Paengaroa than Boundary Stream, and that predation rates may be higher in the first half of the breeding season at both sites.

Rats and mustelids are important predators of North Island robins (Flack & Lloyd, 1978; Brown, 1997a). Mustelid levels were similarly low at both Paengaroa and Boundary Stream at the beginning and end of the first breeding season, but were higher at Boundary Stream during the second season of monitoring (0% at Paengaroa *vs* 6% and 10% at Boundary Stream for the beginning and end of the season respectively). This suggests that robin nest success is not directly linked to mustelid levels at these sites. The most obvious relationship was between rat levels and nest success. Paengaroa had a consistently higher rat tracking rate than Boundary Stream over both breeding seasons, which could account for the poor nest success at Paengaroa. However, this potential reason for nest failure does not explain the lower nest success during the first half of the season at both sites. Boundary Steam had negligible rats across both season types (early/late) for the first and second breeding season. There are several possible explanations for this. Firstly, predation pressure may not be determined solely by predator abundance and may depend on the availability of alternative prey and the predator's food demands. Secondly, tracking tunnel data may not reflect seasonal changes in predator abundance (e.g Murphy & Bradfield, 1992; Alterio *et al.*, 1999). Thirdly, factors other than predator levels such as female experience gained through the season or food supply (Wiehn & Korpimäcki, 1997; Turner & McCarty, 1998) also affect nest success.

The generally lower predator levels at Boundary Stream may be due to the more intensive pest control regime at this site. Although Paengaroa has more bait stations per hectare (0.96) than Boundary Stream (0.44 per hectare), the bait stations at Boundary Stream are placed in a grid pattern with 150m between each station, whereas at Paengaroa the bait stations can be up to 500m apart. Paengaroa may also be a lot more susceptible to reinvasion from the surrounding landscape, as this reserve covers a much smaller area (101 hectares) than Boundary Stream (800 hectares).

It is interesting that despite a lack of significant mammalian predators, the estimated nest success rate on Tiritiri Matangi Island was similarly low (26%) to that at Paengaroa during the first two years after reintroduction. This is partially attributable to the presence of kiore in the first year, as at least two nest failures appeared to be due to kiore predation (Armstrong *et al.*, in press). However, no kiore were present in the second year, and females translocated that year also had poor success. Furthermore, the nest success from 1993-1996 was still much lower than Boundary Streram. This supports the idea that unknown factors such as those mentioned above may be important to nest success. It is also possible that avian predation pressure could be higher on Tiritiri Matangi Island than on the mainland sites. Birds such as harrier hawks (*Circus approximans*), moreporks (*Ninox novaeseelandiae*), and Indian minors (*Acridotheres tristis*) are present on the island, and all are likely predators of robin nests (Armstrong *et al.*, 2000). Paengaroa and Boundary Stream also have moreporks and harriers present, but the higher canopy at these sites may reduce susceptibility to harrier predation.

I recommend Stanley's method for use in nest success studies, as it is easy to use and the model selection procedures allow an increased understanding of the factors affecting nest success for different populations. Stanley's program can be used to obtain more accurate estimates of nest success than traditional methods with less intensive monitoring. For future studies involving North Island robins, it is necessary to ensure that nests are detected as early as possible after laying, and that monitoring occurs equally across both the incubation and nestling stages since survival rates can be quite different during these times. The main advantage of Stanley's method over traditional methods is that it allows the effects of management (e.g. predator control) on nest success to be directly examined. This type of analysis has been previously unavailable to field biologists, and is a major step towards improving our understanding of factors important to nest success rates and assessing the quality of habitat management.

CHAPTER FOUR

Population viability analysis of the robin population at Paengaroa

Juvenile robin fledged in Paengaroa Scenic Reserve.

4.1 **INTRODUCTION**

The success of a translocation is determined by the establishment of a self-sustaining, or viable, population (Griffith *et al.*, 1989). Predictions about the viability of reintroduced populations require an understanding of population dynamics. The dynamics of an animal population result from its mortality, recruitment, immigration and emigration (Lebreton *et al.*, 1993; White, 2000)

In order to accurately assess the viability of a population, long-term monitoring is required. Until recently, most translocations had little post-release monitoring or follow-up (Armstrong & McLean, 1995), so data on the dynamics of a population after translocation were rare. Intensive research following a translocation is becoming more common (e.g. Musil *et al.*, 1993; Green *et al.*, 1996; Armstrong & Perrott, 2000), but long-term data are still largely unavailable. It is often necessary to make management decisions about reintroduced populations within a few years after release, so obtaining long-term data for these decisions is impossible. It is therefore necessary to obtain a preliminary understanding of population dynamics from a limited data set to make these decisions. Making management decisions from such data requires reliable methods of estimating parameters, predicting population viability, and assessing the uncertainty of estimates. Intensive data collection can reduce uncertainty, and may to some extent compensate for a lack of long-term data.

The understanding of population dynamics requires estimation of survival rates. This is particularly challenging as detection of animals within a population can be low or variable. To estimate survival parameters under natural conditions, individually marked animals must be followed through time (Lebreton *et al.*, 1992). Marking individuals (e.g. colour banding) also enables animals to be more easily grouped according to categorical variables such as age, gender, or any other factor likely to influence detection or survival. Since the 1980s, mark-recapture analysis has developed rapidly (Lebreton *et al.*, 1992). This type of analysis enables survival and detection rates for any category to be estimated, and allows a range of models with combinations of potentially important factors to be fit to the data. The model that best fits the data may then be selected, and survival estimates obtained from this model can be used to simulate population dynamics.

Population viability analysis (PVA) is a mathematical modelling tool widely used for assessing the extinction risk of populations over a given number of years. PVA is therefore useful for determining the likelihood of a reintroduced population being self-sustaining, and thus whether that translocation could be considered a success or whether additional management is needed. The advantage of PVA over traditional methods (such as analyses of life tables) is that it takes demographic and environmental stochasticity into account, which is critical for analyses involving small populations (Lacy, 1993).

In this chapter I assess the viability of the North Island robin (*Petroica australis longipes*) population reintroduced to Paengaroa based on data collected over the first two years after reintroduction. Robins are relatively easy to find, but despite this, it should not be assumed that all individuals in the population are detected. I therefore used mark-recapture analysis to estimate survival. I first fit alternative models to mark-recapture data to test whether survival is affected by translocation, yearly variation, gender, or age. I then use survival estimates from the best model in a dynamic simulation model to predict the probability of this population surviving over 10 and 100 year time-frames. I assess sensitivity to uncertainty in parameter estimates, as well as sensitivity to changes in parameters possible through management.

4.2 METHODS

4.2.1 *Monitoring*

Surveys were performed in Paengaroa Scenic Reserve over two years (from May 1999 to May 2001) to obtain data on survival of individuals and population size. Surveys were carried out using the methods described in Chapter 2. The reserve was divided into a grid, where every 100m a robin territorial call was played. Band combinations of the birds seen were recorded. All 40 translocated birds released into Paengaroa were individually colour banded, as were most known fledglings. One female fledgling could not be caught, and she remains unbanded. I organised three searches for each year of monitoring, in May (non-breeding season), early September (start of the breeding season), and late January (end of the breeding season). An encounter histories file was created indicating which birds were found in each survey.

The number of young fledged by all pairs found at the start of the breeding season was recorded. Nests were found by locating the male of each pair and supplying him with mealworms. He would then call the female off the nest to feed her. Nests could be located by following the female after the food exchange. Weekly checks were made of all known nests. If parents were no longer showing any interest in the nest site, they were followed to see whether their chicks had fledged. A nest was assumed to have failed if no fledglings were found. If fledglings were present, the number of young was recorded. Due to difficulty in reaching nests, and to prevent attracting predators to nest locations, juveniles were not banded until they fledged. On one occasion when nestlings were banded in the nest, predation occurred within five days. This suggested that banding may have drawn attention to the placement of the nest, and hence no more nestlings were banded. Fledglings were caught using a hand-net and banded at approximately 3 weeks of age.

4.2.2 *Mark-recapture analysis*

I modelled survival using MARK 1.8 (White & Burnham, 1999; http://www.cnr.colostate.edu/~gwhite/mark/mark.htm). The survey data were made up of "live recaptures", which are analysed by MARK using the Cormack-Jolly-Seber (CJS) model. This model estimates survival (the probability of an animal surviving the time between two consecutive surveys) and resighting (the probability of a live animal being found) for each interval. As the basic CJS model assumes no differences between individuals, a number of alternative models can be created by introducing factors such as an age structure or separating animals into classes, depending on the analysis required.

Plausible candidate models were created for the survival (Ø) of robins based on general knowledge gained about the population during monitoring. Models were selected using an information-theoretic approach (Burnham & Anderson, 1998), which involved comparing models to the data based on AIC_c values (Akaike's Information Criterion, corrected for bias). The best models only include factors that are helpful for explaining variation in the data, and are indicated by the lowest AIC_c value. The AIC_c for each model is calculated by MARK.

I decided that it was reasonable to assume that resighting probability (P) was constant (denoted as P.). The same procedure was followed for each search, and throughout the study there was no indication that year, gender or age affected resighting probability. It is unlikely that weather had an impact on the resighting of robins as all searches were performed on relatively fine days. Keeping the resighting probability constant restricted the number of parameters, which was necessary given the small sample size of birds.

Two analyses were performed. For both analyses, a global model was nominated. The global model must include all factors used in any candidate models, and is thus the most complex model for factors affecting survival and resighting probabilities. A bootstrap goodness-of-fit test was run in MARK to assess suitability of the global models. This test compares the deviance of the real data to the deviance of data sets produced by the program, following the assumptions of the model. If the deviance of the real data falls around the mid range of the manufactured data sets, then the global model is considered to have a reasonable fit to the data. This should detect any major violation of the assumptions underlying the model, such as my assumption that resighting probability was constant. If necessary, the mean bootstrap deviance can also be used to adjust for overdispersion (excess variance in survival and/or resighting rates). This is done by calculating an overdispersion parameter ("variance inflation parameter") termed *c*, and gives an adjusted AIC_c value ($QAIC_c$) (Anderson *et al.*, 1994).

The first analysis included all birds seen in the reserve since the release, and considered the effects of class (*c*) and time (*t*) on survival probabilities. "Time" refers to the interval between surveys (i.e. May-September, September-January, January-

September), and "class" refers to whether an individual was recently-translocated, a juvenile, or an adult during a particular time interval. Birds were categorised as recently-translocated from release (March 1999) until the start of the first breeding season (September 1999) when they became adults. Recently-translocated birds were recorded as first encountered at the time of their release into Paengaroa. Birds that fledged in the reserve were recorded as first encountered at the January survey, and were considered to be juveniles until the start of the first breeding season after fledging. The initial global model created was $\{Ø_{c*t}, P.\}$, where * denotes interactions among factors – i.e. survival was not only estimated for each class and each time interval, but also for each class within each time interval. As monitoring took place over two years, grouping intervals into years was an obvious simplification of the global model. The first year started in September 1999 as birds were still considered recently-translocated until that time. Thus, Year 1 went from September 1999 to September 2000, and Year 2 went from September 2000 to May 2001. Candidate models were created by coding the parameter index matrix (PIM) of the global model to distinguish between years (y) and the three classes. Juvenile and recently-translocated birds had to survive two time intervals before they were considered adults. It seemed reasonable that there might be different survival rates in the first interval compared to the second interval for these classes. For instance, translocated birds may suffer a higher level of mortality in the first few weeks after release due to stress during the establishment phase. Therefore, in addition to distinguishing between years, the model was modified to test whether keeping the two intervals separate or grouping them together gave a better explanation of the data (Table 4.3.1). Once the best model from this analysis was selected, the design matrix was used to create simpler models by removing the interaction between class and year $\{Ø_{c+y}, P.\}$, removing the effect of class $\{Ø_{y}, P.\}$, and removing the effect of year $\{Ø_{c}, P.\}$. Survival was constrained to be constant under $\{Ø., P.\}$(Table 4.3.2).

The second analysis was to determine the effect of gender (g) on survival probability. This data set included adults only, as the gender of both recently-translocated birds and juveniles were unknown. Birds entered the data set when first recorded as adults. The global model was $\{Ø_{g*t}, P.\}$, where survival was estimated allowing an interaction between gender and time interval. As in the first analysis, the number of parameters were reduced by grouping time intervals together in their respective years

$\{\varnothing_{g*y}, P.\}$. This model was further simplified by removing the interaction between gender and years $\{\varnothing_{g+y}, P.\}$, removing the effect of year $\{\varnothing_{g}, P.\}$, and removing the effect of gender $\{\varnothing_{y}, P.\}$. Survival was constrained to be constant across gender and year in the model $\{\varnothing_{.}, P.\}$(Table 4.3.5).

4.2.3 ***Population viability analysis***

A simulation model was created based on the best model selected from the analysis above and from the fecundity data. VORTEX 8.3 (Lacy 1993; http://pw1.netcom.com/~rlacy/vortex.html) was used to simulate population dynamics and estimate the population's probability of survival over 10 and 100 year time frames. I tested the sensitivity of these predictions to the uncertainty in population parameters by varying parameter estimates within their 95% confidence intervals. Fecundity for the two breeding seasons was obtained by calculating the mean and standard deviation of the number of fledglings produced per female alive at the start of both seasons. I assumed that there was a 50:50 probability of a fledgling being male or female, that there would be no catastrophes, that there would be no inbreeding, and that the birds were monogamous. One example of polygyny was witnessed (two females paired with one male), but this is uncommon in New Zealand robins (Armstrong *et al.*, 2000; Powlesland *et al.*, 2000), and has not been seen again in this population.

4.3 RESULTS

The known breeding population was 19 birds (10 females, 9 males) at the start of the 1999/00 breeding season, and 14 birds (6 females, 8 males) at the start of the 2000/01 breeding season. The resighting probability was estimated to be 94% (see below), so it is unlikely that more than 1-2 birds were missed.

A juvenile robin that fledged during the 2000/01 breeding season at Paengaroa was found (after an unconfirmed sighting) approximately 7 km away in Taihape Scenic Reserve. This finding prompted me to make weekly trips from Palmerston North to the largest accessible patches of bush within a 15km radius of Paengaroa. These searches took place over a three month period (May 2001-July 2001), but no further robins were located. This juvenile dispersal suggests that birds have been leaving the reserve. Hence, "survival" as considered in this chapter could be affected by dispersal as well as mortality.

4.3.1 *Survival*

The global model $\{\O_{c*t}, \text{P.}\}$ used in the survival analysis for all birds had a good fit to the data (Bootstrap test, $P = 0.69$). Of the models created to distinguish between years, the most support was shown for keeping intervals separate in both the recently-translocated class and the juvenile class $\{\O_{c1*y}, \text{P.}\}$ (Table 4.3.1). This suggests that survival rates are different between the interval directly after release or fledging, and the following interval.

Table 4.3.1. Comparison of survival models created to distinguish between years for robins in Paengaroa Scenic Reserve.

[a]Model	[b]k	[c]AIC_c	[d]ΔAIC_c
$\{\O_{c*t}, \text{P.}\}$	11	234.49	6.48
[1]$\{\O_{c1*y}, \text{P.}\}$	8	228.01	0
[2]$\{\O_{c2*y}, \text{P.}\}$	7	228.10	0.09
[3]$\{\O_{c3*y}, \text{P.}\}$	7	236.50	8.48
[4]$\{\O_{c4*y}, \text{P.}\}$	6	236.56	8.54

[a] Candidate models for factors affecting survival probability.

[b] Number of parameters in model.

[c] Akaikes's information criterion value.

[d] Delta AIC_c, indicating difference in AIC_c value from that of the best model.

[1] recently-translocated intervals separate, juvenile intervals separate.

[2] recently-translocated intervals separate, juvenile intervals combined.

[3] recently-translocated intervals combined, juvenile intervals separate.

[4] recently-translocated intervals combined, juvenile intervals combined.

Comparison of candidate models developed from $\{Ø_{cl*y}, P.\}$ showed that class has the most effect on survival rates (Table 4.3.2). The model $\{Ø_{cl}, P.\}$ had the lowest AIC_c value. The model $\{Ø_{cl+y}, P.\}$ had an almost identical AIC_c value, indicating a possible difference in survival between years. Despite this ambiguity, $\{Ø_{cl}, P.\}$ was chosen as the best survival model due to the slightly higher support. The resighting rate was 94% under $\{Ø_{cl}, P.\}$.

Table 4.3.2. Comparison of survival models for robins in Paengaroa Scenic Reserve.

Model	k	AIC_c	ΔAIC_c
$\{Ø_{cl*y}, P.\}$	8	228.01	1.48
$\{Ø_{cl+y}, P.\}$	7	226.83	0.3
$\{Ø_{cl}, P.\}$	6	226.53	0
$\{Ø_{y}, P.\}$	5	242.53	16
$\{Ø., P.\}$	2	252.99	26.46

The estimates of survival rates under $\{Ø_{cl}, P.\}$ show that for translocated individuals, survival in the interval directly after release is lower than survival in the following time period. A similar pattern is shown for juveniles, with the initial interval after fledging having lower survival. For both juveniles and recently-translocated birds, survival for the second interval after fledging/release was similar to adult survival (Table 4.3.3).

Table 4.3.3. Comparison of monthly survival probabilities for robins in Paengaroa Scenic Reserve.

Class	Monthly survival probability (and 95% Confidence Limilts)
Recently-translocated (first interval)	0.74 (0.62-0.83)
Recently-translocated (second interval)	0.94 (0.85-0.98)
Juvenile (first interval)	0.76 (0.64-0.85)
Juvenile (second interval)	0.93 (0.61-0.99)
Adult	0.96 (0.92-0.98)

A combined survival estimate for juveniles across both intervals is required for the population viability analysis. The estimated probability of a juvenile surviving from January to September is

$$\hat{p}_1^{t_1} \hat{p}_2^{t_2}$$

Where $\hat{p}_1$ is the estimated probability of a juvenile surviving the first time interval, $\hat{p}_2$ is the estimated probability for the second interval, and t_1 and t_2 are the associated durations of those intervals (3.5 months and 4 months respectively). The delta method (Seber, 1982; see Chapter 3) was used to obtain the approximate standard error necessary to calculate the confidence interval for this estimate (Table 4.3.4). Yearly adult survival probability was obtained by raising monthly survival probability (Table 4.3.3) to the power of 12.

Table 4.3.4. Survival estimates for adult and juvenile robins in Paengaroa Scenic Reserve.

Class	Survival estimate (and 95% Confidence Limits)
Juvenile (Jan-Sept)	0.29 (0.12-0.54)
Adult (Sept-Sept)	0.59 (0.38-0.75)

Goodness-of-fit testing for the global model {Ø$_{g*t}$, P.} used in the adult analysis suggested that this model had a reasonable fit to the data (P=0.17). However, comparison of observed and bootstrapped deviance suggested some overdispersion (c = 1.46), which was corrected for by using QAIC$_c$ instead of AIC$_c$. Comparison of candidate models showed the highest support for {Ø., P.}, but also reasonable support for {Ø$_{y,}$ P.} and {Ø$_{g,}$ P.}, indicated that there could be a different probability of survival for males and females or between years (Table 4.3.5). The first year after release had a lower adult survival than the second year, and males had higher survival than females. The small sample size (22 birds) involved in this analysis caused the large confidence intervals associated with all three models (Table 4.3.6).

Given that no clear effect of gender was detected, the adult survival probability from the first analysis (Table 4.3.4) was used for the simulations in VORTEX. The adult survival probability for the second analysis (Table 4.3.6) was slightly higher. This would have been due to birds not entering the data set until they were recorded as adults. I therefore considered the first analysis to be more accurate.

Table 4.3.5. Comparison of survival models for adult robins in Paengaroa Scenic Reserve.

Model	k	[1]QAIC$_c$	Δ QAIC$_c$
{Ø$_{g*t}$, P.}	11	81.92	17.70
{Ø$_{g*y}$, P.}	5	67.09	2.87
{Ø$_{g+y}$, P.}	4	66.72	2.50
{Ø$_{g}$, P.}	3	65.96	1.74
{Ø$_{y}$, P.}	3	64.95	0.73
{Ø., P.}	2	64.22	0.00

[1] Corrected Akaike's information criterion adjusted for overdispersion (c = 1.46).

Table 4.3.6. Comparison of yearly survival probabilities for adult robins in Paengaroa Scenic Reserve.

Model		Yearly survival probability (and 95% Confidence Limits)
{Ø$_{y}$, P.}	Year 1	0.54 (0.33-0.74)
	Year 2	0.85 (0.37-0.98)
{Ø$_{g}$, P.}	Male	0.70 (0.43-0.88)
	Female	0.55 (0.29-0.78)
{Ø., P.}		0.66 (0.40-0.81)

Fecundity was lower in the first year after the translocation (1.4 fledglings per female) than in the second year following translocation (2.7 fledglings per female). Mean fecundity was 1.88 fledglings per female, with a standard deviation of 1.80.

4.3.2 *Population viability analysis*

Using fecundity data and survival estimates from the model $\{\text{Ø}_{a/}, P.\}$, the population was predicted to have a 17% probability of survival over 10 years and 0% probability of survival over 100 years. When parameters were set at their lower and upper 95% confidence limits, the probability of survival over 10 years went from 0% to 100% (Table 4.3.7)

Table 4.3.7. Survival probabilities over 10 year and 100 year time frames for the robin population in Paengaroa Scenic Reserve.

	Probability of survival	
	10 years	100 years
Survival estimate	17%	0%
Lower 95% confidence limits	0%	0%
Upper 95% confidence limits	100%	99%

When parameters were varied independently, fecundity had to be increased to 5 fledglings per female before the population had a 100% probability of surviving 10 years (Figure 4.3.1). The annual survival of both adults and juveniles had to increase to 90% before the probability of the population surviving 10 years was 100%. Major increases in survival probability over 100 years occurred when adult survival increased from 80% to 90%, and when juvenile survival increased from 60% to 70% (Figure 4.3.2).

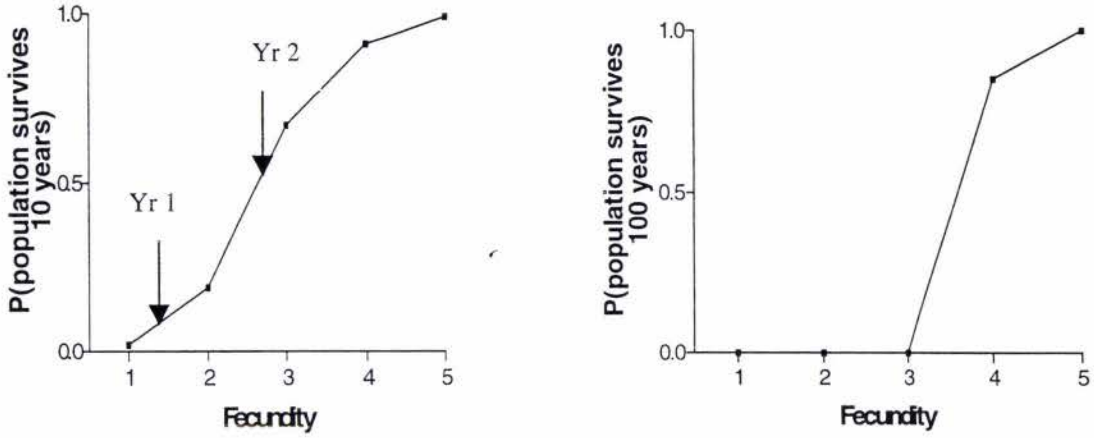

Figure 4.3.1. Predicted increase in population viability with increasing fecundity. Arrows indicate average fecundity rate for the first two years.

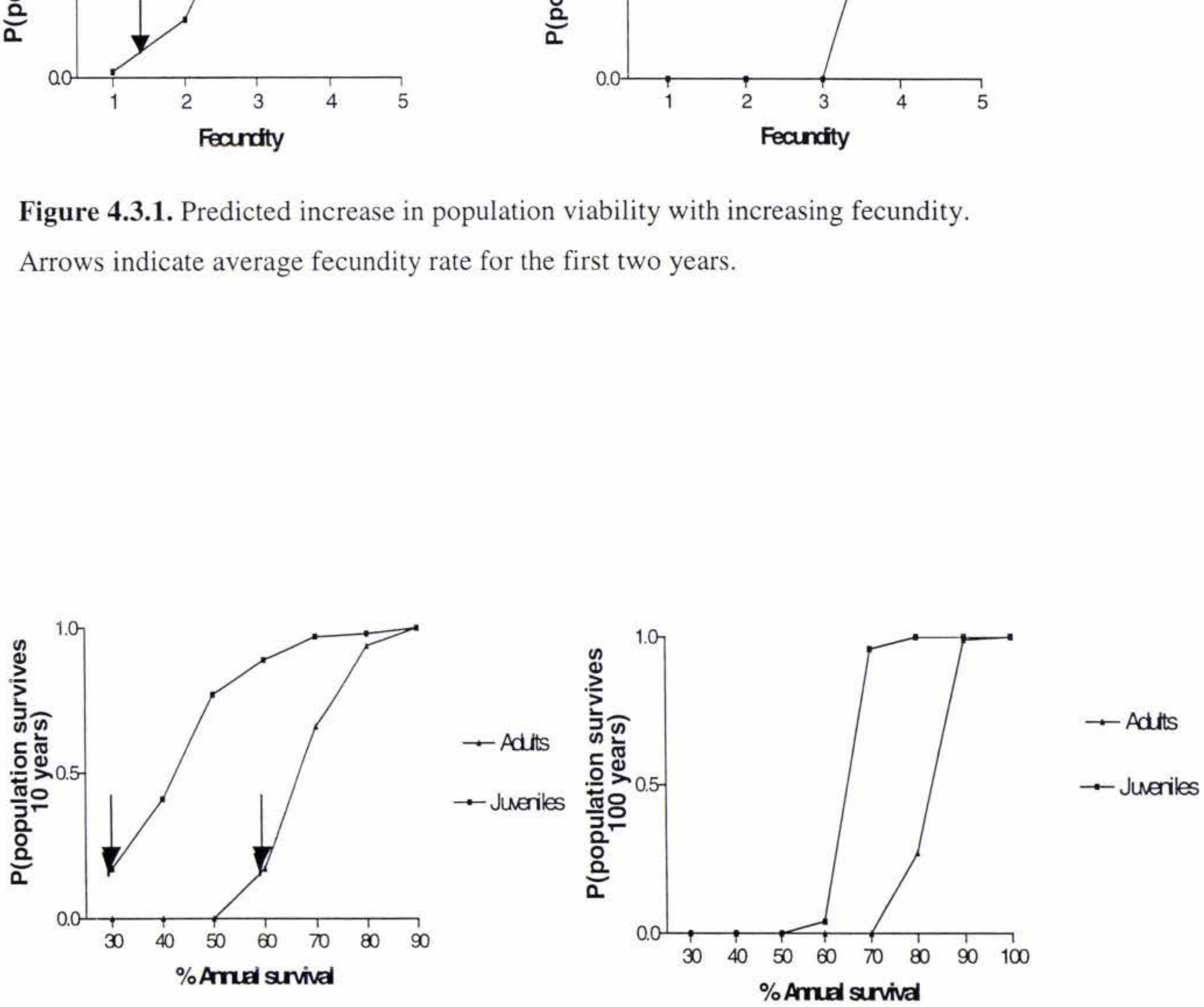

Figure 4.3.2. Predicted increase in population viability with increasing juvenile and adult survival rates. Arrows indicate current survival estimates.

I wanted to test what effect increased predator management would have on the survival probability of the Paengaroa robin population. Tiritiri Matangi Island has a robin population that has been intensively studied in an environment free of mammalian predators. I obtained parameter estimates from the Tiritiri population (Armstrong & Ewen, in press) to use as a baseline for a predator-free location so that I could work out what effect an increased level of management might have on fecundity, juvenile survival, and adult survival at Paengaroa. These Tiritiri data did not include fecundity or survival estimates of recently-translocated birds (i.e. the population parameters were obtained after kiore were eradicated from the island). I calculated mid-way values between estimates from Tiritiri and Paengaroa. It seemed a reasonable assumption that if predator numbers were sufficiently reduced, then parameter estimates at Paengaroa could be predicted to improve to these levels. The predicted mid-way values were 2.15 fledglings per female, 46.5% juvenile survival, and 69% adult survival. Using these estimates, the population had a 99% probability of surviving 10 years and a 94% probability of surviving 100 years. These results suggest that if improved management caused an increase of 0.3 fledglings per female, as well as juvenile and adult survival to improve 24.5% and 10% respectively, then the robin population at Paengaroa would have a low probability of going extinct over the next 100 years, compared to the 100% probability of extinction under current management.

I also estimated the viability of the Paengaroa robin population using parameters obtained from the second year of monitoring only. The results from this analysis predicted the population to have a 100% probability of surviving over 100 years. This suggests that the Paengaroa population may have had unusually low survival and fecundity during the first year after the translocation, and that using data from the first year after a reintroduction may produce misleading results. If parameter estimates from the second year are indicative of future survival and fecundity rates, then the population has a high likelihood of persisting for 100 years, even under the current level of habitat management.

4.4 DISCUSSION

Model selection procedures effectively identified factors that had the most effect on survival, and eliminated factors that were relatively unimportant. Gould and Nichols (1998) suggest that the failure to account for sampling variability can lead to inaccurate estimates of survival probability. The small sample size in these analyses increased the likelihood of Type II error due to the low power of statistical tests. However, the sensitivity analysis using confidence intervals for parameter estimates should have given a reasonable indication of the uncertainty in population viability.

Whether an individual was recently-translocated, a juvenile, or an adult was the key influence on survival estimates. Of these three classes, adults had the highest survival probability. Post-translocation survival rates are often lower than subsequent survival rates (Kurzejeski & Root, 1988; Lovegrove, 1992; Musil *et al.*, 1993; Armstrong, 1995). The low survival of recently-translocated birds and juveniles was largely attributable to the first interval after release or fledging where monthly survival was estimated to be similarly low for both classes (74% and 76%, respectively). This is consistent with results obtained from a robin population on Tiritiri Matangi Island (Armstrong & Ewen, in press), and suggests that the high mortality may be due to problems faced when finding and establishing a territory.

Under current parameters obtained from the most plausible model, the robin population at Paengaroa Scenic Reserve is predicted to have a low probability (17%) of surviving the next 10 years. The accuracy of PVA depends on reliable data (Coulson *et al.*, 2001). Due to the small sample size available for this analysis, the 95% confidence intervals for survival estimates were large. Sensitivity analysis is commonly used in PVA (Taylor, 1995). The sensitivity analyses performed using the upper and lower limits indicate how sensitive population persistence is to changes in survival. Under the worst-case scenario, the population had 0% probability of surviving 10 years. However, using the upper confidence limits, the population had a 99% chance of surviving 100 years. This large variation in survival probabilities demonstrates the requirement of a large data set to achieve more reliable results. However, Fieberg & Ellner (2000) argue that even with extensive data sets it is impossible to accurately predict survival probabilities unless the predicted probabilities are very close to 0 or 1.

The sensitivity analysis investigating the effect of altering population parameters to mid-way between their current levels and those of robins on Tiritiri Matangi Island produced the most important result. With a modest increase in predator control (operations specifically targeting rats and mustelids), it is realistic that survival and fecundity could be raised to these levels. The results from this analysis predicted that under these conditions, the population had a negligible chance of becoming extinct in the next 10 years, and a 94% probability of surviving over 100 years.

These results show how useful PVA can be for identifying risk factors (e.g. predator levels) for a population (Taylor, 1995). However, caution should be taken to ensure that the risk has been correctly identified, as there may be multiple factors interacting to cause low expected viability rather than one obvious threat. Nevertheless, it is clear that relatively small improvements to survival and fecundity will lead to a much higher probability of population survival.

Armstrong and Ewen (2001) noted that fecundity was much lower for female robins in the first year after a translocation, which led to an inaccurate population viability analysis. This suggests that doing a PVA where much of the data is from the first year after a translocation (as in my study) may produce misleading results. Fecundity and survival rates were both lower in the first year after the translocation than in the subsequent year. The analysis excluding parameter estimates from the first year after reintoduction predicted the population to have a 100% probability of persisting for 100 years. This suggests that survival and fecundity may have been unusually low in the first year after release, causing population viability to be greatly underestimated. It would be useful to obtain longer-term (at least 5 years) data from the Paengaroa population to discover whether estimates from the second year after reintroduction are more indicative of future survival and fecundity than those from the first year. However, it is not reasonable for my analysis to exclude the first year's data until it is known whether these estimates were unusually low.

As the population has a low predicted viability under current conditions, this suggests that the most crucial pre-requisite for a reintroduction "identification and elimination, or reduction to a sufficient level, of previous causes of decline" (IUCN, 1998) has not been met. Whether these factors have not been correctly identified, or that the habitat has not been adequately repaired is unclear. It is possible that the predator levels at Paengaroa remain too high to allow a population to persist. Whatever the case, further research is required to determine whether population parameters remain at similar levels to those found in the second year after translocation, and to increase the understanding of why a self-sustaining population has not been able to be established in Paengaroa. This understanding is essential to the prevention of unsuccessful reintroductions in the future.

Male North Island robin in Paengaroa Scenic Reserve.

CHAPTER FIVE

Habitat factors limiting population viability at Paengaroa

5.1 **INTRODUCTION**

One of the key concepts in New Zealand conservation over recent years is the effective management of certain habitats, often as part of a restoration program. "Habitat" relates the presence of a species to features of the physical and biological environment (Armstrong & McLean, 1995), and may include aspects such as vegetation type, food supply, climate, predator levels, or presence of competitors. The basis for any habitat management plan is the ability to assess habitat quality accurately (Van Horne, 1983). Restoration programs generally incorporate the reintroduction of one or more species that have disappeared from the habitat prior to the start of management. Determination of the habitat requirements of those species is a critical part of the translocation (Kleiman, 1989; Maunder, 1992), and there is a low chance of success without high habitat quality (Griffith *et al.*, 1989).

The success of a reintroduction is determined by the establishment of a self-sustaining population (Griffith *et al.*, 1989), and hence relies on recruitment into the population post-release. In a geographically isolated population, recruitment will depend entirely on reproduction. Reproductive success appears to be constrained primarily by environmental quality (Riddington & Gosler, 1995). Thus, it seems important to assess the habitat quality of a site for the species before reintroduction. However, our knowledge of habitat requirements is often inadequate, and synergistic effects among resources tend to be either ignored or misunderstood (Van Horne, 1983). This makes it difficult to decide which habitat features are essential to species survival.

New Zealand's forest is now highly fragmented due to deforestation (Armstrong, 2000). Accordingly, New Zealand robin (*Petroica australis*) populations have become patchily distributed, but can still be found in some native forest and exotic plantations throughout the country (Duncan *et al.*, 1999). North Island robins (*Petroica australis longipes*) are not yet endangered, and are often targeted for reintroduction in restoration programs as they are considered a "low-risk" species. However, there is still a poor understanding of the management required to restore robin populations in areas where extinction has occurred (Armstrong, 2000).

Food availability is often used as the primary factor for assessing habitat quality in studies of passerine birds (Riddington & Gosler, 1995; Seki & Takano, 1998). There is little doubt that food-supply could place an ultimate limit on the population growth of any bird species (Newton, 1980), and there is a large amount of research that gives evidence on food shortage reducing breeding success (Lack, 1954; Newton, 1980; Simons & Martin, 1990; Catchpole & Phillips, 1992; Wiehn & Korpimäki, 1997; Turner & McCarty, 1998). Low food supply has been found to shorten the breeding season (Dijkstra, 1982; Zanette *et al.*, 2000), reduce clutch size (Lack, 1947; Dijkstra, 1982), and cause a lower body mass of chicks (Quinney *et al.*, 1986; Richner, 1992; Bukaciñski *et al.*, 1998). This lower mass could result in poorer body condition, higher mortality during the first winter, and less recruitment into the population (Spaans *et al.*, 1987; Richner, 1992). These difficulties may be compounded because even if adequate food is available to juveniles during the fledgling-dependency period, they may starve because they are inefficient and inexperienced foragers (Sullivan, 1989). Fledglings in territories with a low food supply would have even more difficulty finding food, and would be more dependent on their parents (Seki & Takano, 1998). Parental feeding success can also be affected by the reduced food availability (Quinney & Smith, 1980).

Another factor likely to be important to the survival of a North Island robin population, and hence a key component of habitat quality, is predator abundance. North Island robins are vulnerable to predation by avian predators such as the morepork (*Ninox novaeseelandiae*) and exotic mammals including stoats (*Mustela erminea*), ship rats (*Rattus rattus*), Norway rats (*Rattus norvegicus*) and brushtail possums (*Trichosaurus vulpecula*) (Flack & Lloyd, 1978; Brown, 1997a; Powlesland, 1999). Predation is often found to be the primary cause of nest failure in avian studies (Hoover *et al.*, 1995; Ward & Kennedy, 1996; Brown *et al.*, 1998), and must therefore be an important limit to population productivity. Introduced predators such as those mentioned above also feed on invertebrates (Innes, 1979; Green, 1996; Brown *et al.*, 1993), and hence may have an indirect effect on robin breeding success by reducing their food supply.

This chapter aims to explain the poor viability of Paengaroa's robin population suggested in Chapter 4. I compare the habitat quality of Paengaroa with that of

Waimarino where robins are presumed to be viable, or have at least managed to persist longer. I assess habitat quality for both the native and pine forest at Waimarino as it is unknown whether the habitat quality of pine forest is enough to independently support a viable population of robins without immigration of individuals from adjacent native patches (source-sink dynamics). I use food and predator levels to assess habitat quality as these are obvious factors that may limit population viability. The vulnerability of North Island robin populations to predation is probably the main cause of population declines on the mainland. Thus, it seems sensible to get an estimate of predator abundance as a measure of habitat quality. Predator levels are assessed using tracking tunnels. As food supply is also known to be a highly important factor of habitat quality, prey biomass is estimated for the three sites using pitfall traps. I also study the foraging behaviour of North Island robins at the three sites to determine whether it is influenced by food supply.

5.2 METHODS

5.2.1 *Pitfall traps*

New Zealand robins obtain almost all of their food from ground level foraging (Powlesland, 1981a; McLean *et al.*, 1994; pers. obs.). Continuous-kill pitfall trapping is a technique that enables the monitoring of ground-based invertebrates (Green, 2000). This method was therefore used to assess the relative food supply for robins at Paengaroa and at both the pine and native forests at Waimarino. Ten pitfall traps were spaced at 100m intervals along a transect line at all three sites during October 1999. The pitfall trap design used for this monitoring can be seen in Figure 5.2.1. Holes were dug into the ground using a spade, with care taken to minimise any disturbance to the surrounding area as some invertebrates may be either attracted or repelled by this soil movement (Digweed *et al.*, 1995). The collection container was placed into the hole and soil was packed firmly so that the lip was at ground level. If the edge is too high the number of invertebrates able to fall in is reduced (Green, 2000). The container was then filled with 200 mL of antifreeze, and covered with the lid.

The traps were left in the ground over a 6 month period. They were cleared every 2 months (December 1999, February 2000, and April 2000) by emptying the contents of each trap into a plastic sealable container. The trap container was then refilled with antifreeze, and placed back into the ground. Care was taken to prevent spillages, minimise movements near the trap, and replace the trap cover in exactly the same position to avoid damage to the area. The sealable container was then clearly labelled with the site, date and pitfall trap number. Once all ten traps had been cleared from each site, the samples were taken to a laboratory. They were then drained through a 2mm sieve so only the very small invertebrates were lost from the sample. The samples contained a lot of soil and plant matter, so each sample was hand-sorted to remove all invertebrate bodies and appendages. These were placed in a watertight vial and preserved in 75% ethanol. Powlesland (1981a) noted in his observations of South Island robin (*Petroica australis australis*) foraging that few invertebrates were able to deter robin attacks, but that *Megadromous* and *Mecodema* species (Carabid beetles) were usually ignored, probably due to their pungent smell when disturbed (Sharell, 1971) and their strong exoskeletons. These species were therefore removed from the samples.

After I had sorted all of the samples, they were stored for a period in a laboratory. During this time they were moved by an unknown person, and 10 samples (Paengaroa, February-April) were lost. The samples from Waimarino over this time period are included in the analysis.

The remaining samples were drained and placed in a fumehood overnight to remove any ethanol residue. They were then moved into a drying oven at 80°C for 6 days. Each sample was weighed and its dry mass recorded.

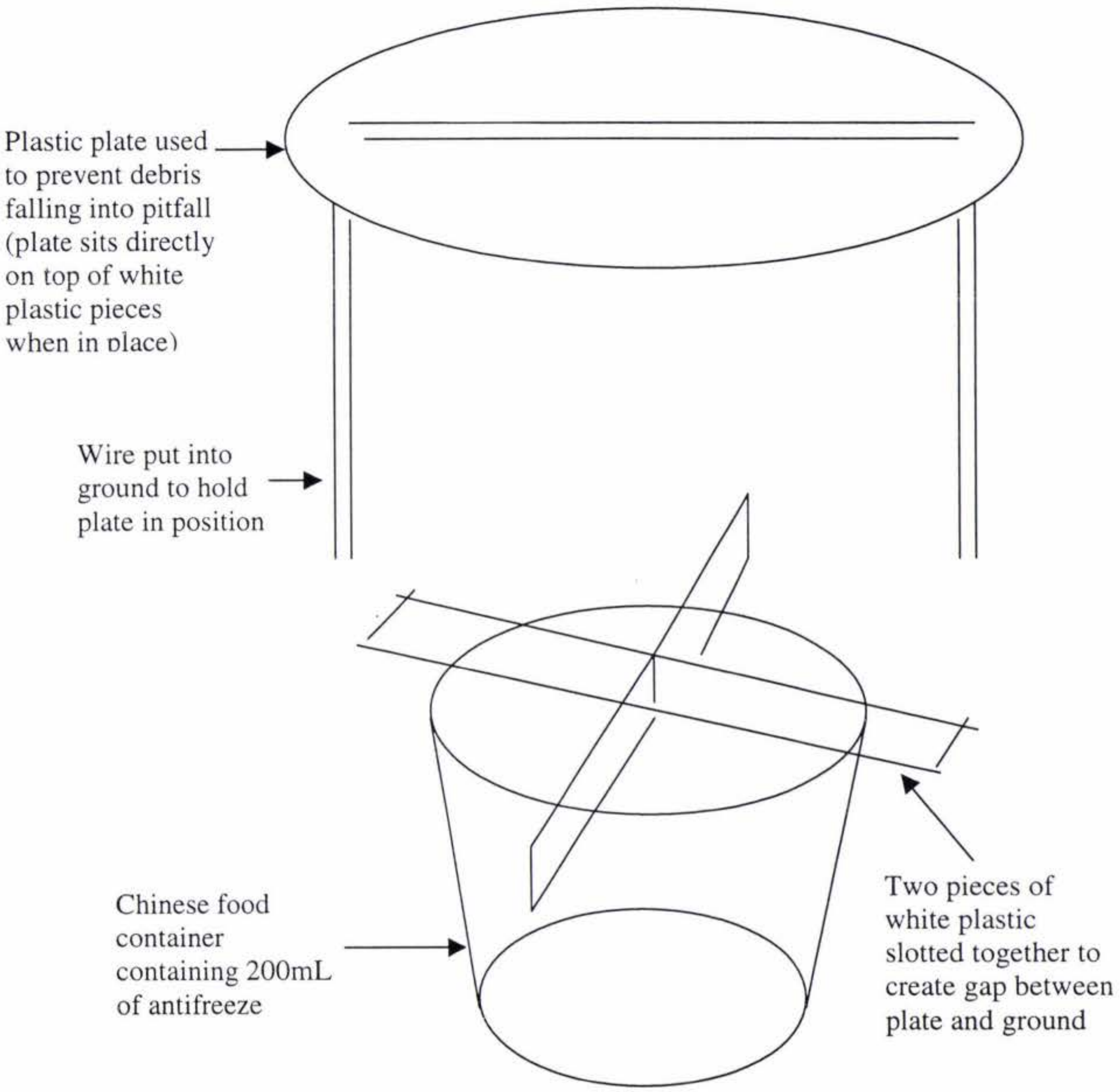

Figure 5.2.1. Pitfall trap design used to obtain measures of invertebrate biomass at Paengaroa Scenic Reserve, Wamarino native forest, and Waimarino pine forest.

As samples could not always be collected from the field at precisely the same number of days between sites and months, the results were standardised for 60 trap nights by dividing the dry weight by the actual number of trap nights, and multiplying the result by 60. Statistical analysis was performed at all three sites for each trapping period. The effect of time in the season (early = October 1999-December 1999, late = December 1999-February 2000) on biomass levels was examined over the three sites using a repeated measures ANOVA (Systat version 6). Repeated measures were incorporated as the samples were taken from pitfall traps that stayed in the same location for all three trapping periods (October-December, December-February, February-April). The output also included information on whether there was any difference in biomass levels between the three sites over the first two trapping periods. Another repeated measures ANOVA was run for all three trapping periods in Waimarino native and pine forests only (due to the missing data from Paengaroa) to see whether invertebrate levels varied between the two vegetation types over the course of sampling.

5.2.2 *Foraging behaviour*

The foraging activity of 5 male robins at each site was recorded for the December to February pitfall trapping period. Observations were made at approximately the same time each day (as close to mid-day as possible) to avoid any possible bias caused by daily variations in foraging patterns (e.g. Powlesland, 1981b).

I considered robins to be foraging if they were killing prey, eating prey, caching prey, or appeared to be searching for prey, similar to Powlesland (1981b). Searching for prey often consisted of the robins vibrating one or both feet on the ground. The behaviour was noted by Flack (1973), and it is thought that this "foot-trembling" may stimulate movement from hidden prey (Powlesland, 1981a). Powlesland (1981a) also noted that South Island robins (*Petroica australis australis*) seem to rely mainly on sight for searching, by hopping over litter or vegetation, or scanning from a vantage point. Small prey were eaten immediately after capture. Larger prey were often killed and broken into smaller pieces by swinging them against logs or the ground. Food was sometimes killed and then carried away to a cache site (often in the apex of branches). This behaviour was termed caching.

Once a male robin was located, I observed him quietly for 10 minutes to get him accustomed to my presence. The bird would initially focus its attention on me, but would appear to ignore me after a couple of minutes. Once 10 minutes elapsed I then watched the robin continually for 30 minutes. During this period I used a stop-watch to measure the amount of time the bird spent foraging. I stopped timing if I lost sight of the bird, and started again once he was resighted.

For some bird species, seasonal changes in feeding routine are closely associated with seasonal changes in food supply. However, it is often unknown whether changes in foraging time indicate food shortage, as there is no information on whether birds succeed or fail in their attempts to find food (Newton, 1980). I wanted to discover whether the availability of food had any effect on how successfully the birds foraged. Hence, I also noted the number of foraging attempts made by each bird, and whether the attempt was successful or unsuccessful. A foraging attempt was classed as any obvious movement towards an area in search of prey (sitting on a perch and scanning for prey [Powlesland 1981a] was not included as a foraging attempt). For example, an attempt could involve a robin flying down to ground level, then foot-trembling and examining the leaf litter, or flying onto a tree trunk and searching that trunk for prey. The end of a foraging attempt was marked by the bird moving away from the localised area (generally around $1m^2$) that it had been searching. An attempt was successful if the bird managed to catch a prey item. If more than one prey item was attained in the area, this was still classed as a single successful foraging attempt. An attempt was unsuccessful if a movement to the area resulted in no prey being caught.

The proportion of time spent foraging was calculated by dividing the number of minutes spent foraging by the total observation time (30 minutes). The number of attempts per minute of foraging time was also calculated, as was the proportion of those foraging attempts that were successful.

Supplementary feeding has been found to cause birds to adjust their foraging behaviour to the food supply (Wiehn & Korpimäki, 1997), so I supplied as few mealworms as possible to each pair during the course of the study.

5.2.3 *Tracking tunnels*

Tracking tunnels were set for rodents and mustelids using the methods described in Chapter 2. The tunnels were run every two months from December 1998 to February 2000 in Paengaroa. I took over predator monitoring from the DoC contract workers after January 1999. I ran tracking tunnels at Waimarino from December 1998 to October 1999. I intended to continue the monitoring at Waimarino until February 2000, but abandoned it in November 1999 for safety reasons.[1]

5.2.4 *Monitoring breeding success*

Robins at Paengaroa were monitored throughout the 1999/00 and 2000/01 breeding seasons, as described in Chapter 3. North Island robin pairs at Waimarino were also monitored during the 1999/00 breeding season. Sixteen pairs (8 in native forest, 8 in pine forest) were checked weekly (over a period of 2 days) throughout the breeding season. The same methods were used at each site. Once a robin was sighted, mealworms were fed to it, and the bird was followed in an effort to locate the nest site. Unfortunately, due to the largely impassible nature of the terrain at Waimarino (steep gullies and dense vines), nests could not be found except on four occasions. The birds were also unbanded, which meant that it was often unclear whether the bird being observed was male or female, or even if they were part of the pair that I was wanting to observe. These limiting factors meant that insufficient data were obtained to estimate nest success at Waimarino.

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[1] A friend came out to Waimarino in November to help me set the tunnels for rodent monitoring. She had the misfortune of disturbing a bees nest on one of the lines and was stung many times. We had to go directly to Taihape Hospital. As I am personally quite allergic to bees, I was unable to complete the predator monitoring at Waimarino due to the health risk.

5.3 RESULTS

5.3.1 *Pitfall traps*

There was a very significant increase in biomass levels between the early (October-December) and late (December-February) parts of the breeding season ($p = 0.0002$). At all 3 sites the mean biomass increased by at least 140% (Table 5.3.1). The biomass at Waimarino dropped again after the breeding season (February 2000-April 2000), and there was a significant difference in biomass between the three sampling periods ($p = 0.007$).

Table 5.3.1. Mean (and 95% Confidence Limits) of invertebrate dry weight (g) obtained from Paengaroa and Waimarino native and pine forests*.

Site	Oct '99-Dec '99	Dec '99-Feb '00	Feb '00-Apr '00
Paengaroa	0.32 (0.18-0.46)	0.76 (0.37-1.15)	
Waimarino Native	0.25 (0.16-0.33)	0.63 (0.33-0.93)	0.42 (0.15-0.70)
Waimarino Pine	0.18 (0.08-0.28)	0.55 (0.21-0.89)	0.35 (0.52-0.17)

* The data from Paengaroa for the February to April period are missing due to lost samples (refer to Methods section, 5.2.1).

Mean biomass at Paengaroa was consistently higher than that at both Waimarino sites, and Waimarino's native forest had a consistently higher mean biomass than the pine forest (Table 5.1). However, the difference among the three sites was not statistically significant ($p = 0.385$).

5.3.2 *Foraging behaviour*

The proportions of time male robins spent foraging were not significantly different for the three sites ($p = 0.187$). There were fairly big differences between the means, with birds spending the least time foraging in the Waimarino pine forest (Figure 5.3.1). However the sample sizes (5 birds) per site were too small for this to be significant.

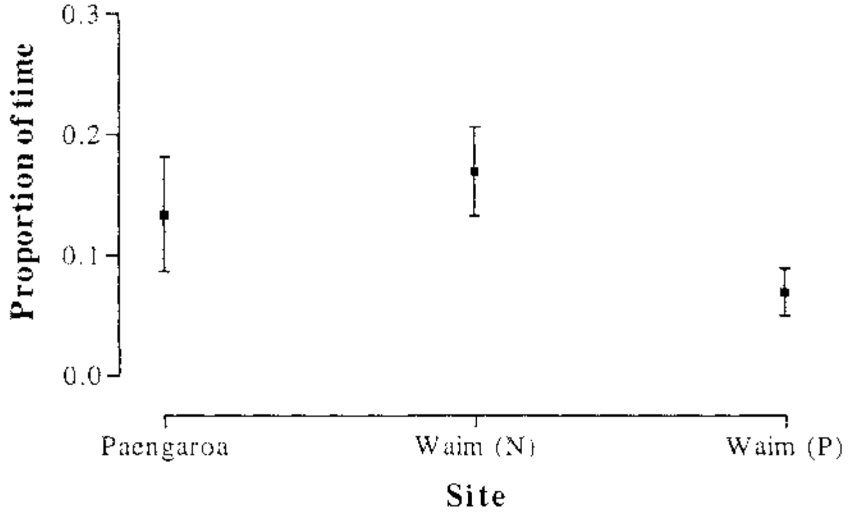

Figure 5.3.1. Proportion of time spent foraging by male North Island robins at Paengaroa, and Waimarino pine and native forests.

The number of foraging attempts per minute showed a difference between sites that was close to significance (p = 0.062), with fewer foraging attempts at Paengaroa than the two Waimarino sites (Figure 5.3.2). The proportion of successful foraging attempts at the 3 sites was also close to a significant difference (p = 0.084), with the highest proportion found at Paengaroa (Figure 5.3.3). Differences in mean values between sites were fairly large (Figures 5.3.2 & 5.3.3). The low power of these statistical tests was due to small sample sizes. The higher proportion of successful foraging attempts and fewer attempts per minute at Paengaroa both suggest more efficient feeding at this site.

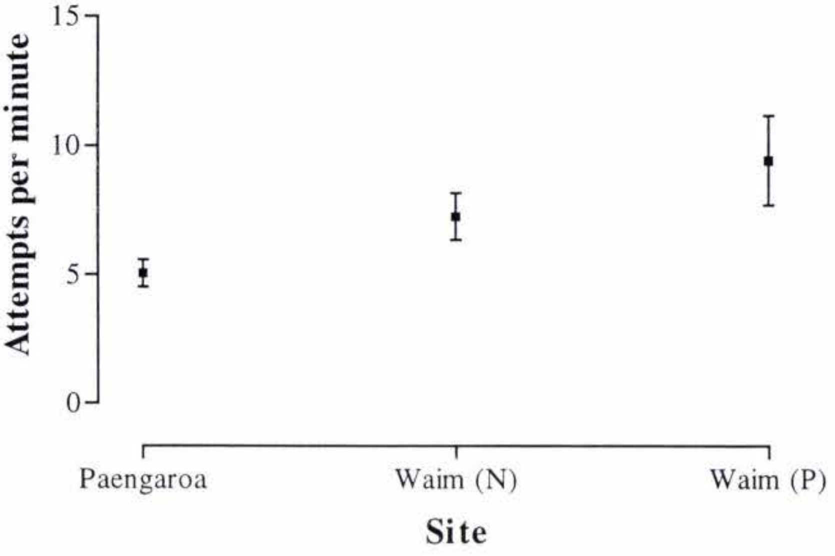

Figure 5.3.2. Number of foraging attempts per minute of time spent foraging by male robins at Paengaroa, and Waimarino pine and native forests.

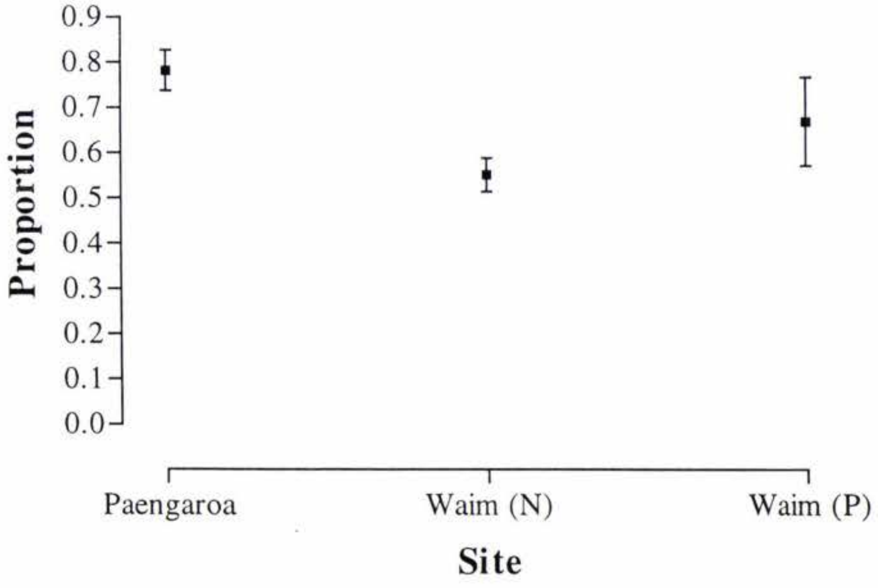

Figure 5.3.3. Proportion of foraging attempts that were successful at Paengaroa, and Waimarino pine and native forests.

5.3.3 *Tracking tunnels*

Rat levels were highest in Waimarino native forest in both peanut butter and meat baited tunnels (Figures 5.3.4 and 5.3.6 respectively), with Waimarino pine forest and Paengaroa having fairly similar levels. Tunnels set with peanut butter bait showed a higher rat tracking rate in the pine forest (except during June 1999). However, tunnels baited with meat suggested that levels were sometimes higher in Paengaroa (April 1999, August 1999), and sometimes higher in pine (December 1998, October 1999).

Mustelid levels were low at all three sites when using tunnels baited with peanut butter. Meat baited tunnels suggested that Paengaroa has consistently low mustelid levels (maximum tracking rate was 9% in April 1999), whereas both Waimarino native and pine had highly variable levels. In the pine forest, the lowest rate was measured to be 10% in April 1999, and the highest was 71% in August 1999. The lowest tracking rate (8%) in Waimarino native forest was in February 1999, and the highest (70%) in October 1999.

Mouse levels at Paengaroa were always much higher than at both Waimarino sites throughout the course of the study for both bait types. There was no sign of mouse activity in Waimarino native forest. The pine forest had almost no mouse tracking except for low levels in June 1999 (12%) and August 1999 (18%), when some tracks occurred in the tunnels baited with peanut butter.

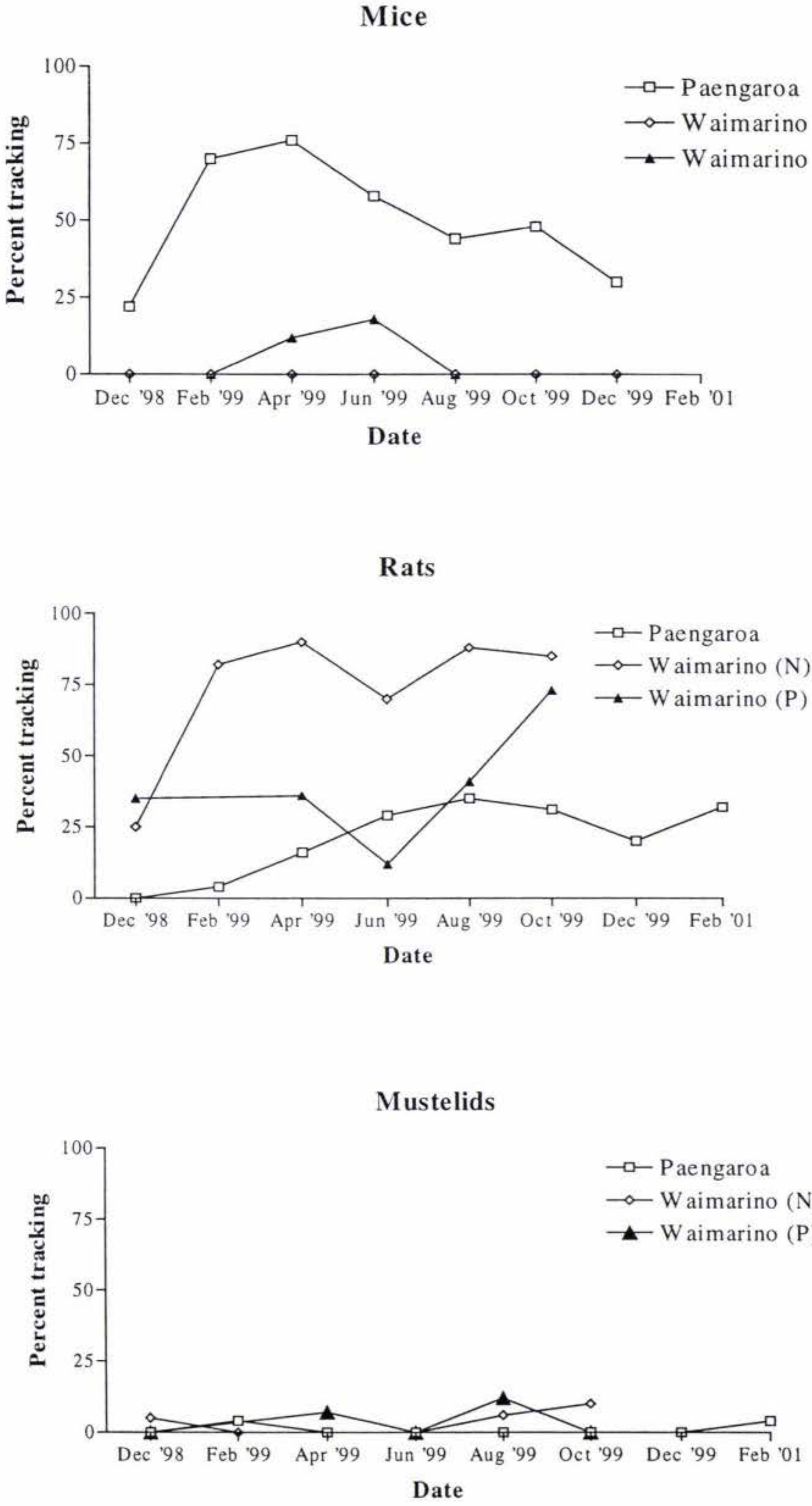

Figure 5.3.4. Predator indices using tunnels baited with peanut butter in Paengaroa, and Waimarino pine and native forests.

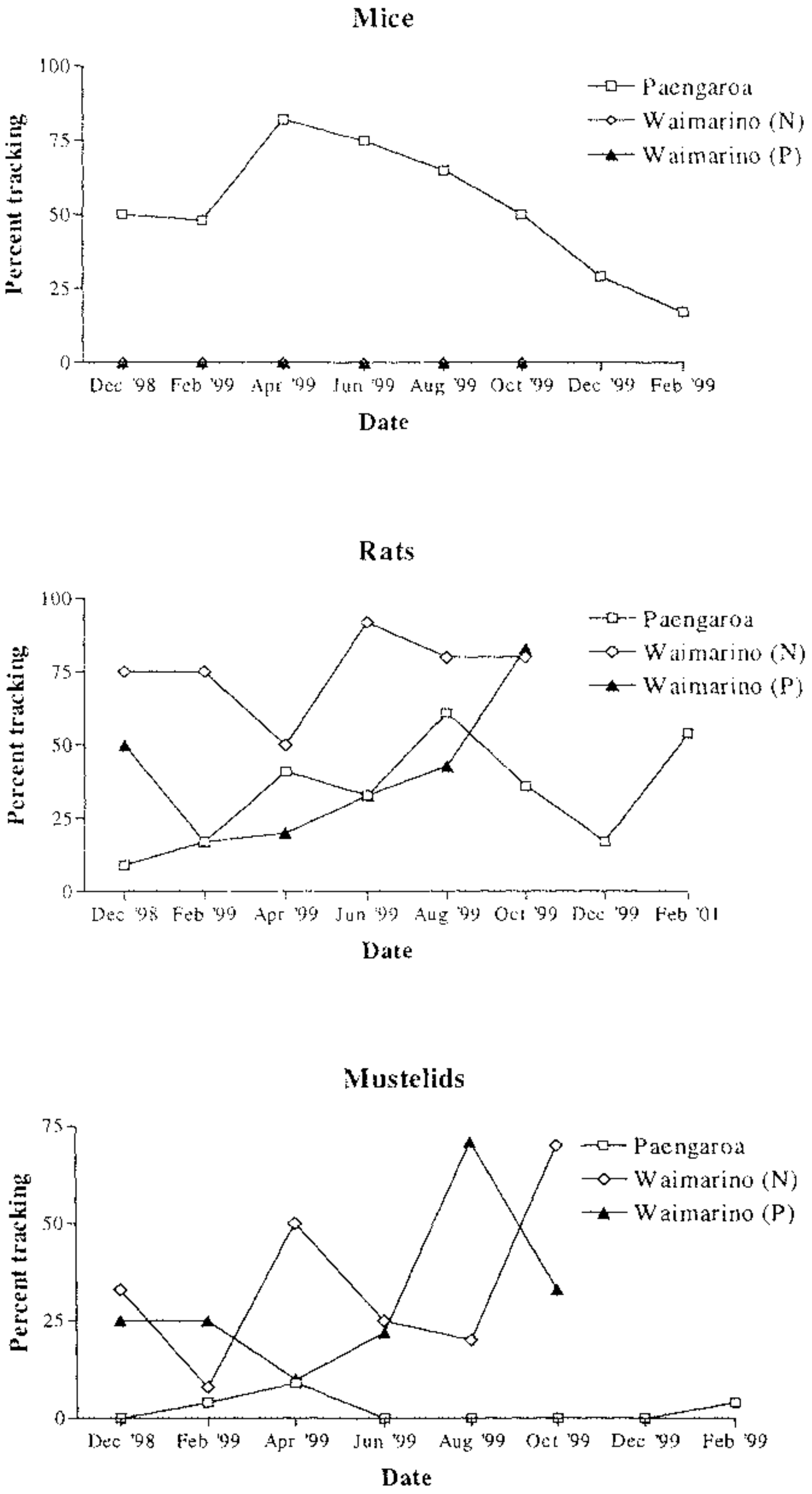

Figure 5.3.5. Predator indices using tunnels baited with meat in Paengaroa, and Waimarino pine and native forests.

5.4 **DISCUSSION**

The estimation of habitat quality with regard to species requirements is of great importance when planning a reintroduction. Pre-translocation predator levels were compared for Paengaroa Scenic Reserve, Boundary Stream Scenic Reserve, and Tiritiri Matangi Island in Chapter 2. This study took place post-translocation in an effort to improve the understanding of habitat features influencing North Island robin populations. The two factors thought to be primarily affecting habitat quality for robins were food supply and predator abundance. These factors were quantified for a pine and native forest where robins are known to persist, as well as Paengaroa Scenic Reserve where robins were released as part of a restoration program.

Data collected to assess food supply and predator levels provide no indication of why robins have survived at Waimarino, but previously disappeared from Paengaroa. The reintroduced robin population at Paengaroa still appears non-viable despite predator control. While statistical power is low, all the data suggest that Paengaroa is the best site, with the highest food supply, most efficient foraging, and fewer rats and stoats than at Waimarino. Thus, the underlying reasons for the suggested low population viability at Paengaroa are still unclear.

Measurements of food supply are difficult to obtain (Quinney *et al.*, 1986). There has been a growing realisation of the importance of quality as opposed to quantity of food. However, this is difficult to assess. There is always the possibility that when favoured foods are finished, birds may turn to alternatives that may have been previously unknown to the observer (Newton, 1980). Due to these difficulties, general measures of biomass from pitfall trap catches were used as an estimate of food supply. Pitfall traps are essentially a measure of invertebrate activity as they rely on the invertebrates moving to the trap in order to fall into it. As a result, the more active species (most of which are nocturnal) tend to have a higher chance of being caught than less mobile species (Green, 1996). North Island robins are diurnal foragers. This means that the invertebrates caught using pitfall traps may not be representative of the prey abundance available for robins. However, the active species mentioned by Green

(1996) include spiders, earthworms and wetas, all of which were observed being consumed by robins during the study. This suggests that although these invertebrates may not be active during the day, the foraging techniques used by robins enabled the capture of nocturnal species. The catch of pitfall traps is also readily influenced by invertebrate activity changes caused by weather (Thomas & Sleeper, 1977; Green, 2000). This type of problem could not be avoided. However, every effort was made to cater for preventable biases. For instance, traps were collected at approximately the same time between sites (Thomas & Sleeper, 1977), and the location of each trap remained the same between sampling occasions to remove all variables apart from site (Green, 2000).

The large increase in biomass levels between the early and late parts of the season was found at all 3 sites (Paengaroa, Waimarino native, Waimarino pine). There was no significant difference in biomass levels between the sites during the course of the study, but mean biomass was consistently lowest in the pine forest. The lower biomass level in the pine forest was also found by Borkin *et al.* (unpubl. data), where lower invertebrate biomass was present in pine plantations than in both Douglas Fir and native forest sites (Duncan *et al.*, 1999). The increase in biomass through the breeding season in the present study could be important to the reproductive success of robins, as late in the season more pairs will be caring for large nestlings or fledglings, both of which have a very high energy demand (Bukaciñski *et al.*, 1998).

Measurements of the proportion of time spent foraging are often used to define when birds spend the most time feeding, and thus infer when they have the most difficulty meeting their food requirements (Newton, 1980). The findings of this study do not necessarily support this inference. Food levels seemed lowest in the pine forest, but birds in that vegetation type also appeared to spend the least time foraging. These results were not statistically significant, but suggest that either higher measures of biomass do not necessarily correspond with the birds meeting their food requirements more easily, or that studying the proportion of time spent foraging can be misleading when estimating food availability.

Variations in habitat quality or foraging efficiency both affect parental energy expenditure (Tolonen & Korpimäki, 1994, 1996). Thus, when considering breeding performance, it is often difficult to distinguish between inputs made by parental quality from those made by territory quality (Seki & Takano, 1998). Unfortunately, the breeding performance of birds at Waimarino could not be assessed. During the course of this study I did, however, notice that there was an apparent difference in density of birds between the two Waimarino sites. Both before and after the translocation, Waimarino pine forest seemed to have a low density of robins as there were few territorial calls heard, and there was a poor response to the tape-recorded call. This could have been either because there were unoccupied areas in this forest type, or because the territories were large and the robins did not respond unless they were in close proximity to a call. In comparison, regular unsolicited territorial calls were heard in Waimarino native forest, with a rapid response time to the recorded call. These behaviours suggested a higher density of robins in this site than in the pine forest (see also Duncan *et al.*, 1999). Paengaroa has large areas of bush unoccupied by robins. In a future study it would be interesting to resample invertebrate biomass within each robin territory, and in other randomly chosen sites throughout the reserve, to determine whether robins demonstrate preferential selection for territories rich in food (e.g. Burke & Nol, 1998, Wiles, 2000).

It has been suggested that the density of insectivorous bird populations are often correlated to similar variations in insect populations (Newton, 1980). As the food supply is relatively similar at the three sites, any large difference in density found in this study would be in contradiction to this. This suggests that densities at these sites are affected by factors other than resource abundance.

Duncan *et al.* (1999) suggest that the low density of South Island robins (*Petroica australis australis*) in pine plantations was due to low invertebrate biomass in this habitat type relative to native forest and Douglas Fir stands. However, accurate estimates of density are hard to obtain (Van Horne, 1983) so no attempt was made to quantify the densities of birds at Waimarino. Density data may also be of limited use as densities may reflect conditions in the recent past rather than long-term habitat quality (Van Horne, 1983). Van Horne (1983) cautions against the assumption that the density of a species in a habitat is a direct measure of the quality of that habitat, and some recent studies have found that the higher abundances of birds may occur in poor-quality sink habitat (e.g. Zanette *et al.*, 2000).

New Zealand robins are vulnerable to predation by introduced mammals, and this is likely to be the primary cause of population declines on the mainland. If the observed predator tracking indices are assumed to reflect actual abundance, the generally higher rat and mustelid levels at both Waimarino sites would suggest a higher density of these predators than in Paengaroa. The significantly higher mouse tracking rate at Paengaroa may be due to the lack of competition for food (King, 1982), or reduced predation caused by lower rat and mustelid levels (Innes *et al.*, 1995). The rat levels at Paengaroa and Waimarino pine forest were quite similar throughout the course of the study, which suggests that the negligible mouse tracking in Waimarino pine is not caused by rat levels, but either higher mustelid levels (e.g. Moors, 1983) or some unknown factor. King *et al.* (1996) found that mice and stoats were most abundant in pine forest and least abundant in unlogged native forest, whereas ship rats were most abundant in the native forest type. This largely contradicts the results from my study, and demonstrates that general statements about predator abundance for certain habitat types cannot be made.

The use of tracking tunnels to compare predator levels between sites assumes equal trackability in the different habitats. Competitors and predators may confound the tracking tunnel indices by altering the response of another species to them. For example, it is thought that ship rats may limit mouse use of tunnels by scent marking (Brown *et al.*, 1996). Calibration of stoat indices may be affected by rat and mouse levels, as when rodents are scarce, stoats may enter baited tracking tunnels more readily out of hunger (Alterio *et al.*, 1999). This could lead to lower mustelid tracking indices at sites with high mouse levels such as Paengaroa. It is unlikely that this occurred in my study, but it is important to be aware such confounding factors when drawing conclusions about predator abundance from tracking tunnel data. Direct links between predator abundance and predation rates must also be treated with caution. It has been shown that although ship rats are a major component of the diet of stoats, when rat levels drop after a poison operation, stoats will readily switch to birds as their main prey type (Murphy & Bradfield, 1992). Hence, although habitats with low rat abundance may initially seem of higher quality than habitats with high rat abundance, the overall effect on bird species is linked to other factors, perhaps not obvious from the outset. There has been increasing research in recent years aimed at improving the understanding of these complex relationships. This is crucial to estimate the effects of predators on New Zealand's native birds.

As predators are known to feed on invertebrates (Green, 1996), it is interesting to note that despite differences in predator abundance between sites, the invertebrate biomass did not correspond directly with predator levels. It is likely, however, that if the invertebrates had been individually identified, there would have been some difference in species composition caused by differing predation pressure at the 3 sites.

Interactions between nest predation and food supply have been suggested in the findings of some studies. For example, female Northern Goshawks (*Accipter gentilis*) have been found to leave their nest more regularly when there are low food levels, which leads to a higher risk of nest predation as the nest is left unattended (Ward & Kennedy, 1996). It is possible that robin nests would be affected by the lack of protection caused by the female leaving the nest site to forage. Female robins do not leave the nest to forage very often, and probably only do so if their male partners are unable to find enough food. Stoats are active during daylight hours (King, 1995), and it has been suggested that they may use cues such as parental feeding visits to locate nests (Dunn, 1977). Another link has been found between begging and predation risk. Begging signals of offspring tend to be very conspicuous (Godfray, 1991). Food-deprived chicks are shown to beg more (Price & Ydenberg, 1995), and that this begging could attract predators (Leech and Leonard, 1997). G. Sherley (pers. comm.) observed a rifleman's (*Acanthisitta chloris*) nest being preyed upon by a stoat, which was apparently attracted by the loud calls of the nestlings (Moors, 1983). North Island robin chicks beg to solicit food from their parents, and this begging becomes very audible (from about 50m, pers. obs.) as the chicks near fledging. It is possible that this behaviour does increase the predation risk of robin nests at the nestling stage, but it is unknown whether food supply has any affect on begging intensity in robin chicks.

The lack of any obvious evidence from this study that food or predator levels have a strong influence on population parameters does not mean that these factors are unimportant to robin survival. The estimation of habitat quality using food supply is only useful if there are no other factors that will keep the population at levels below those where it would be constrained by food. Predator levels are a constant threat to mainland populations. However, it seems that at some sites high predator levels do not prevent robin numbers from becoming apparently abundant (e.g. Waimarino native forest).

This poor understanding of why robins are present and apparently thriving in some mainland areas and not in others, without any obvious difference in habitat quality (Armstrong, 2000), suggests that there may be one or more hidden factors crucial to robin survival. It is important to discover and increase our understanding of these underlying influences to enable adequate protection of existing populations, and ensure the success of future robin translocations.

CHAPTER SIX

General Discussion

Male North Island robin in Paengaroa Scenic Reserve.

The North Island robin (*Petroica australis longipes*) reintroduction to Paengaroa Scenic Reserve was the first of its kind. Previously, successful robin reintroductions were to predator-free offshore islands, or to large areas of mainland habitat with intensive predator control. It was therefore unknown what the minimum habitat requirements were to establish a mainland population of North Island robins. With this study I wanted to find out whether the robin population released into Paengaroa is sustainable.

The estimated viability of the robin population at Paengaroa was very low. The population was estimated to have a 17% probability of surviving over 10 years if parameters remain the same as in the first two years. The population viability analysis excluding data from the first year after reintroduction predicted the population to have a 100% probability of surviving 100 years. It is unknown whether lower survival and fecundity in the first year were due to factors that will continue to act upon the population in future years, or whether it was the result of problems faced only in the first year during establishment in a new site. If the latter is true, then the population may already be highly viable. However, it is unreasonable to exclude the first year's data from my analysis at this point, as it is unclear whether parameter estimates from the first year were unusually low. These results highlight the uncertainty that is associated with having only short-term data available. Another source of uncertainty is demonstrated by the large 95% confidence intervals associated with obtaining population viability estimates for a small population.

The parameters influencing population viability are initial population size, fecundity, and survival. It is therefore necessary to correctly identify which factors are affecting these three parameters so that the underlying causes of high or low viability can be understood.

Reintroductions of birds and mammals have tended to be more successful where large numbers have been released (Griffith *et al.*, 1989). The main reason for this may be that many animals die or disperse shortly after release, so that the founder group is much smaller than the number released. The loss of 45% of the birds released into the reserve within the first two months after translocation meant that the founder group was smaller than anticipated, and the continuing decline in numbers since release has lead to the currently small population size. Paengaroa did have low post-translocation survival in comparison to Boundary Stream Scenic Reserve and Tiritiri Matangi Island. Predation is often the main cause of mortality after translocation (Wilson *et al.*, 1992; Van Vuren *et al.*, 1997). Therefore, the lower post-translocation survival rate at Paengaroa could have been due to the higher predator levels at this site. Another possible explanation is the increased capacity for dispersal out of Paengaroa Scenic Reserve due to its small relative size and lack of isolation from surrounding bush remnants.

Increased recruitment of individuals into the robin population is essential to increase population viability. Recruitment into the Paengaroa population depends on fecundity, which is largely determined by nest success. It is therefore important to know which factors have the most effect on nest success. Despite increasing awareness of the importance of obtaining accurate estimates of nesting success (Willis, 1973; Dixon, 1978; Miller & Johnson, 1978; Johnson, 1979; Green, 1989; Johnson & Shaffer, 1990; Rotella *et al.*, 2000; Stanley, 2000), many avian population studies continue to calculate nesting success using the traditional method. My analysis showed nest success estimates calculated using the traditional method have a large positive bias. Stanley's (2000) method has advantages over the Mayfield method as it enables a greater appreciation of the factors affecting nest success. Using Stanley's program, I found that nesting stage and laying date were both affecting daily survival rates of nests at Paengaroa. Nests at the incubation stage that were laid early in the breeding season had the lowest survival rates. This suggests that nests may be most vulnerable to predation under these circumstances, and has important management implications. If attempts to improve nest success are made (such as poison bait or traps around the base of nest trees), these could be concentrated in the early half of the season, reducing both labour and cost.

The most important influence on nest success appears to be predator abundance. Paengaroa had higher overall predator levels and lower nest success than Boundary Stream (25% nest success compared to 47% nest success at Boundary Stream). Nest success on Tiritiri Matangi using data from the first 5 years after release (Armstrong *et al.*, in press) is 37%. It is interesting that nest success at Tiritiri Matangi Island is lower than at Boundary Stream, and could be a result of higher avian predation pressure on the island (Armstrong *et al.*, 2000).

In addition to nest success, the survival of individuals is important to population viability. The most important influence on survival of robins at Paengaroa was whether a bird was recently-translocated, a juvenile, or an adult. There were similarly low survival rates for recently-translocated individuals and juveniles, which suggests that their low survival may be due to problems faced when finding and establishing a territory. Adult survival in the second year (85%) was higher than the average rate at Tiritiri Matangi (79%; Armstrong & Ewen, in press), whereas survival was much lower in the first year (54%). It therefore remains to be seen whether survival needs to be enhanced by management. It is even more difficult to assess whether juvenile survival is limiting, as only one year of data are available at the time of writing.

The combination of small initial population size, low nest success, and low survival rates has lead to the low viability currently predicted for the robin population at Paengaroa. Habitat quality is important in determining the success of a translocation (Griffith *et al.*, 1989). It is important to understand which habitat features might be limiting viability, both to quide future management at Paengaroa and select sites for reintroduction. Predator levels were much higher in Waimarino's native and pine forest than in Paengaroa. Invertebrate levels were found to be relatively similar among sites, but slightly higher in the native forest types (Waimarino native and Paengaroa) than in the pine forest. However, birds in pine forest spent the least amount of time foraging. This suggests that either higher levels of biomass don't correspond with birds meeting their food requirements, or that studying the proportion of time spent foraging is misleading when estimating food availability. There was a large increase in biomass through the breeding season, which could be an underlying cause of the increased nest success also found through the breeding season. Supplementary feeding has been used to increase the breeding success of many avian populations (Dijkstra *et al.*, 1982; Simons & Martin, 1990; Richner, 1992; Ward & Kennedy, 1996; Wiehn & Korpimäki, 1997; Bukacinski *et al.*, 1998). If further research suggests increased food supply improves nest success, supplementary feeding could be introduced to reduce levels of nest failure in robin populations. Data on food and predator levels provided no indication of why robins have persisted at

Waimarino but disappeared from Paengaroa, and currently appear non-viable despite predator control.

New Zealand robins are vulnerable to predation by introduced mammals (Flack & Lloyd, 1978; Brown, 1997a; Powelsland *et al.*, 1999), and this may be the primary cause of population decline on the mainland. However, it seems that high predator levels do not prevent robin populations from persisting and apparently thriving at some mainland sites. This suggests that there may be one or more hidden factors crucial to robin survival. More research must go into determining what these factors are, and how they act upon robin populations if the success of future robin reintroductions is to be ensured.

The apparent changes in predator levels through the breeding season at Paengaroa and Boundary Stream did not correspond to changes in nest success. This suggests that tracking tunnels either do not pick up changes in abundance or that predation pressure is not determined by abundance. If tracking tunnels are not picking up changes in abundance, it seems important to develop an accurate way of doing this since approval for the reintroduction of robins to Paengaroa was entirely contingent on lower rodent and mustelid tracking rates at that site than the proposed source site. It is therefore important to discover if predation pressure is determined by abundance. This could be achieved by setting up infa-red video equipment on robin nests at the same time a predator monitoring program is being run to accurately identify nest predators, and to see whether predator tracking indices reflect the type of predation occurring. Traditional methods of identifying nest predators (Flack & Lloyd, 1978; Moors, 1978) from nest remains can be misleading (Brown *et al.*, 1998), causing inaccurate conclusions to be drawn about which predator is the most important for a given population. This would be an important step towards understanding how much weight should be given to predator tracking levels when assessing habitat quality for a target species. The results from my study also suggest that other factors such as food supply may affect breeding success and should be taken into account when assessing the suitability of release sites.

There have been many robin translocations to managed mainland habitats over the last few years. However, it is obvious that there is still a lack of information available about what factors determine whether a translocation will be a success or a failure. My study was the first North Island robin mainland reintroduction with intensive monitoring post-release. It has established a methodology that can be used for follow-up study of future robin reintroductions. If similar monitoring techniques are used for these translocations, comprehensive information will become available through time about what habitat features are necessary to ensure robin reintroductions to mainland sites are successful in the future.

REFERENCES

Alterio, N., Moller, H., and Brown, K. 1999. Trappability and densities of stoats (*Mustela erminea*) and ship rats (*Rattus rattus*) in a South Island *Nothofagus* forest, New Zealand. *New Zealand Journal of Ecology 23*: 95-100.

Ambrose, H.W.III. 1972. Effect of habitat familiarity and toe-clipping on rate of owl predation in *Microtus pennsylvanicus. Journal of Mammalogy 53*: 909-912.

Anderson, D.R., Burnham, K.P., and White, G.C. 1994. AIC model selection in overdispersed capture-recapture data. *Ecology 75*: 1780-1793.

Armstrong, D.P. 1995. Effects of familiarity on the outcome of translocations, II. A test using New Zealand robins. *Biological Conservation 71*: 281-288.

Armstrong, D.P. 2000. Re-introductions of New Zealand robins: a key component of ecological restoration. *Reintroduction News 19*: 44-47.

Armstrong, D.P., and Ewen, J.G. 2001. Assessing the value of follow-up translocations: a case study using New Zealand robins. *Biological Conservation 101*: 239-247.

Armstrong, D.P., and Ewen, J.G. (in press). Dynamics of a New Zealand robin population reintroduced to regenerating fragmented habitat. *Conservation Biology.*

Armstrong, D.P., and McLean, I.G. 1995. New Zealand translocations: theory and practice. *Pacific Conservation Biology* 2: 39-54.

Armstrong, D.P., and Perrot, J.K. 2000. An experiment testing whether condition and survival are limited by food supply in a reintroduced hihi population. *Conservation Biology 14*: 1171-1181.

Armstrong, D.P., Ewen, J.G., Dimond, W.J., Lovegrove, T.G., Bergström, A., and Walter, B. 2000. Breeding biology of North Island robins (*Petroica australis longipes*) on Tiritiri Matangi Island, Hauraki Gulf, New Zealand. *Notornis 47*: 106-118.

Armstrong, D.P., Raeburn, E.H., Powlesland, R.G., Howard, M., Christensen, B., and Ewen, J.G. (in press). Obtaining meaningful comparisons of nest success: data from New Zealand robin (*Petroica australis*) populations.

Atkinson, I.A.E., and Cameron, E.K. 1993. Human influence on the terrestrial biota and biotic communities of New Zealand. *Trends in Ecology and Evolution 8*: 447-451.

Barkla, J. 1996. Paengaroa Scenic Reserve: a strategy for management as a "mainland island" 1996-2001. Unpublished report. Department of Conservation, Wanganui Conservancy.

Bell, B.D. 1978. The big South Cape Islands rat irruption. Pp. 33-40 *in* The Ecology and Control of Rodents in New Zealand Nature Reserves. Dingwall, P.R., Atkinson, I.A.E. and Hay, C. (Ed.s). Information Series No. 4. Department of Lands and Survey, Wellington, New Zealand.

Bell, B.D. 1986. The conservation status of New Zealand Wildlife. *Occasional Publication No. 12*. New Zealand Wildlife Service, Department of Internal Affairs.

Brown, J.A., and Miller, C.J. 1998. Monitoring Stoat (*Mustela erminea*) Control Operations: Power Analysis and Design. Department of Conservation, Wellington, New Zealand.

Brown, K.P. 1994. Predation at North Island robin and North Island tomtit nests. Unpublished MSc thesis. University of Otago, Dunedin, New Zealand.

Brown, K.P. 1997a. Impact of brodifacoum poisoning operations on South Island robins *Petroica australis australis* in a New Zealand *Nothofagus* forest. *Bird Conseration International 7*: 399-407.

Brown, K.P. 1997b. Predation at nests of two endemic passerines; implications for bird community restoration. *Pacific Conservation Biology 3*: 91-98.

Brown, K.P., Innes, J., and Shorten, R.M. 1993. Evidence that possums prey on and scavenge birds' eggs, birds and mammals. *Notornis 40*: 1-9.

Brown, K.P., Moller, H., Innes, J., and Alterio, N. 1996. Calibration of tunnel tracking rates to estimate relative abundance of ship rats (*Rattus rattus*) and mice (*Mus musculus*) in a New Zealand forest. *New Zealand Journal of Ecology 20*: 271-275.

Brown, K.P., Moller, H., Innes, J., and Jansen, P. 1998. Identifying predators at nests of small birds in a New Zealand forest. *Ibis 140*: 274-279.

Bukaciñski, D., Bukaciñska, M., and Spaans, A.L. 1998. Experimental evidence for the relationship between food supply, parental effort and chick survival in the lesser black-backed gull *Larus fuscus*. *Ibis 140*: 422-430.

Burke, D.M., and Nol, E. 1998. Influence of food abundance, nest-site habitat, and forest fragmentation on breeding ovenbirds. *The Auk 115*: 96-104.

Burnham., D.P., and Anderson, D.R. 1998. Model Selection and Inference. Springer-Verlag, New York, U.S.A.

Burnham, K.P., Anderson, D.R., White, G.C., Brownie, C., and Pollock, K.H. 1987. Design and analysis methods for fish survival experiments based on release-recapture. *American Fisheries Society Monograph 5.*

Butler, D., and Merton, D. 1992. The Black Robin: Saving the World's Most Endangered Bird. Oxford University Press, Auckland.

Catchpole, C.K., and Phillips, J.F. 1992. Territory quality and reproductive success in the Dartford warbler *Sylvia undata* in Dorset, England. *Biological Conservation 61*: 209-215.

Clout, M. 1997. Predator management in New Zealand: an overview. Pp. 3-5 *in* National Predator Management Workshop 1997. Sim, J. and Saunders, A. (Ed.s). Department of Conservation, Wellington, New Zealand.

Coulson, J.C. 1956. Mortality and egg production of the meadow pipit with special reference to altitude. *Bird Study 3*: 119-132.

Coulson, T., Mace, G.M., Hudson, E., and Possingham, H. 2001.The use and abuse of population viability analysis. *Trends in Ecology and Evolution 16*: 219-221.

Cunningham, A.A. 1996. Disease risks of wildlife translocations. *Conservation Biology 10*: 349-353.

Daugherty, C.H., Gibbs, G.W., and Hitchmough, R.A. 1993. Mega-island or micro-continent? New Zealand and its fauna. *Trends in Ecology and Evolution 8*: 437-442.

Digweed, S.C., Currie, C.R., Carcamo, H.A., and Spence, J.R. 1995. Digging out the "digging in" effect of pitfall traps: Influences of depletion and disturbance on catches of ground beetles (Coleoptera: Carabidae). *Pedobiologia 39*: 561-576.

Dijkstra, C., Vuursteen, L., Daan, S., and Masman, D. 1982. Clutch size and laying date in the kestrel *Falco tinnunculus*: Effect of supplementary food. *Ibis 124*: 210-213.

Dixon, C.L. 1978. Breeding biology of the Savannah sparrow on Kent Island. *The Auk 95*: 235-246.

Duncan, P.J., Webb, P.I., and Palmeirim, J.M. 1999. Distribution of New Zealand robins within a forest mosaic. *Emu 99*: 222-226.

Dunn, E. 1977. Predation by weasels *(Mustela nivalis*) on breeding tits (*Parus* spp.) in relation to the density of tits and rodents. *Journal of Animal Ecology 46*: 633-651.

Fieberg, J., and Ellner, S.P. 2000. When is it meaningful to estimate an extinction probability? *Ecology 81*: 2040-2047.

Fitzgerald, B.M., Karl, B.J., and Mollër, H. Spatial organization and ecology of a sparse population of house mice (*Mus musculus*) in a New Zealand forest. *Journal of Animal Ecology 50*: 489-518.

Flack, J.A.D. 1973. Robin research – a progress report. *Wildlife – a review 4*: 28-36.

Flack, J.A.D. 1979. Biology and ecology of the South Island robin. Pp. 22-26 *in* The Ecology of Kowhai Bush, Kaikoura. Hunt, D.M., and Gill, B.J. (Eds). Mauri Ora Special Publication 2. Canterbury University of Christchurch, New Zealand.

Flack, J.A.D., and Lloyd, B.D. 1978. The effects of rodents on the breeding success of the South Island robin. Pp. 59-66 i*n* The Ecology and Control of Rodents in New Zealand Nature Reserves. Dingwall, P.R., Atkinson, I.A.E., and Hay, C (Ed.s). Information Series No. 4. Department of Lands and Survey, Wellington, New Zealand.

Gaines, M.S., and McClenaghan, Jr., L.R. 1980. Dispersal in small mammals. *Annual Review of Ecology and Systematics 11*: 163-196.

Godfray, H.C.J. 1991. Signaling of need by offspring to their parents. *Nature 352*: 328-330.

Gould, W.R., and Nichols, J.D. 1998. Estimation of temporal variability of survival in animal populations. *Ecology 79*: 2531-2538.

Green, C. 1996. Survey and monitoring techniques for insects. *Ecological Management 4*: 73-90.

Green, C. 2000. Pitfall trapping for long-term monitoring of invertebrates. *Ecological Management 8*: 73-93.

Green, R.E. 1989. Transformation of crude proportions of nests that are successful for comparison with Mayfield estimates of nest success. *Ibis 131*: 305-306.

Green, R.E., Pienkowski, M.W., and Love, J.A. 1996. Long-term viability of the reintroduced population of the white-tailed eagle *Haliaeetus albicilla* in Scotland. *Journal of Applied Ecology 33*: 357-368.

Griffith, B., Scott, J.M., Carpenter, J.W., and Reed, C. 1989. Translocation as a species conservation tool: Status and strategy. *Science 245*: 477-480.

Hammond, M.C., and Forward, W.R. 1956. Experiments on causes of duck nest predation. *Journal of Wildlife Management 20*: 243-247.

Hofer, H., and East, M.L. 1998. Biological conservation and stress. *Advances in the Study of Behaviour 27*: 405-497.

Van Horne, B. 1983. Density as a misleading indicator of habitat quality. *Journal of Wildlife Management 47*: 893-901.

Hooker, S., and Innes, J. 1995. Ranging behaviour of forest-dwelling ship rats, *Rattus rattus*, and effects of poisoning with brodifacoum. *New Zealand Journal of Zoology 22*: 291-304.

Hoover, J.P., Brittingham, M.C., and Goodrich, L.J. 1995. Effects of forest patch size on nesting success of wood thrushes. *The Auk 112*: 146-155.

Howard, M. 1998. North Island robin (*Petroica australis longipes*) transfer – Tarawera Conservation Area to Boundary Stream Mainland Island. Unpublished report for the Department of Conservation.

Howard, M. 2000. Reintroduction and monitoring of North Island robin (*Petroica australis longipes*) at Boundary Stream Mainland Island June 1998-June 1999. Unpublished report for the Department of Conservation.

Howard, M., and Christensen, B. 2001. Reintroduction and monitoring of North Island robin (*Petroica australis longipes*) at Boundary Stream Mainland Island June 1998-June 2000. Unpublished report for the Department of Conservation.

Innes, J. 1979. Diet and reproduction of ship rats in the northern Tararuas. *New Zealand Journal of Ecology 2*: 85-86.

Innes, J., and Barker, G. 1999. Ecological consequences of toxin use for mammalian pest control in New Zealand – an overview. *New Zealand Journal of Ecology 23*: 111-127.

Innes, J., Warburton, B., Williams, D., Speed, H., and Bradfield, P. 1995. Large-scale poisoning of ship rats (*Rattus rattus*) in indigenous forests of the North Island, New Zealand. *New Zealand Journal of Ecology 19*: 5-17.

IUCN, 1987. Position Statement on the Translocation of Living Organisms: Introductions, Reintroductions, and Re-stocking. IUCN Council, Gland, Switzerland.

IUCN. 1998. Guidelines for reintroductions. Prepared by the IUCN/SSC reintroduction specialist group. IUCN, Gland, Switzerland.

Jones, J.M., and Witham, J.H. 1990. Post-translocation survival and movements of metropolitan white-tailed deer. *Wildlife Society Bulletin 18*: 434-441.

Johnson, D.H. 1979. Estimating nest success: the Mayfield method and an alternative. *Wilson Bulletin 96*: 651-661.

Johnson, D.H., and Shaffer, T.L. 1990. Estimating nest success: when Mayfield wins. *The Auk 107*: 595-600.

King, C.M. 1982. Age structure and reproduction in feral New Zealand populations of the house mouse (*Mus musculus*) in relation to seed fall of southern beech. *New Zealand Journal of Zoology 9*: 467-480.

King, C.M. 1984. Immigrant killers. Introduced predators and the conservation of birds in New Zealand. Oxford University Press, Auckland.

King, C.M. (Ed). 1995. The Handbook of New Zealand Mammals. Oxford University Press, Auckland.

King, C.M., Innes, J.G., Flux, M., Kimberley, M.O., Leathwick, J.R., and Williams, D.S. 1996. Distribution and abundance of small mammals in relation to habitat in Pureora Forest Park. *New Zealand Journal of Ecology 20*: 215-240.

Kleiman, D.G. 1989. Reintroduction of captive mammals for conservation. *Bioscience 39*: 152-161.

Kurzejeski, E.W., and Root, G. 1988. Survival of reintroduced ruffed grouse in north Missouri. *Journal of Wildlife Management 52*: 248-252.

Lack, D. 1947. The significance of clutch size. *Ibis 89*: 302-352.

Lack, D. 1954. The Natural Regulation of Animal Numbers. University Press, Oxford.

Lacy, R.C. 1993. VORTEX: A computer simulation model for population viability analysis. *Wildlife Research 20*: 45-65.

Lebreton, J-D.,Burnham, K.P., Clobert, J., and Anderson, D.R. 1992. Modeling survival and testing biological hypotheses using marked animals: a unified approach with case studies. *Ecological Monographs 62*: 67-118.

Lebreton, J-D., Pradel, R., and Clobert, J. 1993. The statistical analysis of survival in animal populations. *Trends in Ecology and Evolution 8*: 91-95.

Leech, S.M., and Leonard, M.L. 1997. Begging and the risk of predation in nestling birds. *Behavioural Ecology 8*: 644-646.

Lovegrove, T.G. 1992. Effects of introduced predators on the saddleback (*Philesturnus carunculatus*) and implications for management. PhD thesis, University of Auckland, Auckland.

Maloney, R. F., and McLean, I.G. 1995. Historical and experimental learned predator recognition in free-living New Zealand robins. *Animal Behaviour 50*: 1193-1201.

McLean, I.G., Hölzer, C., and Studholme, B.J.S. 1999. Teaching predator-recognition to a native bird: Implications for management. *Biological Conservation 87*: 123-130.

Maunder, M. 1992. Plant reintroduction: an overview. *Biodiversity and Conservation 1*: 51-61.

Mayfield, H. 1961. Nesting success calculated from exposure. *Wilson Bulletin 73*: 255-261.

Mayfield, H. 1975. Suggestions for calculating nest success. *Wilson Bulletin 87*: 456-466.

McLean, I.G., Hölzer, C., and Sagar, P.M. 1994. Niche overlap and foraging ecology of island *Petroica* species. *Notornis (Supplement) 41*: 39-48.

Miller, H.W., and Johnson, D.H. 1978. Interpreting the results of nesting studies. *Journal of Wildlife Management 42*: 471-476.

Mollër, H., and Alterio, N. 1998. Secondary poisoning of predators. *The Green Monitor 18*: 1-4.

Moors, P.J. 1978. Methods for studying predators and their effects on forest birds. Pp. 47-57 *in* The Ecology and Control of Rodents in New Zealand Nature Reserves. Dingwall, P.R., Atkinson, I.A.E., and Hay, C. (Ed.s). Information Series No. 4. Department of Lands and Survey , Wellington, New Zealand.

Moors, P.J. 1983. Predation by stoats (*Mustela erminea*) and weasels (*M. nivalis*) on nests of New Zealand forest birds. *Acta Zoologica Fennica 174*: 193-196.

Moors, P.J. 1995 Norway Rat. Pp. 192-206 *in* The Handbook of New Zealand Mammals. King, C. (Ed.). Oxford University Press, Auckland.

Murphy, E., and Bradfield, P. 1992. Change in diet of stoats following poisoning of rats in a New Zealand forest. *New Zealand Journal of Ecology 16*: 137-140.

Musil, D.D., Connelly, J.W., and Reese, K.P. 1993. Movements, survival and reproduction of sage grouse translocated in central Idaho. *Journal of Wildlife Management 57*: 85-91.

Newton, I. 1980. The role of food in limiting bird numbers. *Ardea 68*: 11-30.

O'Brian, M.K., and McCullough, D.R. 1985. Survival of black-tailed deer following relocation in California. *Journal of Wildlife Management 49*: 115-119.

Powlesland, R.G. 1981a. The foraging behaviour of the South Island robin. *Notornis 28*: 89-102.

Powlesland, R.G. 1981b. Comparison of time-budgets for mainland and Outer Chetwode Island populations of adult male South Island robins. *New Zealand Journal of Ecology 4*: 98-105.

Powlesland, R.G. 1997. Protocols for monitoring New Zealand robins (*Petroica australis*). Department of Conservation Technical Series 13. Department of Conservation, Wellington, New Zealand.

Powlesland, R.G., Knegtmans, J.W., and Marshall, I.S.J. 1999. Costs and benefits of aerial 1080 possum control operations using carrot baits to North Island robins (*Petroica australis longipes*), Pureora Forest Park. *New Zealand Journal of Ecology 23*: 149-159.

Powlesland, R.G., Knegtmans, J.W., and Marshall, I.S.J. 2000. Breeding biology and success of North Island robins (*Petroica australis longipes*) in Pureora Forest Park. *Notornis 47*: 97-105.

Price, K., and Ydenberg, R. 1995. Begging and provisioning in broods of asynchronously-hatched yellow-headed blackbird nestlings. *Behavioural Ecology and Sociobiology 37*: 201-208.

Quinney, T.E., Hussell, D.J.T., and Ankey, C.D. 1986. Sources of variation in growth of swallows. *The Auk 103*: 389-400.

Quinney, T.E., and Smith, P.C. 1980. Comparative foraging behaviour and efficiency of adult and juvenile great blue herons. *Canadian Journal of Zoology 58*: 1168-1173.

Richner, H. 1992. The effect of extra food on fitness in breeding carrion crows. *Ecology 73*: 330-335.

Riddington, R., and Gosler, A.G. 1995. Differences in reproductive success and parental qualities between habitats in the Great Tit *Parus major*. *Ibis 137*: 371-378.

Rotella, J.J., Taper, M.L., and Hansen, A.J. 2000. Correcting nesting-success estimates for observer effects: Maximum-likelihood estimates of daily survival rates with reduced bias. *The Auk 117*: 92-109.

Saunders, A. 1992. Workshop introduction. Pg. 3 *in* Proceedings of the National Predator Management Workshop. Department of Conservation, Wellington, New Zealand.

Saunders, A. 2000. A review of Department of Conservation Mainland Restoration Projects and Recommendations for Further Action. Department of Conservation, Wellington, New Zealand.

Seber, G.A.F. 1982. The estimation of animal abundance and related parameters. 2nd edition. Charles Griffin & Company, London.

Seki, S., and Takano, H. 1998. Caterpillar abundance in the territory affects the breeding performance of great tit *Parus major minor*. *Oecologia 114*: 514-521.

Sharell, R. 1971. New Zealand Insects and Their Story. Auckland: Collins.

Siegel, H.S. 1980. Physiological stress in birds. *Bioscience 30*: 529-534.

Simons, L.S., and Martin, T.E. 1990. Food limitation of avian reproduction: An experiment with the Cactus Wren. *Ecology 71*: 869-876.

Slough, B.G. 1989. Movements and habitat use be transplanted marten in the Yukon Territory. *Journal of Wildlife Management 53*: 991-997.

Spaans, A.L., de Wit, A.A.N., and van Vlaardingen, M.A. 1987. Effects of increased population size in Herring Gulls on breeding success and other parameters. *Studies in Avian Biology 10*: 57-65.

Spurr, E.B. 1999. Developing a long-life toxic bait and lures for mustelids. Pp. 1-24 *in* Progress in Mammal Pest Control on New Zealand Conservation Lands. Department of Conservation, Wellington, New Zealand.

Spurr, E.B., and O'Connor, C.E. 1999. Sound lures for stoats. Pp. 25-38 *in* Progress in Mammal Pest Control on New Zealand Conservation Lands. Department of Conservation, Wellington, New Zealand.

Stanley, T.R. 2000. Modeling and estimation of stage-specific daily survival probabilities of nests. *Ecology 81*: 2048-2053.

Sullivan, K. 1989. Predation and starvation: Age-specific mortality in juvenile juncos (*Junco phaenotus*). *Journal of Animal Ecology 58*: 275-286.

Taylor, B. L. 1995. The reliability of using population viability analysis for risk classification of species. *Conservation Biology 9*: 551-558.

Taylor, R. 1992.The eradication of Norway Rats from Breaksea Island. Pg. 30 *in* Proceedings of the National Predator Management Workshop. Department of Conservation, Wellington, New Zealand.

Thomas, M.D. 1999, Feasibility of using wax-blocks to measure rodent and possum abundance and changes in population size. Pp. 39-48 *in* Progress in Mammal Pest Control on New Zealand Conservation Lands. Department of Conservation, Wellington, New Zealand.

Thomas, D.T., and Sleeper, E.L. 1977. The use of pit-fall traps for estimating the abundance of arthropods, with special reference to the Tenebrionidae (Coleoptera). *Annals of the Entomological Society of America 70*: 242-248.

Tolonen, P., and Korpimäki, E. 1994. Determinants of parental effort: a behavioural study in the European kestrel, *Falco tinnunculus*. *Behavioural Ecology and Sociobiology 35*: 355-362.

Tolonen, P., and Korpimäki, E. 1996. Do kestrels adjust their parental effort to current or future benefit in a temporally varying environment? *Ecoscience 3*: 165-172.

Towns, D.R., and Ballantyne, W.J. 1993. Conservation and restoration of New Zealand island ecosystems. *Trends in Ecology and Evolution 8*: 452-457.

Turner, A.M., and McCarty, J.P. 1998. Resource availability, breeding site selection, and reproductive success of red-winged blackbirds. *Oecologia 113*: 140-146.

Van Vuren, Kuenzi, A.J., Loredo, I., Leider, A.L., and Morrison, M.L. 1997. Translocation as a non-lethal alternative for managing California ground squirrels. *Journal of Wildlife Management 61*: 351-359.

Veitch, C.R. 1995. Habitat repair: a necessary pre-requisite to translocation of threatened birds in New Zealand. Pp. 97-104 *in* Reintroduction Biology of Australian and New Zealand Fauna. Serena, M. (Ed). Surrey Beatty & Sons. Chipping Norton.

Veitch, D., Fitzgerald, M., Innes, J., and Murphy, E. (Eds.). 1992. Proceedings of the National Predator Management Workshop. Department of Conservation, Wellington, New Zealand.

Viggers, K.L., Lindenmayer, D.B., and Spratt, D.M. 1993. The importance of disease in reintroduction programmes. *Wildlife Research 20*: 687-698.

Ward, J.M., and Kennedy, P.L. 1996. Effects of supplemental food on size and survival of juvenile Northern Goshawks. *The Auk 113*: 200-208.

White, G.C. 2000. Modeling population dynamics. Pp 85-807 *in* Ecology and Management of Large Mammals in North America. Demarais, S., and Krausman, S. (Eds). Prentice-Hall, New Jersey.

White, G.C., and Burnham, K.P. 1999. Program MARK: survival estimation from populations of marked animals. *Bird Study 46 Supplement*: 120-138.

Wiehn, J., and Korpimäki, E. 1997. Food limitation on brood size: experimental evidence in the Eurasian kestrel. *Ecology 78*: 2043-2050.

Wiles, B. 2000. North Island robins (*Petroica australis longipes*) in a restoration landscape (Mana Island, Wellington). MSc thesis. Victoria University, Wellington.

Willis, E.O. 1973. Survival rates for visited and unvisited nests of bicolored antbirds. *The Auk 90*: 263-267.

Wilson, R.J., Drobney, R.D., Hallett, D.L. 1992. Survival, dispersal and site fidelity of wild female ring-necked pheasants following translocation. *Journal of Wildlife Management 56*: 79-85.

Wingfield, J.C., Hunt, K., Breuner, C., Dunlap, K., Fowler, G.S., Freed, L., and Lepson, J. 1997. Environmental stress, field endocrinology, and conservation biology. Pp 95-131 *in* Behavioural Approaches to Conservation in the Wild. Clemmons, J.R., and Buchholz, R. (Ed.s). Cambridge University Press, Cambridge.

Wolf, C.M., Griffith, B., Reed,C., and Temple, S.A. 1996. Avian and mammalian translocations: update and reanalysis of 1987 survey data. *Conservation Biology 10*: 1142-1154.

Zanette, L., Doyle, P., and Trémont, S.M. 2000. Food shortage in small fragments: Evidence from an area-sensitive passerine. *Ecology 81*: 1654-1666.