Applying Structured Decision Making to management of the reintroduced hihi population in Bushy Park

Female hihi. Photo by Author.

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ABSTRACT

The use of Structured Decision Making (SDM) for choosing optimal management actions in reintroduction projects has been recently pioneered by the North American whooping crane *Grus americana* programme. SDM requires projecting population dynamics under different scenarios to predict outcomes of management strategies. In this thesis, I applied SDM to a population of an endangered New Zealand forest bird, the hihi *Notiomystis cincta*, which was reintroduced to Bushy Park in March 2013 when 44 birds were released. My aim was to determine the optimal management of the Bushy Park population.

The need of this decision was triggered by Bushy Park Trust application for additional translocation of 15 females in order to reinforce the population. The Hihi Recovery Group developed four fundamental objectives, which included maximizing the number and persistence of female hihi in Bushy Park as well as to minimize the impact on the source population on Tiritiri Matangi Island and minimizing costs; and three management alternative actions, including the status quo and follow-up translocations of 15 females in either 2015 or 2016.

In order to project population dynamics under each alternative, I estimated the survival and reproduction rates of the Bushy Park population based on the 18 months of the monitoring data. Comparison of the survival rates of translocated juveniles and juveniles that were born in Bushy Park allowed distinguishing between age and post-release effects. Modelling indicated that translocated juveniles experienced post-release effects that resulted in the rapid population decline during the first 6 months. Survival rates were then used in population viability analysis in program OpenBUGS. An integrated population model was designed to model fecundity and the Bushy Park population dynamics over 10 years under the above-mentioned management alternatives. The median projections suggested a slow decline of the population under each management alternative, but with great uncertainty.

I used the novel approach for decision analysis, whereby uncertainty was incorporated into the decision. In one single model I combined the released population, the source population and the Simple Multi-Attribute Rating Technique for decision analysis. This approach showed that none of the alternatives were clearly preferred and the decision was sensitive to uncertainty in the projections.

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Chapter 1. General introduction

1.1 MANAGEMENT OF THREATENED SPECIES

Humans have become the dominant species on the earth. During the 20th century alone, the human population grew from 1.65 billion to six billion. Now the population size is more than seven billion people and it continues to grow. This fact is important because people live on a limited surface and need to compete with other creatures for food and habitat. Hence, the more people that live on Earth, the fewer the resources available for other fauna. Meine (2010) reported that people have already directly consumed, or devastated by their activities, almost 50% of all the food available to other animals. Enormous habitat loss around the globe has become one of the major reasons for rapid biodiversity decline (Meine 2010).

To control and manage this significant impact on species and their habitats, most countries have developed or ratified conservation legislations. The Convention on International Trade in Endangered Species of Wild Fauna and Flora and Agenda 21 of the United Nations are examples of conventions at the international level. On the other hand, The United States' Endangered Species Act, The New Zealand Biodiversity Strategy, and the French Nature Conservation Act 1976 are examples of national legislation (Caughley & Gunn 1996). Many countries, such as New Zealand, the USA, Australia and Canada, have developed species-specific recovery plans, which are documents describing the protocol for managing and protecting an endangered species.

One of the main goals of species recovery programs is to manage the conflict between human activities and the biodiversity conservation (Young et al. 2005). In addition, according to Norton (1992) the primary goal of conservation biologists is to protect the ability of an ecosystem to self-organize and regenerate. Ironically, in many cases this means managing human activity in order to reduce or eliminate their impact on the ecosystem.

Although restoration and conservation projects are becoming more popular throughout the world, many of these ventures do not bring long-term outcomes (Hilderbrand et al. 2005; Seddon et al. 2012). Nature is a complex system, and attempts to over-simplify it may lead to project failure (Holling et al. 1995; Holling & Meffe 1996). It is important to understand this complexity, and develop long-term strategies that will consider species' interactions, as well as a link between species diversity and ecosystem function (Soulé 1985; Schwartz et al. 2000). Successful conservation projects should be designed to test a range of different hypothesis, based on adaptive learning in order to understand different outcomes and include uncertainties (Figure 1.1). In order to help people who are working with threatened species in New Zealand, the Department of Conservation developed species-specific Recovery groups. Members of these groups are scientists, community groups and others who are interested and/or experienced in conservation of particular threatened species. The activity of Species Recovery Groups follows the goals of Species Recovery Plans.

In this chapter, I described the process of reintroduction, a method that is increasingly used in the conservation of threatened species. I also described and discussed other essential aspects of threatened species management, such as Structured Decision Making, Population Viability Analysis and Monitoring, and described how I applied these tools to managing a recently reintroduced population of hihi, an endangered New Zealand forest bird hihi.

1.2 REINTRODUCTION

One of the most effective methods of ecological restoration and conservation of species is conservation translocation, which means international movement of a species with the aim to improve its conservation status (Seddon et al. 2014). When the aim of such translocation is to re-establish a self-sustaining population of the species within its native habitat it is called *Reintroduction* (IUCN 2013). According to the IUCN (2013), reintroduction is the “intentional movement and the release of an organism inside its indigenous range from which it has disappeared”. In the early years reintroduction projects had low success. However, success is increasing as we learn more about species needs and translocation techniques. In New Zealand, the number of successful conservation translocations has increased since the 1960s due to improved techniques to capture and protect animals, such as mist nets and pest control (Miskelly & Powlesland 2013). New Zealand offshore islands have been used successfully for numerous conservation translocations. Although there are many species suitable for translocation, in New Zealand the majority have been birds and plant species. Due to island translocations, many native bird species such as little spotted kiwi *Apteryx oweni*, black robin *Petroica traversi* and South Island saddleback *Philesturnus carunculatus carunculatus* were saved from extinction (Armstrong & McLean 1995).

Many studies have been done in order to investigate the factors that contribute to a successful reintroduction project (e.g. Griffith et al. 1989; Wolf et al. 1996). Findings suggest that habitat (quality and location in relation to the historical range), number of released animals, number of releases and general adaptability of the translocated species affect the success of reintroduction the most.

Often it is difficult to find suitable sites for translocation. Armstrong & McLean (1995) suggested that while the standard strategy for habitat analyses is to look for sites with similar characteristics to where the species currently exists, current location of threatened species does not always indicate a suitable environment. For example, populations of long-lived animals may continue to exist in the same area long after it becomes unsuitable for reproduction and therefore long-term persistence (Seddon et al. 2012). As highlighted by Seddon et al. (2012), the present location of the species may not remain suitable in the future, due to the factors such as climate change, unsustainable land use and human demographic growth. The historical presence of a species is also not a reliable indicator of a suitable release site, because the longer the time between extinction and reintroduction, the greater the chance that the habitat has changed and requires management before the species is released.

Consequently, in some cases suitable release site needs to be created through restoration. A good example of habitat restoration is Tiritiri Matangi Island. For a long time, the island was used by humans for agricultural purposes, but in 1984, after the publication of the "Tiritiri Matangi Island working plan" by Department of Land and Survey, a big restoration project was initiated. Now this island is the habitat for many reintroduced species, although some of them, including hihi *Notiomystis cincta*, still need intensive management. In the hihi's case, management is in the form of supplementary feeding and provision and maintenance of nest boxes. In addition, since the main reason for declines of New Zealand native species is introduced mammals (Craig et al. 2000), such release sites may need constant monitoring for introduced predator incursions, and sometimes ongoing predator control.

It is also important to find the right number of individuals that need to be translocated. As well as habitat quality, the number of released individuals would be different for each taxa and vary from project to project. Armstrong and McLean (1995) noted that in New Zealand, managers often use 40 individuals for bird translocation projects. Although it is not a strict rule, smaller numbers of released animals may lead to project failure, because small populations are more likely to become extinct due to demographic or environmental stochasticity, inbreeding and lack of genetic variation (Armstrong & McLean 1995; Keller et al. 2012). From the genetic point of view, the release of 20 individuals or more insures a higher level of gene diversity (proportion of polymorphic loci across the genome > 97.5%; Jamieson & Lacy 2012). However, conservation managers have to take into account available resources, e.g. monetary costs and labour, and most importantly the size of the source population, no matter whether it is captive or wild. For example, a population of black robin *Petroica traversi* was established from the remaining group of 5 individuals (including 1 female), and a population of the Mauritius kestrel *Falco punctatus* has recovered from only a single wild-breeding pair (Armstrong & McLean 1995; Jamieson et al. 2006).

To project population dynamics under different scenarios (e.g. different numbers of individuals translocated or origin of release birds) and predict short-term consequences of management strategies, managers can use mathematical models. Seddon et al. (2007) asserts that some elements of population modelling should be a part of every reintroduction evaluation. With the help of population modelling, managers may estimate vital rates and project short- or long-term viability of the establishing populations. This sort of analysis may also help to understand the value of additional translocations to reinforce current population. Armstrong and Ewen (2001) defined this kind of movement as "follow-up translocation".

To summarise, reintroduction is a multifaceted action. Sarrazin and Barbault (1996) described it as "a way to experiment in ecology". According to them, the learning component of reintroduction projects gives an opportunity for population ecologists to study species' behaviour in new conditions, as well as colonization features, density-dependent processes, intraspecific competition and consequences of disturbance of food chains. Moreover, it is a unique opportunity to study a "population in which the origin of each individual is known" (Sarrazin & Barbault 1996). The importance of the learning component of a reintroduction project was also highlighted by Armstrong et al. (1995), who recommended including an experimental approach to improve the success of reintroduction projects. Reintroduction is a very popular tool in conservation biology; nonetheless I agree with Seddon et al. (2012) that it is a "highly unnatural intervention". Humanity has irreversibly changed the habitat of many New Zealand native species and as a result, some of reintroduced populations will never become self-sustained and their viability and competitiveness will depend on constant habitat management and monitoring.

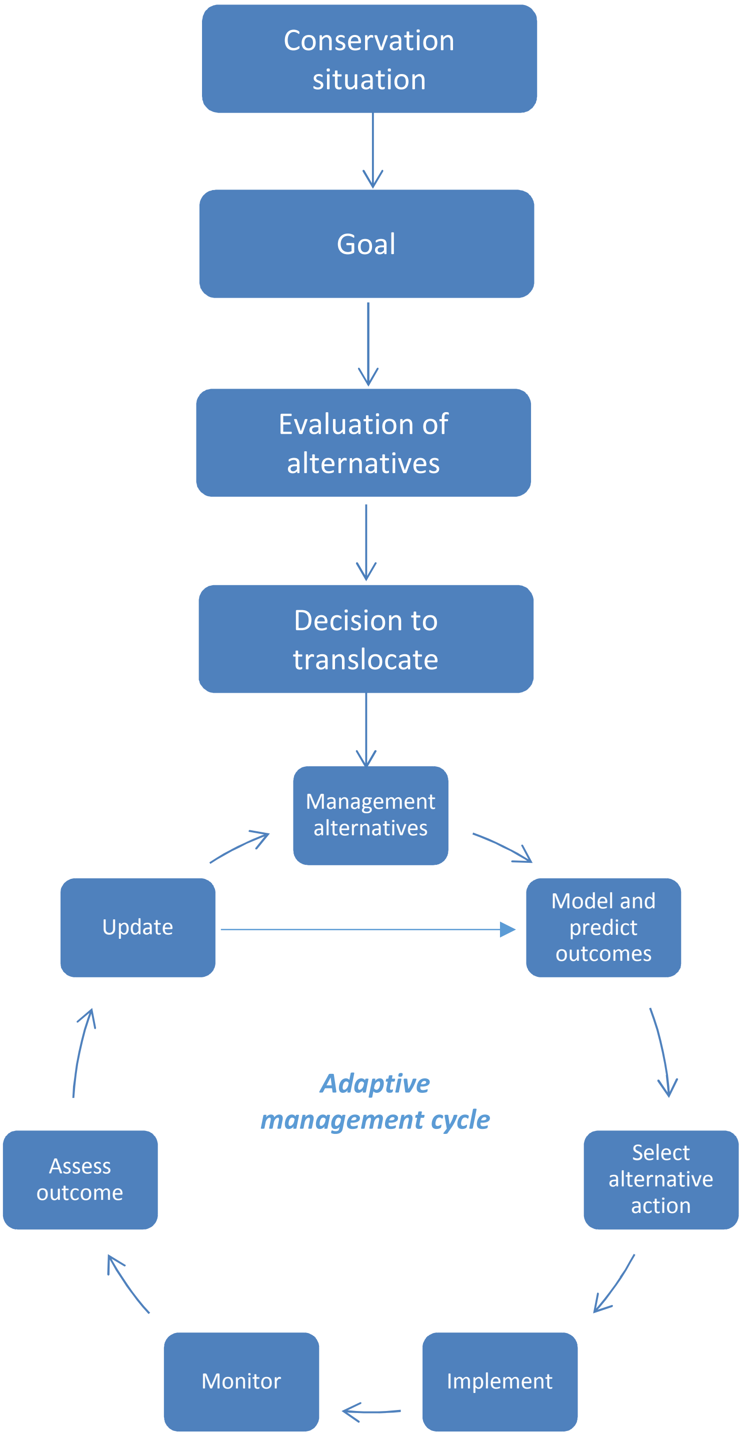

Figure 1.1. Reintroduction project cycle. Each box represents a step in developing a reintroduction project and contains either an approach or tool. The steps are arranged in chronological order. Arrows indicate how approaches evolved from or influence other approaches. Adapted from IUCN (2013).

1.3 STRUCTURED DECISION MAKING

Decisions about how to manage habitat or which individuals to translocate require a strategic approach. Ecological problems have become more complex and affect the interests of many people and organizations. Businessmen, general public, government and scientists may consider the same ecological problem differently, and solutions to this problem may be controversial. Structured Decision Making (SDM) is a useful tool for resolving these complex problems and finding a decision that is a good compromise for stakeholders. SDM, in contrast to unstructured decision making, allows different stakeholders to participate in problem solving, and incorporate their diverse interests in the decision making process (Wilson & Arvai 2011). SDM cycles that I found in the literature do not differ markedly (e.g. Gregory & Long 2009; Gregory et al. 2012; Wilson & Arvai 2011; McCarthy et al. 2012). The proposed cycles often include the following main steps:

1. Define a problem or question.
2. Define objectives and measures.
3. Develop alternatives.
4. Estimate consequences.
5. Consider trade-offs and select preferred alternatives.
6. Implement, monitor and learn.

I briefly describe the main idea of each step here, but the detailed explanation is given in Chapter 4 using the example of the reintroduced hihi population in Bushy Park. During the first steps, stakeholders (or Species Recovery Groups) have to identify their values and express them as objectives. First, they brainstorm different ideas and then identify together the most important objectives, called fundamental objectives. The next step is to develop a set of alternative actions to achieve the defined objectives. This process can be challenging because effective alternatives may not exist or they may be unacceptable for some stakeholders (Lyons et al. 2008). And again, working as a group and discussing even unrealistic ideas could help to choose the sensible set of alternatives. The fourth step is to design a mathematical model to estimate consequences of different management actions. Models can be simple or very complex and it will depend on available knowledge and data. The fifth step is to choose one alternative action over another. There are a lot of techniques that could be used for analysing each potential candidate action, including a decision tree, influence diagram, Bayesian network, and the Simple Multi Attribute Rating Technique (SMART). The last step is to implement the “best” alternative action and observe the new state of the system. The learning stage begins when managers compare their predicted outcome with the observed. This stage is a master link that transforms linear SDM scheme into a cycle (Figure 1.1). If the management is based on the concept “learning by doing” and is designed to improve management it is called *Adaptive management* (Williams & Brown 2012). With the help of adaptive management, managers may learn the effectiveness of different management actions and monitoring techniques. Adaptive management should be used to improve ongoing management of reintroduction projects (Armstrong et al. 2007; McCarthy et al. 2012).

1.4 MONITORING FOR REINTRODUCTION

In modern conservation biology, monitoring should be seen not as a stand-alone activity, but instead as a component of a larger activity (Nichols & Williams 2006) such as SDM or adaptive management, which were described above (Figure 1.1). Monitoring is a process that with the help of different tools provides information on the state of a system over some period of time (Stem et al. 2005).

Monitoring is often costly, so the main question for managers would be “Do we need monitoring?” Monitoring has three main tasks: to inform managers when the system undergoes major changes; to measure the success of management actions; and to detect how disturbances and interventions affect system functioning (Legg & Nagy 2006). In the case of reintroduction projects, monitoring is conducted before and after release.

During *Pre-release* monitoring, the main objective is to determine the suitability of a selected reintroduction area (Nichols & Armstrong 2012). For example, monitoring may be used to assess whether predator density is low enough for successful reintroduction of a particular species, or whether there are competitors for food resources present, or whether the required amount of food is available. If the selected area was defined as *not suitable*, the next question, which monitoring may answer is “What should be done, if anything, to make this area suitable?” (Ewen & Armstrong 2007).

Post-release monitoring is intended to assess whether the project has achieved its goal. It focuses on abundance, population growth and size, as well as recording the vital rates of reintroduced species. Sutherland et al. (2010) developed “Standards for documenting and monitoring bird reintroduction projects”, where they suggested that post-release monitoring should be conducted at least during the first year after release, then in five years and for long-lived species even in 10, 15 or 20 years. Well-designed post-release monitoring makes it possible to estimate vital rates and the duration of the post-release period, providing a strong basis for population viability analysis and for further decision making (Ewen & Armstrong 2007).

1.5 POPULATION VIABILITY ANALYSIS

Population Viability Analysis (PVA) has become one of the most powerful and useful tools in conservation biology, and is an important part of decision making. It is a method used to predict the impact of different deterministic and stochastic factors on a given population (Lindenmayer & Possingham 1996). With the help of PVA, managers are able to assess various risks of reaching some thresholds, such as extinction, and project population dynamics under current management or after some changes, like follow-up translocation or harvesting. McCarthy et al. (2003) identified three types of predictions that could be made with the help of PVA:

- How does the particular management action affect population dynamics of the declining population? This type of prediction helps to identify the effectiveness of a management action.
- What species or populations need management in order to prevent decline? In other words, this type of prediction helps to identify threatened taxa that need protection.
- What management action is the most effective? The third type of prediction helps to choose the best management action from the set of alternatives.

The other question that may be asked by conservation manager is *what is the minimum size of the population that is viable*? Although many studies have used PVA to answer this question, many authors now argue that it is a wrong conservation focus and that it is impossible to accurately estimate the viable population size because it is sensitive to small errors in demographic data (Caughley 1994; Lindenmayer & Possingham 1996; Reed et al. 2002).

There are several types of demographic models used in PVA. Beissinger and Westphal (1998) defined five types, which are analytical models, deterministic, stochastic, metapopulation and spatially explicit. Analytical models are used mostly to examine system behaviour rather than predict quantitative outcomes. In contrast, deterministic and stochastic models are used to project changes in population size. Deterministic models are the simplest because they require the fewest parameters. Stochastic models are more complex as they incorporate uncertainties. Metapopulation and spatially explicit models are spatially structured and incorporate dispersal rates of individuals.

Although conservation managers widely use PVA to predict risk of population extinction, it also has limitations that could affect the reliability of the projections obtained. First of all, PVA should be treated as a model, which is a simplification of complex system and which cannot be fully correct (Reed et al. 1998). Second of all, with small populations, four factors must be considered during PVA: demographic stochasticity, genetic stochasticity, environmental stochasticity and catastrophes (Shaffer 1981). Moreover, to estimate extinction probabilities of a given population, one needs to estimate survival rates and reproductive success, and how these parameters vary among years (Reed et al. 1998). Obtaining reliable data for long-term extinction probability may take many years. McCarthy et al. (2003), in their study of the reliability of the relative predictions in PVA, found that 10 years of data make it possible to predict relative risk of extinction within 100 years into the future. Thus to make a reliable prediction with the help of PVA, managers need (1) to develop an appropriate model structure, allowing for uncertainty and (2) have unbiased vital rates estimates that incorporate uncertainties. PVA was the essential part of this thesis as it was used for evaluation of the alternative management actions applied to increase the viability of hihi population in Bushy Park.

1.6 BRIEF HISTORY OF HIHI REINTRODUCTION AND MANAGEMENT

The hihi (or stitchbird) *Notiomystis cincta* is an endangered New Zealand forest bird. Before European colonization hihi were present all over the North Island mainland, and offshore islands including Great Barrier, Little Barrier (Hauturu) and Kapiti. However, by the 1880s they had become extinct everywhere except Little Barrier Island (Boyd & Castro 2000). Because only a single hihi population remained, they were classified as vulnerable, and the New Zealand Wildlife Service developed a program of translocations, whereby hihi from Little Barrier Island were released on predator-free islands during the period 1980–1987 (Boyd & Castro 2000). The first attempts to recover hihi population during this time were unsuccessful – the translocated populations on Cuvier, Hen and Kapiti Islands declined slowly. The main reason hypothesised for failure was a limitation of food availability and diversity throughout the year (Rash et al. 1996, Castro et al. 2003). Castro et al. (1994), in their study of feeding and breeding behaviour of hihi, recommended assisting hihi on Kapiti Island with supplementary food, especially during the spring when nectar was limited. This hypothesis led to a series of food-supplementation experiments in an adaptive management framework when hihi were reintroduced to Mokoia Island. During these experiments Armstrong et al. (2007) manipulated the availability, distribution and quality of supplementary food that was provided to hihi. Although the population on Mokoia had only marginally positive growth when food was provided, the experiments showed that feeding had a major effect (the population would have declined rapidly without it), and therefore has been used to improve hihi population growth on other sites. For example, on Tiritiri Matangi supplementary food has been provided continuously since 1996 (Armstrong et al. 2002). On Kapiti Island, continuous supplementary feeding, which has been provided continuously since 2000, also appears to have improved population growth (Chauvenet et al. 2012).

The translocation program has been continued by the New Zealand Department of Conservation since its formation in 1987. The first recovery plan, which documented regulated hihi conservation, was published in 1996. The main long-term goal of the Plan was "*To increase the number of self-sustaining stitchbird populations to five*" (Rash et al. 1996). The plan was updated in 2005 (Taylor et al. 2005).

Since 1980 hihi have been translocated to 10 sites – 5 islands, 4 mainland sanctuaries, and the captive facility at Mt Bruce (Appendix A). They are currently present at 5 of these sites (2 islands and 3 mainland sanctuaries), but are believed to require management at all of these sites (Figure 1.2). All populations receive supplementary food, and some populations are provided with nest boxes which are managed for nest mites.

In 2013 44 hihi were reintroduced to Bushy Park, which has 87 ha of lowland rainforest and is enclosed in a predator-exclusion fence. This site has been managed by Bushy Park Homestead and Forest Trust since 1994, and is administered by a core of volunteers. As a community based project, management of reintroduced species in Bushy Park mostly relies on the volunteer work, donations and grants. Although this feature makes this hihi reintroduction project limited in terms of management opportunities, it was considered as a suitable site for translocation because:

- the area was free of non-native predators,
- the area has a history of successful reintroductions,
- the forest is mature and has tree cavities that could be used by hihi for nests,
- natural sources of fruit and nectar are available,
- the size of the area allows close monitoring.

To prevent food limitation and competition for nest sites, Bushy Park was equipped with sugar-water feeders and nest boxes.

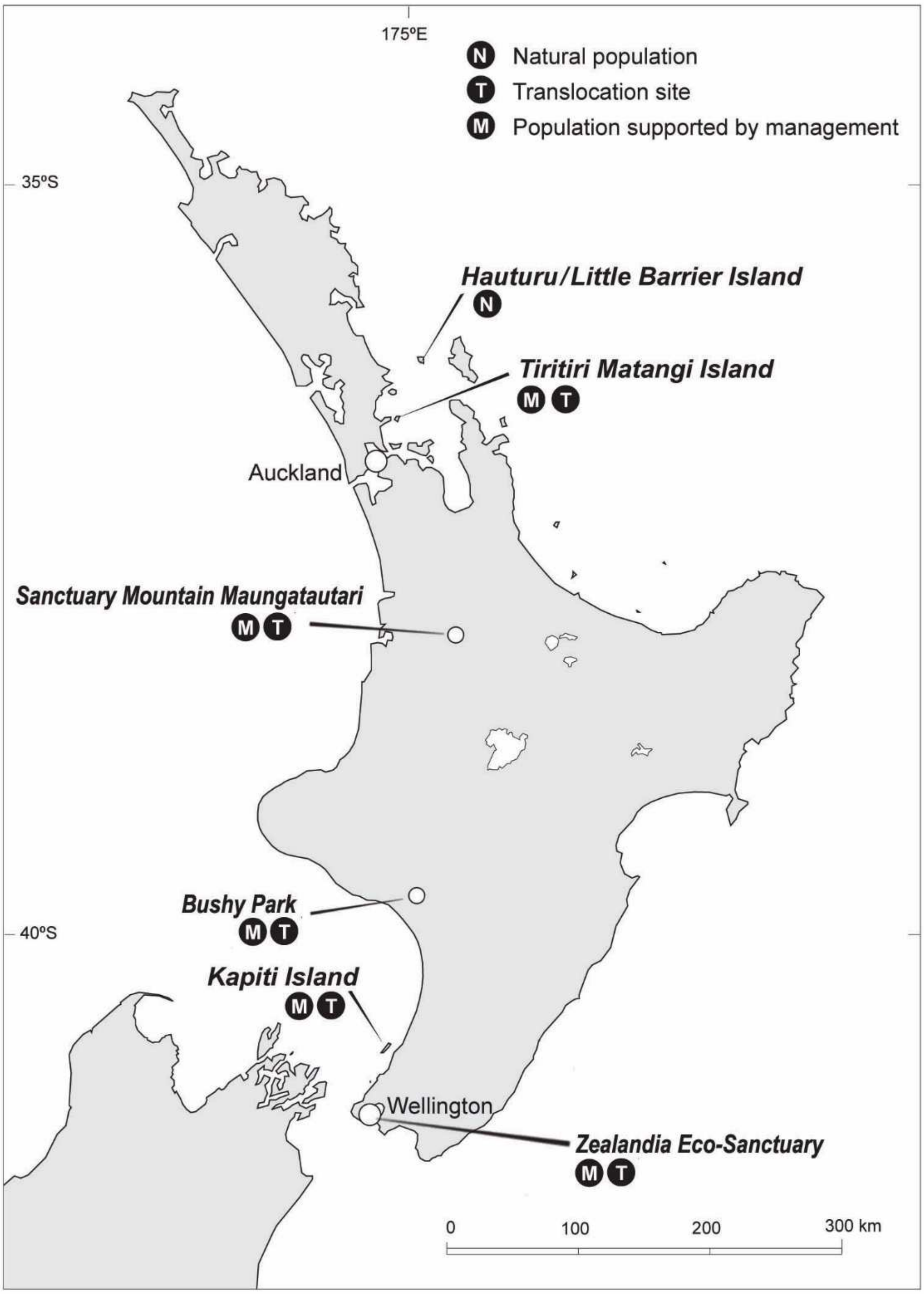

Figure 1.2. The current distribution of hihi Notiomystis cincta on the North Island New Zealand. Adapted from Hihi/stitchbird (Notiomystis cincta) recovery plan, 2004-2009.

1.7 THESIS OBJECTIVES

My thesis aims to find an optimal way of managing the recently reintroduced hihi population in Bushy Park by applying Structured Decision Making (SDM). In this thesis, I assess the value of possible management actions and probability of persistence under each management action. This involves looking at the Bushy Park population as well as the source population, Tiritiri Matangi Island, in order to avoid any risks that harvesting may have.

The subsequent chapters are as follows:

Chapter 2. Modelling survival of the reintroduced population of hihi in Bushy Park.

The aim of this chapter is to obtain reliable survival estimates of the Bushy Park population, based on the monitoring data, in order to perform a population viability analysis. I compare survival models looking at three factors: age, sex and post-release effects. I estimate survival rates for adults and juveniles using the information-theoretical approach and model averaging to account for model uncertainty.

Chapter 3. Assessing short-term viability of the reintroduced hihi population in Bushy Park in the face of uncertainty.

I use PVA to assess population growth and persistence under different management actions. I examine four alternative management actions, including three possible follow-up translocations and just maintaining the current management. For PVA I use survival estimates that were obtained in Chapter 2, and model fecundity simultaneously within the population model. Important uncertainties were incorporated into the model. I project the Bushy Park population size and estimate risk of extinction under the four management actions, and model the short-term effect of harvesting on the source population.

Chapter 4. Applying Structured Decision Making to the reintroduced hihi population in Bushy Park.

The aim of this chapter is to show, using this example, how multi-objective decision problem with uncertainty can be applied. I sub-divide the complex problem into smaller steps in accordance with SDM cycle. I describe steps of SDM, technique for trade-off evaluation and provide necessary information for decision makers, taking into account available information, objective weights and uncertainties.

Chapter 5. General discussion.

The aim of this chapter is to sum up all the chapters and to discuss how reasonable management decisions for reintroduction can be made when post-release data are sparse and projections are uncertain.

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Chapter 2. Modelling survival of the reintroduced population of hihi in Bushy Park

2.1 INTRODUCTION

Modelling survival probability of reintroduced populations is critical for understanding population dynamics and modelling consequences of management actions (Williams et al. 2002). Estimating survival is especially important for geographically closed populations, where it becomes one of the main factors that determine population growth. Age- and sex-specific changes in survival have significant impacts on population growth rates and population viability; in the case of reintroduced populations, age and sex structure is often unstable and depends on the structure of the initial release group. Therefore, in order to project the dynamics of reintroduced population, it is essential to estimate age and sex specific survival (Leslie 1945).

Young individuals are often expected to have lower survival probability than the adults (e.g. Anders et al. 1997; Perkins & Vickery 2001). Younger individuals may have reduced survival because of the low social status, poor locomotors skills, inexperience in avoiding predators and other sources of mortality (Botkin & Miller 1974). Moreover, due to poor foraging skills juveniles are more sensitive to variation in environmental conditions (Robinson et al. 2007; Dybala et al. 2013). Older individuals also may have low survival due to senescence. Defining age groups depends on the species' biology, dispersal phase, reproductive age, and subsequent age-specific reproductive effort (Botkin & Miller 1974). For example, in the case of long-lived animals, like loggerhead turtles *Caretta caretta*, who reach sexual maturity at about 45 years old (Scott et al. 2012), three age groups can be determined: juvenile, sub-adult and adult (Chaloupka & Limpus 2005). On the other hand, many passerine birds reach sexual maturity when they are about one year old. In this case, two age-specific groups may be considered: juveniles and adults.

Males and females may have different survival rates due to some morphological or behavioural characteristics. For example, Promislow et al. (1992), studying the North American passerine bird species, showed that sex specific mortality is greater in species with sexual dimorphism (e.g. size and plumage differences). Population growth of a geographically closed population is often the most sensitive to survival rates of adult females (Williams et al. 2002). In most New Zealand bird species, females appear to be more vulnerable to predation by introduced mammals, because they do not recognize mammals as predators and do not actively avoid them especially while incubating the eggs (Innes et al. 2010). Considering that loss of females would reduce population growth, it may be reasonable to model female survival rates and use female-only models to project population dynamics in such populations (Thorogood et al. 2013).

Finally, when modelling dynamics of reintroduced populations, one should account for post-release effects on survival. Translocated animals experience numerous stressors, including capture, captivity, transportation, release, and adaptation to a new environment (Teixeira et al. 2007, Parker et al. 2012). Stress can increase susceptibility to disease and reduce the ability of translocated animals to learn and remember (Teixeira et al. 2007; Dickens et al. 2010). As a result of stress, vital rates (survival, reproduction and dispersal) might change during some period after translocation (acclimation period). These temporary effects of the translocation process on vital rates are called *post-release effect*s (Armstrong & Reynolds 2012). Post-release effects should be taken into account when estimating survival, because data that were collected after the likely acclimation period would more accurately predict future survival in the population (Armstrong et al. 1999; Hamilton et al. 2010). The length of the acclimation period also depends on factors such as translocation technique, quality of the release site, food availability, and seasonal variation, and will vary among species (Letty et al. 2007). For instance, mortality of European wild rabbit *Oryctolagus cuniculus* was elevated during the first few days after translocation (Letty et al. 2000), and translocated African elephants *Loxodonta africana* had an elevated mortality for about two months after release (Pinter-Wollman et al. 2009). In contrast, reintroduced Red-billed Curassows *Crax blumenbachii* had lower survival during the first 12 months after release as a result of vulnerability to predation (Bernardo et al. 2011). Post-release effects are also observed among translocated birds in New Zealand. For example, survival of North Island saddlebacks *Philesturnus rufusater* in Bushy Park was reduced during the first month after translocation (Gedir et al. 2013) and North Island kokako *Callaeas wilsoni* translocated to Whirinaki Forest had an elevated mortality during the first 44 days (Bradley et al 2012). However, saddlebacks translocated to Mokoia Island and kokako to the Ngapukeriki native forest did not show any signs of post-release effects (Armstrong et al. 2005, Bradley et al. 2012; Molles et al. 2008). Although the literature suggests that acclimation periods are generally relatively short for New Zealand birds (Parker et al. 2013), the time frame is usually hard to predict for each specific case.

In this chapter I estimated survival rates of an endangered New Zealand forest bird, the hihi *Notiomystis cincta*, which was reintroduced to a conservation area – Bushy Park. Modelling of survival probability was carried out to answer the following questions:

- Is there a difference in survival between sexes, and if so, does it occur during the post-release period as well as later on?
- Do post-release effects significantly impact survival, and if so for how long?
- Is there a difference in survival between translocated juveniles and juveniles that were born in Bushy Park?

To answer these questions I compared survival models looking at the three factors that were described above: age, sex and post-release effects.

2.2 METHODS

2.2.1 Species and study site

The hihi *is* a small forest bird (30-40 g) that is the only member of the passerine family Notiomystidae. In their remnant habitat on Little Barrier Island (Hauturu), hihi are mainly found in tawa/rata and tawa/tawhero forests, where they feed on a variety of fruit, flowers and insects (Angehr 1984). Hihi play an important role in ecosystems as they pollinate forest plants and disperse seeds. They are sexually dimorphic: males have a black head with white "ear" and distinctive yellow band on the chest and shoulders, black wings with white bars, whereas females are olive-brown but also have a white bar on the wings (Rash et al. 1996). Their mating system includes various types of polygamy as well as social monogamy (Castro et al. 1996; Ewen et al. 1999). Hihi are obligated cavity nesters, which means they build nests in mature or semi mature trees with deep cavities. Females usually produce two clutches with 2 to 5 eggs in each.

Figure 2.1. Male hihi *Notiomystis cincta* on the water bowl in Tiritiri Matangi Scientific Reserve. Photo: Iuliia Panfylova

Females incubate eggs alone. However, after hatching, most males assist to some extent them with feeding young (Angehr 1984; Ewen 2008; Low et al. 2012). Hihi were formerly relatively common and occupied most of the North Island. By 1885 hihi became extinct on the mainland as a result of European colonization, which brought along the loss of habitat, predation by introduced mammals and avian diseases. Little Barrier Island was the only place where hihi population remained.

44 juvenile hihi were translocated to a new conservation area, Bushy Park, from Tiritiri Matangi Island on March, 24, 2013. Bushy Park, which has 87-ha of near-pristine rainforest, is situated approximately 24 km northwest of Whanganui, New Zealand (N 39°55’48”; E175°2’52”). There is a variety of vegetation including mature rimu-matai forest, mamaku, mahoe, rata, colonies of fern and mosses. Bushy Park has an Xcluder ™ protection fence and at the time of the release was free from non-native predators, except mice. Bushy Park has 11.6 km of monitoring trails, spread across the 17 roughly parallel lines running from the south to the north through the park. There are also over 3 km of public walking tracks.

The success of hihi establishment on a release site depends on the availability of food sources (Castro et al. 2003; Makan et al. 2012). Unfortunately, most of the release sites cannot provide enough quality food sources or nesting sites as they were highly modified by farming. For this reason, all conservation areas provide hihi with supplementary feeding and all, except Maungatautari and Kapiti, with nest boxes. Bushy Park volunteers installed 5 sugar-water feeding stations and 43 nest boxes for hihi.

2.2.2 Monitoring

All individuals were previously marked with one individually numbered metal ring and three coloured plastic rings, thus each bird has a unique colour combination and can be recognised without capture. The 44 initial birds were banded on Tiritiri Matangi Island. About 80% of birds that hatched in Bushy Park were banded by the Department of Conservation.

As monitoring was done by different persons and different monitoring methods were used, I identified four monitoring categories (4k):

1. Radio-tracking during March – April 2013. Forty of the first reintroduced birds had Holohil BD2 tail-mounted transmitters with a nominal 28-day lifespan. Hihi were monitored daily during the first 28 days after release.
2. Systematic surveys during May – September 2013. The monitoring was conducted every 2 weeks: 3-4 days were spent walking along monitoring lines followed then by 3-4 days of feeders monitoring (Figure 2.2).

3. Occasional observations during October 2013 – February 2014. During this period data were collected occasionally by Bushy Park volunteers. This monitoring included watching the feeders and walking along some of the monitoring lines.
4. Monthly systematic surveys February – October 2014. I conducted this systematic monthly surveys and on average spent two days (at least 16 hours) searching for birds by walking along monitoring lines and watching the feeders for at least one hour each. I also included in this category a survey that was conducted with the participation of the Department of Conservation in July 2013, as the same monitoring method was used.

For every sighting, the identity of the individual and its location were recorded. I collected data only on banded birds, because of the type of survival analysis I used.

Monitoring data allowed estimation of adult survival, as well as survival of translocated juveniles and juveniles produced in the first breeding season. Data on juvenile survival allowed distinguishing the effect of age from the post-release effects by comparing survival of translocated juveniles with survival of juvenile that hatched during the 2013/2014 breeding season in Bushy Park.

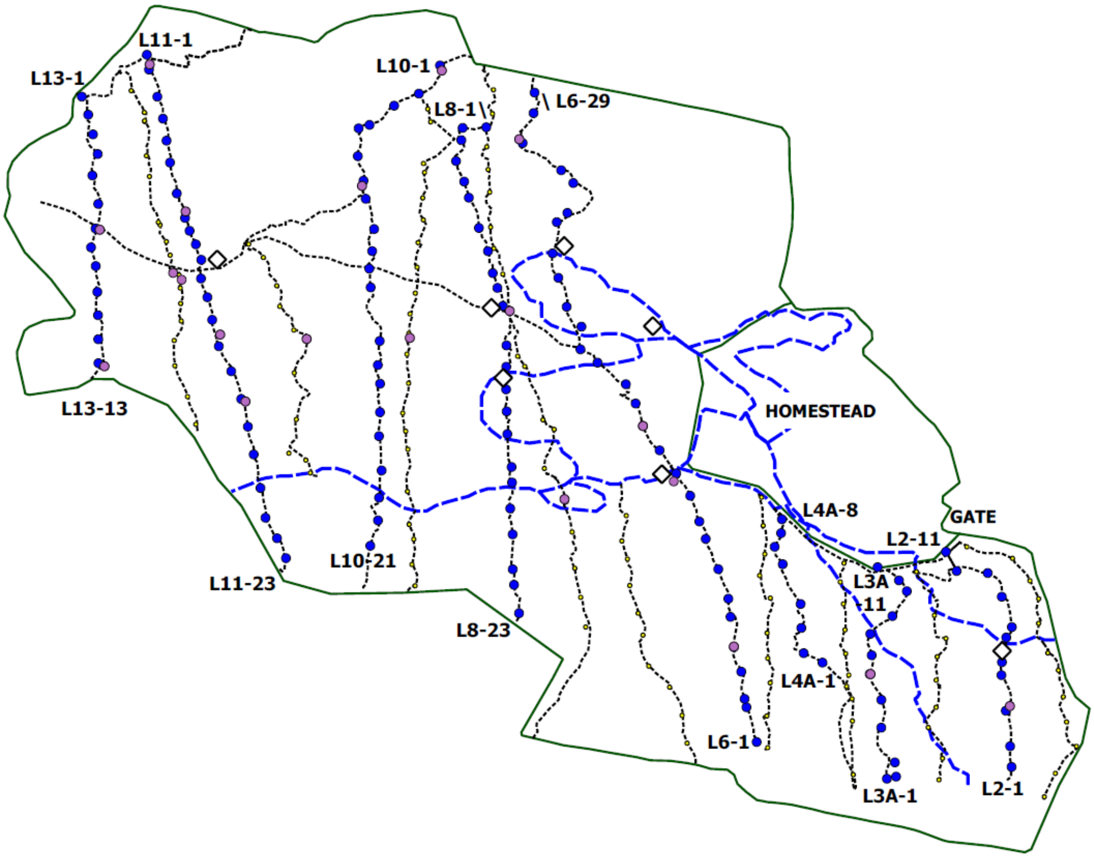

Figure 2.2. Bushy Park tracks and trap lines. Lines walked during hihi survival surveys are shown with blue filled dots (●). Numbers at the beginning and at the end of each line show the numbers of rodent tracking tunnels, which were used as reference points. The rhombus signs (◊) show the position of the feeders. Dashed blue lines show public walking tracks.

2.2.3 Model selection and parameter estimation.

The basis of most models developed to study survival from capture–recapture data is the Cormack–Jolly–Seber model (CJS; Cormack 1964; Jolly 1965). This model allows estimation of two types of parameters: (1) re-capture probability, the probability of a live animal being captured or otherwise detected at one occasion, and (2) survival probability, the probability of an animal alive at one occasion being alive at the next (Cooch & White 2006). The probability of an animal being observed on any occasion depends on both its survival and re-capture probability (Lebreton et al. 1992). The CJS model assumes that the re-capture and survival probabilities at a single capture occasion are the same for all animals in the population. However, it is also possible to fit models with different individual effects (e.g. sex and age) and time variations (e.g. constant, full-time dependent, temporal variations). Other assumptions of the model are that no animals are killed in the capture process, that any emigration is permanent, and that individuals behave independently of one another (Williams et al. 2002).

To analyse survival probabilities of marked individuals, program MARK 7.2 (Cooch & White 2006) was used. It can be freely downloaded from the website http://www.cnr.colostate.edu/~gwhite/mark/mark.htm. The encounter histories for individuals were written in “0-1” format, where “0” means not seen and “1” means seen. A “0” does not necessarily represent death of the individual. Often during the surveys some individuals are not seen but are still alive. Program MARK analyses the proportion of individuals that were missed on each occasion for separate estimation of survival (φ) and re-capture (p) probabilities.

I used parameter index matrices (PIM) and design matrices to add effects that I wanted to test. In this thesis, I refer to *re-capture* parameters as *re-sighting* parameters, because there was no need to capture birds as they were banded with three coloured plastic rings and had a unique colour combination. The time intervals between re-sighting occasions varied from 14 to 74 days, thus I also needed to adjust this in MARK. Under the CJS model the last survival and re-sighting parameter cannot be distinguished. In order to avoid this I conducted two last surveys with 1 day interval.

Re-sighting probability. I first modelled factors that affect re-sighting probability, and then used the best re-sighting model when comparing survival models (Lebreton et al. 1992). For modelling re-sighting I used one of the survival models {Phi(a+1mo+6mo+s)} (see below). I tested the effects of time (t; time-dependent or constant) and sex (s; presence or absence of sex effect) on re-sighting probability. I assumed that the detection probability of juveniles and adults was the same. I also assumed that the detection probability may vary with monitoring categories, since monitoring over the first year was done by different people and different methods of monitoring were used. Hence, I combined re-sighting occasions in four categories (4k). I developed six re-sighting models (Table 2.1).

Table 2.1. Description of models developed to study factors affecting re-sighting probability of hihi in Bushy Park.

Re-sighting models (p)	Description
.	Constant
s	Constant over time, but varies between sexes
t	Fully time dependent with no sex effect
s+t	Varies over time and between sexes, with no interaction between time and sex
4k	Re-sighting occasions grouped into 4 survey types (4k). Re-sighting is otherwise constant over time and between sexes
s+4k	Same as 4k, but with sex effect

s=sex, t=time, 4k= four survey types.

Survival probability. I considered 20 survival models (Table 2.2) and tested the effects of age, sex and post-release effects.

1. Effect of age. I considered survival to be divided into three age classes: fledgling survival (up to March), juvenile survival (March-September) and adult survival (Maness & Anderson 2013). For simplicity I assumed that birds born in Bushy Park entered the population in January 2014, and that was the starting point in their re-sighting histories. During the first 3-8 weeks after fledging, hihi survival is expected to be reduced (Low & Pärt 2009). Thus, *fledging survival* was considered to be different from juvenile survival in all models. The birds translocated to Bushy Park were already 3-4 months old when released, thus their survival over the first 6 six months (March – September 2013, until their first breeding season) was defined as *juvenile survival*. Comparing juvenile survival in 2013 (translocated birds) and 2014 (birds born in Bushy Park) allowed me to estimate the possible post-release effect on juvenile survival.

2. Effect of sex. I tested for sex differences in adult and juvenile during the post-release period and after.

3. Post-release effects. I considered four groups of models in respect of the length of acclimation period:

a) No post-release effects – survival of translocated juveniles was not reduced compared to the survival of juveniles that were born in Bushy Park;

b) One-month **or** six-month post-release effects are considered as plausible periods, meaning that translocated individuals had reduced survival for either of these periods;

c) One-month **and** six-month post-release effect, which means post-release effects act during the 6-month period, but the strongest effect occurs in the first month after translocation. The survival during the acclimation period was called *post-release survival*

(Armstrong & Reynolds 2012), and the subsequent survival after this period was called *post-acclimation survival* (Hamilton et al. 2010).

Table 2.2. Description of the models developed to study factors affecting the survival probability of hihi in Bushy Park following translocation.

Model name*	Description
	Models with no post-release effects
.	Survival does not vary with age or sex effects.
a	Survival of translocated juveniles (first cohort) is the same as juvenile survival from second cohort. Survival of adults is the same for males and females.
s	Survival varies only between sexes and is the same for juveniles and adults.
a+s	Same as {a}, but survival also varies between sexes.
	Models with 1-month post-release effects
1mo	Survival is constant after 1 month
1mo+s	Same as {1mo}, but survival varies between sexes.
a+1mo	Same as {1mo}, but survival varies between adult/juveniles
a+1mo+s	Survival varies between sexes and adult/juveniles.
a+1mo+s(1mo)	Same as {a+1mo+s}, but sex effect acts only during one-month post-release.
1mo+s(1mo)	Same as {a+1mo+s(1mo)}, but with no age effect
	Models with 6-months post-release effects
6mo	Survival is constant after 6-months
6mo+s	Same as {6mo}, but survival varies between sexes
a+6mo	Same as {6mo}, but survival varies between adult/juveniles
a+6mo+s	Survival varies between sexes and adult/juveniles.
a+6mo+s(6mo)	Same as {a+6mo+s}, but sex effect acts only during 6-months post-release
6mo+s(6mo)	Same as {a+6mo+s(6mo)}, but with no age effect
	Models with 6-month post-release effects and stronger effect during the first month
1mo+6mo	Survival is constant after 6-months
1mo+6mo+s	Same as {1mo+6mo}, but survival varies between sexes
a+1mo+6mo	Same as {1mo+6mo}, but survival varies between adult/juveniles
a+1mo+6mo+s	Survival differs between sexes and adult/juveniles.
a+1mo+6mo+ s(6mo)	Same as {a+1mo+6mo+s}, but sex effect acts only during 6-months post-release
a+1mo+6mo+ s(1mo)	Same as {a+1mo+6mo+s}, but sex effect acts only during 1-months post-release
1mo+6mo+ s(6mo)	Same as {a+1mo+6mo+ s(6mo)}, but with no age effect
1mo+6mo+ s(1mo)	Same as {a+1mo+6mo+ s(1mo)}, but with no age effect

a = age effect, 6mo and 1mo= post-release effects act during 1 or 6 months, s=sex;

*each model in a set includes age effect on fledgling survival.

Goodness of fit (GOF) and overdispersion. Goodness of fit test is used to check how well the most general model (the global model) fits the data. There were three closely related models in a set that could be considered as the global models: {Phi(a+1mo+6mo+s))P(4k)}, {Phi((a+1mo+6mo)+s(6mo)) P(4k)} and {Phi((a+1mo+6mo)+s(1mo))P(4k)}. I did GOF on all of these models. In MARK, GOF of the global model to the data can be assessed using parametric bootstrapping. Parametric bootstrapping means that a new data set is generated based on the estimates of the model (in contrast to the nonparametric bootstrap, where many random samples are taken from the original data of the same size as the original sample) to simulate the unknown distribution of the parameters. MARK then fits the model to these simulated data sets to check whether the deviance from the real data falls on the distribution of deviances from the bootstrap values. If the deviance is higher than expected, this is referred to as overdispersion, and may indicate factors affecting survival or re-sighting that are not accounted for in the model (Burnham & Anderson 2002). The degree of overdispersion is measured by the overdispersion parameter $\hat{c}$ (called “c-hat”). If the model fits the data well, the estimate for c-hat will be close to 1. One of the most common approaches for estimating is the median c-hat. In MARK the median c-hat approach simulates data with a range of c values, obtaining a deviance c-hat (deviance divided by the degrees of freedom) for each of the simulated data sets. Then, program MARK performs a logistic regression based on the assumption that the best value of deviance c-hat falls on half of the distribution of all possible deviance c-hat values, and hence half of the simulated values are less than the observed deviance c-hat and half are greater (Cooch & White 2006). Median c-hat probably eliminates the bias associated with the bootstrap method (Cooch & White 2006), and appears to work particularly better with uneven intervals between re-sighting occasions (D. Armstrong, personal comments, November 22, 2014).

Akaike’s Information Criterion (AIC). I compared models based on Akaike’s Information Criterion (Burnham & Anderson 2002). AIC is a measure that assesses the quality of a model in relation to a given set of models. Cooch and White (2006) note that AIC is a useful “tool to achieve an optimal balance between model fit and precision”. The smaller the log likelihood, the better the model fits. On the other hand the more parameters, the lower the precision. It is given by the equation:

$$AIC = -2\ln(L) + 2K,$$

where L is the model likelihood and K is the number of parameters. The model with the smallest AIC value is the best model, meaning it is most likely to be closer to the unknown reality that generated the data. For small sample sizes, a second-order variant of AIC, which is AIC_c, should be used (Burnham & Anderson 2002) and above mentioned formula should be changed to

$$AIC_c = -2\ln(L) + 2K + \frac{2K(K+1)}{n-K-1},$$

where $\frac{2K(K+1)}{n-K-1}$ is a correction factor and n is the sample size. Burnham and Anderson (2002) suggest that when the data are overdispersed ($c>1$), it is best to use the quasi-likelihood, $\ln(L)/c$, and the equation should be modified to $QAIC_c$ which is given by

$$QAIC_c = \frac{-2\ln(L)}{c} + 2K + \frac{2K(K+1)}{n-K-1}.$$

The model with the lowest AIC_c is considered to be the best model, and Delta AIC (ΔAIC), which represents the differences in AIC_c between the best model and every other model, is used to identify the other likely models. The delta AIC is calculated as

$$\Delta AIC = AIC_i - \min AIC,$$

where AIC_i is the AIC value for model *i*, and $\min AIC$ is the AIC value of "the best model". The likelihood (L) of a model (M_i), given the data, is given by

$$L(M_i|data) = \exp(-0.5 * \Delta AIC_i).$$

Model averaging. Often there is no single model in the set of models that is obviously better than the others. If the predicted estimates vary between models, it would be risky to use inference from only one the best model as this means that uncertainty about model selection is ignored (Burnham & Anderson 2002). Model averaging allows estimates under the full range of candidate models to be considered while accounting for their different degrees of support. This degree of support is measured by Akaike weights (ω_i), which are the relative likelihood value (L) of a model divided by the sum of these values across all models. Under model averaging, the estimated value of a parameter is given by:

$$avg(\hat{\theta}) = \sum_{i=1}^{R} \omega_i \, \hat{\theta}_i,$$

where $\hat{\theta}$ is the estimate for the parameter of interest and ω_i is the Akaike weight for model *i*.

MARK generates unconditional standard errors for averaged parameter estimates, meaning it incorporates the uncertainty in model choice. Unconditional standard error is the square root of the sum of the model selection variance and the conditional variance.

2.3 RESULTS

After translocation the Bushy Park population suffered extensive mortality (Figure 2.3). Of the 44 birds translocated in March 2013, 16 females and 21 males survived the first month, and approximately two females and 16 males survived the first year. The fate of the majority of the birds is unknown. However, monitoring of radio-tracked birds in the first month revealed that three females fell prey to morepork *Ninox novaeseelandiae* or New Zealand falcon *Falco novaeseelandiae*; and one or two males died from aspergillosis (Frost 2014). At the beginning of the breeding season in September 2013, 4 females and 16 males were known to be alive.

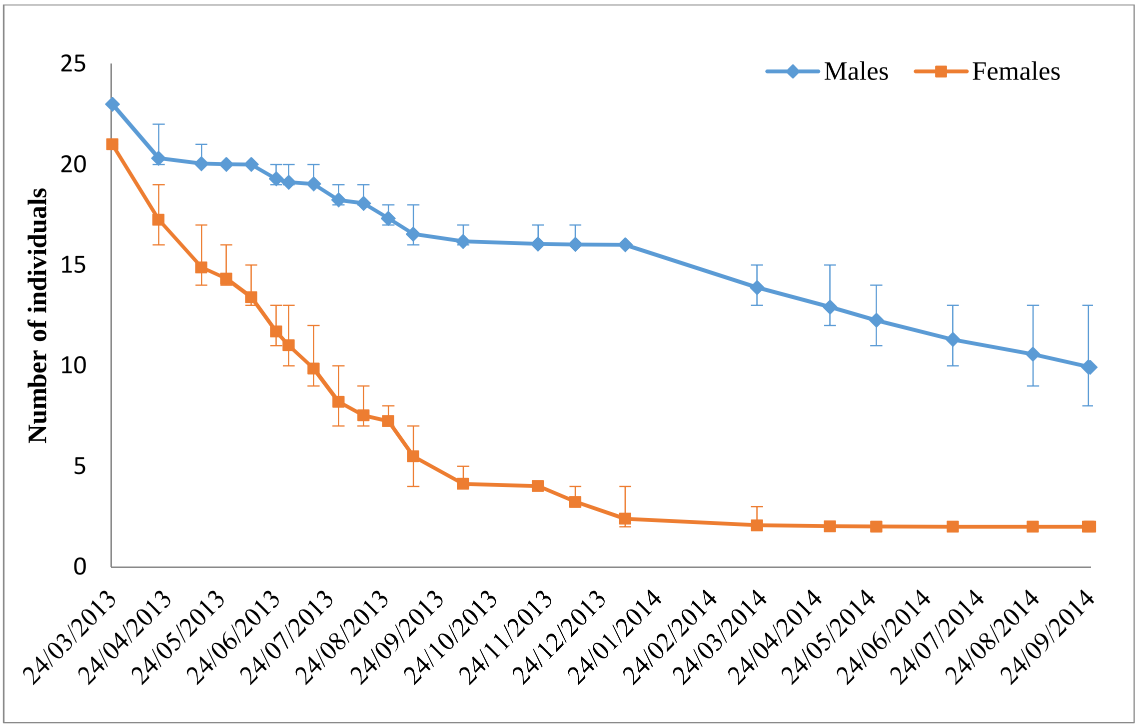

Figure 2.3. Survival of translocated hihi in Bushy Park from 23/03/2013 to 24/09/2014. The starting population was 21 females and 23 males. Error bars show 95% confidence intervals for numbers alive at each survey.

The best re-sighting model was clearly {4k} (Table 2.3), where the effect of sex was not included, and surveys could be grouped into 4 types according to survey methodology. The second best model incorporated sex effects {s+4k}, but had less support with Δ AIC$_c \approx$ 1.7. The estimated re-sighting probability was: 0.80 for the first four weeks of radio tracking (March – April 2013), 0.68 for the systematic surveys during the post-release period (May – September 2013), 0.70 for occasional surveys during the first breeding (October 2013 – February 2014), and 0.57 for the systematic surveys till second breeding (March – September 2014).

Table 2.3. A set of models fitted to hihi re-sighting data to determine whether re-sighting rates of hihi in Bushy Park were affected by sex (s), time (t) and survey type (4k). Models are ordered according AICc value, with the best model at the top.

Model	AIC$_c$	Δ AIC$_c$	ω_i	L	K	Deviance
Phi(a+1mo+6mo+s)P(4k)	904	0	0.58	1	10	603
Phi(a+1mo+6mo+s)P(s)	905	1.7	0.25	0.43	11	602
Phi(a+1mo+6mo+s)P(4k+s)	908	4.0	0.08	0.13	7	613
Phi(a+1mo+6mo+s)P(.)	909	5.5	0.04	0.06	28	568
Phi(a+1mo+6mo+s)P(t)	909	5.6	0.03	0.06	8	613
Phi(a+1mo+6mo+s)P(s*t)	911	7.4	0.01	0.02	29	568
Phi(a+1mo+6mo+s)P(s+t)	923	20.0	0.00	0.00	44	544

ω_i =AIC weights, L=Model likelihood, K=number of parameters.

Three global models, {Phi(a+1mo+6mo+s))P(4k)}, {Phi((a+1mo+6mo)+s(6mo)) P(4k)} and {Phi((a+1mo+6mo)+s(1mo))P(4k)}, fitted the data well. Based on the median c-hat test, the c-hat was almost exactly the same (≈1.19), regardless of which of the three models I used. The most parsimonious survival model was {6mo+s(6mo)}, indicating the presence of post-release effects over the first 6 months after release and sex-specific differences in survival only during this period. This model, as well as all models in a set, included the age-effect on fledgling survival (Table 2.5). Including an additional age effect or a 1-month post-release period (models {1mo+6mo+s(6mo)} and {a+6mo+s(6mo)}) increased the QAIC$_c$, indicating lower support (Table 2.4). Extension the sex effect to post-acclimation survival (i.e. after the first 6 months) and to juvenile survival also reduced model support (Δ QAIC$_c$ > 2.5). Survival estimates under the three best models were roughly the same for each sex and age class (Figure 2.4). However, in order to account for uncertainties in model selection, I used model averaging instead of using estimates from only one the most parsimonious model (Burnham & Anderson 2002). Averaged estimates for adults, fledglings and juveniles slightly decreased in comparison with estimates from the most parsimonious model. In contrast, estimates of survival during the 6-month post-release period were higher. This happened because model averaging takes into account not only the best models, but also other models in a set and, what is more important, increases uncertainty around the estimates so model selection uncertainty is accounted for.

Table 2.4. The set of models fitted to hihi survival data to determine whether survival rates of hihi in Bushy Park were affected by sex, age and post-release effects. Models are ordered according to AIC_c value, with the best model at the top.

Model	$QAIC_c$	Δ $QAIC_c$	ω_i	*L*	*K*	Deviance
Phi(6mo+s(6mo))P(4k)	756.73	0.00	0.34	1.00	8	504.76
Phi(1mo+6mo+s(6mo))P(4k)	758.64	1.91	0.13	0.38	9	504.58
Phi(a+6mo+s(6mo))P(4k)	758.81	2.08	0.12	0.35	9	504.75
Phi(6mo+s)P(4k)	759.28	2.55	0.09	0.28	8	507.32
Phi(s)P(4k)	760.68	3.95	0.05	0.14	7	510.80
Phi((a+1mo+6mo)+s(6mo))P(4k)	760.73	4.00	0.05	0.14	10	504.57
Phi(a+s)P(4k)	760.92	4.19	0.04	0.12	8	508.95
Phi(1mo+6mo+s)P(4k)	761.09	4.36	0.04	0.11	9	507.03
Phi(a+6mo+s)P(4k)	761.14	4.41	0.04	0.11	9	507.08
Phi(1mo+s)P(4k)	761.28	4.56	0.03	0.10	8	509.32
Phi(a+1mo+s)P(4k)	762.36	5.64	0.02	0.06	9	508.31
Phi(a+1mo+6mo+s)P(4k)	762.96	6.23	0.01	0.04	10	506.79
Phi(6mo)P(4k)	763.61	6.88	0.01	0.03	7	513.73
Phi(a)P(4k)	764.98	8.25	0.01	0.02	7	515.10
Phi(1mo+6mo)P(4k)	765.22	8.50	0.00	0.01	8	513.26
Phi(a+6mo)P(4k)	765.68	8.96	0.00	0.01	8	513.72
Phi((1mo+6mo)+s(1mo))P(4k)	765.71	8.98	0.00	0.01	9	511.65
Phi(a+1mo)P(4k)	766.26	9.54	0.00	0.01	8	514.30
Phi(1mo)P(4k)	766.59	9.86	0.00	0.01	7	516.71
Phi(a+1mo+s(1mo)P(4k)	766.75	10.02	0.00	0.01	9	512.69
Phi(.)P(4k)	766.96	10.23	0.00	0.01	6	519.15
Phi(1mo+s(1mo)P(4k)	767.06	10.33	0.00	0.01	8	515.10
Phi(a+1mo+6mo)P(4k)	767.30	10.58	0.00	0.01	9	513.25
Phi((a+1mo+6mo)+s(1mo))P(4k)	767.80	11.07	0.00	0.00	10	511.64

ω_i=AIC weights, *L*=Model likelihood, *K*=number of parameters.

Table 2.5. Monthly survival rates of hihi in Bushy Park estimated from the most parsimonious model {Phi(6mo+s(6mo))P(4k)}.

Age class and sex	Estimate	SE	95% Confidence Interval	
Translocated juveniles ♂	0.95	0.02	0.88	0.98
Translocated juveniles ♀	0.79	0.05	0.67	0.87
Adults/juveniles	0.96	0.02	0.91	0.98
Fledglings	0.77	0.10	0.53	0.91

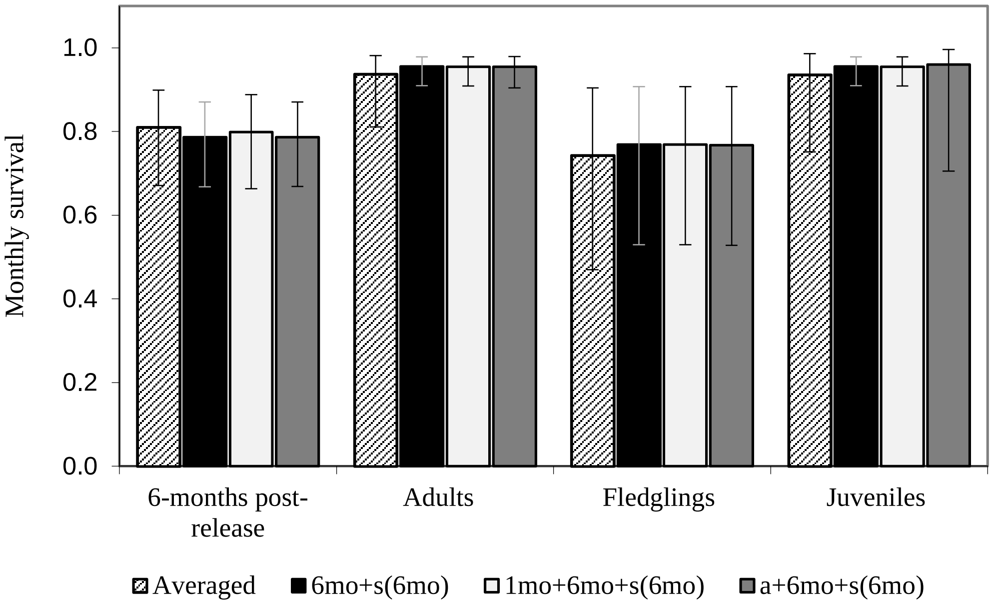

Figure 2.4. Comparison of monthly survival estimates of female hihi in Bushy Park from the three best models individually and from model averaging. Error bars show 95% confidence intervals.

Effect of sex. None of the best three models included a sex effect other than that during post-release period, but the fourth-ranked model included an overall sex effect. Consequently, under the averaged model, survival of females was estimated to be slightly lower than male survival in each age group, but this difference was only pronounced during post-release period, when monthly survival was 0.81 (SE=0. 06) for females and 0.94 (SE=0. 03) for males (Table 2.6).

Effect of age. The most parsimonious model did not include age as an effect except for fledglings up to March. However, the second best model {a+6mo+s(6mo)} included such an effect, with juvenile survival from March-September estimated to be slightly higher than adult survival. Under model averaging, juveniles and adults had approximately the same monthly survival probability as adults (0.96 for males and 0.94 for females); however standard errors for juvenile estimates were higher (Table 2.6) due to the high uncertainty in juvenile survival under model {a+6mo+s(6mo)}. Monthly survival of fledglings was 0.80 (SE=0.1) for males and 0.74 (SE=0.12) for females.

Post-release effects. All of the three best models included 6-month post-release effects, suggesting post-release effects had a significant impact on survival probability over this period. During the first 6 months after release, survival of both males and females was reduced, having the strongest effect on females. The second best model also included a 1-month post release effect, and under this model, survival was estimated to be lower over the first month than the subsequent five months. Consequently, under model averaging, survival was lower in the first month after release (0.93 (SE=0.04) for males, 0.79 (SE=0.08) for females; Table 2.6).

Table 2.6. Monthly survival rates of hihi in Bushy Park, estimated using model averaging.

Age class and sex	Estimate	SE	95% Confidence	
	Males			
0-1 months post-release	0.93	0.04	0.81	0.98
1-6 months post-release	0.94	0.03	0.86	0.98
Adult	0.96	0.02	0.91	0.98
Fledglings	0.80	0.10	0.54	0.93
Juveniles	0.96	0.03	0.86	0.99
	Females			
0-1 month post-release	0.79	0.08	0.60	0.90
1-6 months post-release	0.81	0.06	0.67	0.90
Adult	0.94	0.04	0.81	0.98
Fledglings	0.74	0.12	0.47	0.90
Juveniles	0.94	0.05	0.75	0.99

2.4 DISCUSSION

Modelling the survival of the translocated hihi population in Bushy Park showed that survival probability was reduced during the first 6 months after release. A difference in survival between sexes was observed during the 6-month post-release period and after; however, this difference was greater during post-release period.

Effect of sex. One of the main uncertainties in hihi survival in Bushy Park was the degree of difference between males and females. Modelling the survival probability showed that there was sex-specific difference in survival between adult males and females. Annual male survival was 0.61 (SE=0.15) and female survival was 0.46 (SE=0.22; standard errors were obtained using Delta Method following Armstrong and Reynolds 2012). This difference was much more pronounced during the first six months after release (March-September). The probability that translocated birds survive to first breeding (i.e. 6 months from March to September) was estimated to be 0.28 (SE=0.1) among females and 0.68 (SE=0.12) among males.

Bushy Park is not the first site where adult males and females have shown differences in survival after release. Female hihi had low survival immediately after release on Tiritiri Matangi Island: of 18 females released only four survived the first month, in contrast 12 males survived this period (20 males were released). Sex-specific difference in survival during the first months after release was probably the result of post-release stress. Laboratory studies on rats showed that males and females may show different physiological response to the same stressors, resulting in changes in social, foraging or anti-predator behaviour (Faraday 2002; Teixeira et al. 2007). Similarly, females and males of translocated European rabbits showed different survival in the first day after release in response to stress that was associated with different release techniques (Letty et al. 2000). When Jérôme Letty and colleagues used acclimatization pens for three days before release and provided rabbits with supplementary food, females survived better than males. In contrast, when rabbits were released immediately after translocation males performed better. It is also interesting that the effect of sex on survival was significant only when they used acclimatization pens. In the case of Bushy Park hihi, the birds were released immediately after translocation, as delayed release was found to reduce the long-term viability of birds (Richardson et al. 2013). Difference in females' survival as a response to release technique was also observed on Kapiti Island in 1991. Castro et al. (1994a) reported that 75% of immediate-release females survived first month, in contrast to only 50% of females that spent two weeks in aviaries before release. Apparently hihi females are more vulnerable than males to translocation stress, and this issue should be taken into account while planning translocation.

Natal dispersal is another possible explanation of sex differences in post-release survival of hihi. Richardson et al. (2010) found that juvenile female hihi dispersed significantly further than males on Tiritiri Matangi Island, with a decrease in the mean dispersal distance when the population became male-biased. Bushy Park is surrounded mostly by farmland, but there are also small pockets of bush and plantations within 100-500 m from the boundary that birds can use. Post-release dispersal is a possibility, however, the monitoring during the first month (radio-tracking) showed that none of the females left the release site; moreover there was also no apparent trend for increased dispersal among juvenile females born at Bushy Park.

Post-release effects. Previous translocations showed that hihi have different degrees and durations of post-release period. As was mentioned above, hihi had low survival immediately after release on Tiritiri Matangi Island. However, on Mokoia Island most of the birds (33 out of 40) survived the first two months and there were no significant post-release effects (Armstrong et al. 1999; Armstrong et al. 2002). 75% of released birds survived the first month on Kapiti Island (Castro et al. 1994b). In Bushy Park most of the birds survived the first month after release: out of 44 birds (21 females, 23 males) released in March 2013, 37 birds survived (16 females, 21 males). However, in the next 5 months the majority of females disappeared, leaving the population of 16 males and only 4 females for the first breeding season.

Translocation is certainly a stressful procedure for animals; however, in most cases it is not the stress itself that kills the animals. The stress, first of all, affects immune function and increases susceptibility to infectious disease. Not all similarly stressed individuals are equally likely to develop diseases – susceptibility may depend on differences in behaviour or biological vulnerability (Marsland et al. 2002). Hihi are known to be especially susceptible to *Salmonella* and *Aspergillus*, and such infections could be triggered by stress (Alley et al. 1999). Post-mortem analysis of hihi on Mokoia showed that aspergillosis was the main cause of mortality among birds recovered (Alley et al. 1999), and this can potentially be attributed to a high prevalence of *Aspergillus fumigatus* on that island (Perrott & Armstrong 2011). The cause of death of most hihi in Bushy Park is unknown, but post-mortem analysis of two male hihi showed the presence of *Aspergillus.* Not much is known about the distribution of

Aspergillus fumigatus in Bushy Park; however, some bird species like North Island saddleback *Philesturnus rufusater* and common blackbird *Turdus merula* tested positive for *Plasmodium elongatum*, *Plasmodium relictum*, *Plasmodium noyvella* and *Salmonella* (E. Schoener personal communication, December 21 2014).

Post-release effects are not only due to psychological stress in response to translocation procedure, e.g. catching, holding and transport, but also are associated with new environment and new predators that animals have to deal with. Failure to adjust to new conditions may mean failure to establish. For example, of the two native avian predators in Bushy Park – morepork and New Zealand falcon, the latter is absent on Tiritiri Matangi. Three females fell prey to these predators during the first month after release – it takes some time to learn how to avoid new predator. When planning translocations, it is difficult to predict what individuals will be less stressed and most successful in adjusting to a new environment (Armstrong & McLean 1995). For the most recent hihi translocations (Maungatautari and Bushy Park), juveniles were used. On the one hand, juveniles do not have as much experience as adults in foraging and avoiding predation, possibly resulting in high mortality during post-release. On the other hand, juvenile behaviour may make them more adaptable to new conditions, because during this stage they establish territories and develop foraging skills (Armstrong & McLean 1995). Masuda and Jamieson (2012) showed that sub-adult saddlebacks had lower post-release dispersal than adults, as adults often flew back to their territories at the source location. Sarrazin and Legendre (2000) suggested that adults should be used for translocations of long-lived species and juveniles for translocations of short-lived species. Understanding factors that could potentially be stressors for target species is very important and will help increasing the success of reintroduction projects (Parker et al. 2012).

Age effect vs post-release effects. Taking into account that the birds translocated to Bushy Park were juveniles, it was difficult to distinguish between the post-release effects and age effect during the first six months. However, the data collected on juveniles that were born in Bushy Park allowed these two effects to be distinguished, assuming conditions in the two years were fairly similar. Comparing survival of translocated juveniles and juveniles that were born in Bushy Park indicated that translocated juveniles experienced more than just the effect of age. The probability of a translocated juvenile female surviving from March to September 2013 was 0.28 (SE=0.1), while juvenile survival probability from March to September 2014 was 0.67 (SE=0.21). This difference was smaller in males, 0.68 (SE=0.12) for translocated males versus 0.76 (SE=0.13) for males born in Bushy Park, suggesting that post-release effects had greater impact on females. It is possible that post-release effects were not the only reason of reduced survival of translocated birds. Precipitation and temperature also were found to affect hihi survival even if they are not limited by food. Chauvenet et al. (2013) found that temperature had a small effect on survival when density was low and that high rainfall reduced juvenile survival. According to the Meteorological Service of New Zealand (MetService) the amount of rainfalls in the Manawatu-Whanganui region during March – September 2013 was not higher than the historical average.

Conclusion. Survival rates that were obtained in this chapter are extremely important for this reintroduction project, as they will help assess the initial success of the reintroduction and build demographic models to project the population growth and estimate its viability (Armstrong & Seddon 2008). These survival rates can be used to improve management of this population and reduce uncertainty in the response to further management action. For example, studying the effects of sex, age and post-release effects on survival shows the vulnerability of individuals at different stages after release, and could be used to project the consequences of follow-up translocations.

This kind of analysis can also help selecting individuals that show greater adaptation to a particular release site for future translocations. In addition, the survival rates of Bushy Park hihi can contribute to prior information when the decision about reintroduction to other sites needs to be made (Gedir et al. 2013). Such prior information incorporated into the model can make projections more precise.

Using these survival rates I wished to estimate the increase in viability of the Bushy Park population that could be achieved by follow-up translocations and compare it with the status quo scenario (Chapter 3). The projections will then be evaluated in collaboration with the group of stakeholders to guide the appropriate management action (Chapter 4).

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Chapter 3. Assessing short-term viability of the reintroduced hihi population in Bushy Park in the face of uncertainty

3.1 INTRODUCTION

Population viability analysis is one of the most powerful tools for assessing the value of follow-up translocation and harvesting. To predict and overcome the major uncertainties about system response to management actions, conservation managers can use predictive models that reflect their understanding of how populations function. Predictive models provide managers with information about what will happen under each management action and what will be the best decision to make considering the costs and benefits of each outcome (Runge 2011). Reliable predictive models incorporate uncertainties, because reliability of population predictions depends not only on unbiased estimation of vital rates (survival, reproduction and movement), but also on structural and estimation uncertainty (Tuljapurkar & Orzack 1980). Armstrong & Reynolds (2012) defined this type of uncertainty as *incertitude,* which reflects our limited knowledge of a system. On the one hand, most reintroduced populations are initially small, meaning data are insufficient for reliable estimation of parameters of interest. On the other hand, survival rates and reproductive success of reintroduced individuals may be initially reduced due to post-release effects or Allee effects (Armstrong & Reynolds 2012). An *Allee effect* refers to the process of vital rates being suppressed in some species when their population sizes are small, often due to effects on cooperation or facilitation among individuals (Stephens & Sutherland 1999; Courchamp et al. 1999). Therefore, monitoring should be directed to determine the presence of those effects in order to account for it in a projection.

Another type of uncertainty refers to variability in the state of a system and is called *stochasticity* (Armstrong & Reynolds 2012). Especially important for modelling the growth of small reintroduced populations is *demographic stochasticity* that refers to chance variations in individuals' reproduction or survival. This means that the future of a small population does not depend on the average estimator of vital rates, as it does in a big population, but instead it depends on the survival and fecundity of every single individual in the population (Caughley & Gunn 1996). That is why a small population is more likely to become extinct just by purely bad luck, even when other conditions, like food availability and absence of predators, are met (Caughley & Gunn 1996). However, as the population size becomes larger, the variation due to demographic stochasticity decreases.

A *follow-up translocation* or second release is one of the most common management actions that are used to increase population size and its viability (Armstrong & Ewen 2001). Veltman et al. (1996) found that multiple release increase the chance of exotic bird species establishing in New Zealand. This principle is also applied to reintroduction of New Zealand native birds such as the hihi *Natiomystis cincta*, North Island robin *Petroica longipes*, North Island saddleback *Philesturnus rufusater*, diving petrel *Pelecanoides urinatrix* and others (Parker 2013). Modelling the outcome of follow-up translocation involves less uncertainty than modelling initial translocation if monitoring was used to obtain new information about population parameters. However, even with good post-release data, predicting the dynamics of reintroduced population can be challenging due to uncertainty (Bar-David et al. 2005). Moreover, the potential benefits of follow-up translocation have to be weighed against the impact on the source population. Frequently only one source population is available for harvesting and the impact of follow-up translocation has to be carefully projected in order to maintain an acceptable level of harvesting.

In this chapter I perform PVA of the reintroduced hihi population in Bushy Park in order to see whether this population will be viable, both because the projection of population dynamics might be useful experience for future hihi translocations and because other interventions might be considered. There was a proposal from Bushy Park Trust to release more birds, with a view to increase persistence and population size. Consequently, the Hihi Recovery Group considered two management alternatives: status quo, meaning just continuing supplementary feeding and monitoring nest boxes during the breeding season; and follow-up translocation of 15 females from Tiritiri Matangi Island in March 2015. However, after Bushy Park pest monitoring revealed a serious incursion of two rat species in November 2014, a third alternative was considered – delaying the follow-up translocation to March 2016, allowing the effectiveness of pest management to be assessed before the translocation. I also wanted to consider a fourth alternative – “translocation of all juveniles from Tiri in March 2015”, in order to see how the increased number of translocated individuals would change the dynamics of both populations. All projections assume that rats were eradicated. I incorporated two types of uncertainties mentioned above into quantitative population models, and used these to make 10-year projections of population dynamics under each management alternative. In addition, I assessed the short-term impact of harvesting on the source population on Tiritiri Matangi Island. This population plays a critical role in hihi conservation and is highly valued, because it is usually the most suitable source for hihi translocation.

3.2 METHODS

3.2.1 Species and source population

Hihi are small endemic New Zealand forest birds. They are sexually dimorphic, feed on fruits, nectar and invertebrates, and nest in cavities of mature trees. Hihi formerly inhabited most of the North Island mainland and Great Barrier, Little Barrier (Hauturu) and Kapiti Islands, but by the 1880s had become extinct everywhere except Little Barrier Island (Taylor et al. 2005). Attempts to recover the hihi started more than 30 years ago (Taylor et al. 2005). Since then, hihi were reintroduced from Little Barrier Island to ten sites (five islands and four mainland sanctuaries, plus one site where they were held in captivity). Currently, populations remain at five of these sites: Kapiti Island, Tiritiri Matangi Island, Zealandia Wildlife Sanctuary, Maungatautari and Bushy Park. Experiments on food supplementation have shown that hihi survival (Armstrong & Ewen 2001) or reproduction (Armstrong et al. 2007; Chauvenet et al. 2012) is limited by food, thus all hihi are provided with supplementary feeding at all of these sites on a permanent basis. In order to compensate for the shortage of natural cavities, nest boxes are provided at Tiritiri Matangi, Zealandia and Bushy Park.

To minimise impact on the natural population on Little Barrier Island, most hihi are now sourced for translocations from the reintroduced population on Tiritiri Matangi Island. Tiritiri Matangi is a 254-ha island located in Hauraki Gulf and is managed by the Department of Conservation. A replanting program started on the island about 30 years ago (Mitchell 1985). Tiritiri Matangi Island is free of exotic mammalian predators: Pacific rats, or kiore *Rattus exulans*, the only non-native predators on the island, were eradicated with a poison drop in 1994 (Veitch 2002). The hihi population was re-established by two translocations (in 1995 and 1996) with the release of 53 birds in total from Little Barrier Island (Parker 2013). The Tiritiri Matangi population plays a critical role in hihi conservation in New Zealand. It is carefully monitored each year before and after breeding, providing data for modelling the impact of harvests.

3.2.2 Bushy Park

The most recent hihi reintroduction took place in 2013 in Bushy Park, where 44 juvenile hihi were translocated from Tiritiri Matangi Island. The Hihi Recovery Group approved Bushy Park for hihi reintroduction for a number of reasons. First of all it contains 87 ha of lowland rainforest that is believed to be in the hihi’s historical range. Secondly, 10 years ago mammalian predators were eradicated and the park was surrounded by a 4.8 km pest-proof fence (Xcluder™ "kiwi"). Thirdly, two successful reintroductions of predator-sensitive native species - North Island robin and North Island saddleback – indicate the good quality of this site. Finally, Bushy Park is a community based conservation project, which has a big team of volunteers who help to monitor birds and maintain feeders and nest boxes. However, there are also some challenges in species management at Bushy Park. In order to protect the reintroduced populations, the Bushy Park Trust conducts on-going pest monitoring. This includes fence monitoring as well as regular replacement of baited traps for rodents and possums inside the park. However, often outside the fence non-native predators are present at high density, and they can discover fence damage very quickly (Connolly et al. 2009). When pest monitoring is not regular, which could be the case in community- based conservation, non-native predators may reach high density inside the fence before being discovered. There are also two native predators of birds in Bushy Park: morepork *Ninox novaeseelandiae* and New Zealand falcon *Falco novaeseelandiae.*

3.2.3 Monitoring at Bushy Park

I obtained survival data by doing seven monthly surveys from March 2014 till September 2014 and by analysing data that were collected in the period of March 2013 - February 2014 by Bushy Park volunteers, including the first month when daily monitoring was conducted using radio tracking (the battery life was approximately one month; Chapter 2). Each survey took two days on average and involved walking along monitoring lines and watching the feeders for one hour each. The 44 individuals were colour-banded, as well as about 80% of the juveniles produced in the first (2013/2014) breeding season. Nest monitoring was conducted regularly during the breeding season. Although nest boxes were checked systematically, some nests were never discovered and at least four juveniles left their nests unbanded. These birds were not used in the survival analysis, but were used for modelling fecundity.

3.2.4 Developing the population model for Bushy Park

I combined the reproduction and survival data into one analysis, in order to achieve more precision and accuracy in vital rates estimates and prediction (Abadi et al. 2010). The population model was used to predict population growth and risk of extinction under different management alternatives. I used a female-only model, as they are the sex likely to limit population growth (Figure 3.1). Uncertainties related to demographic stochasticity, model selection and parameter estimation were incorporated into the model. I did not include immigration or emigration in the model, as this population was determined to be geographically closed (radio tracking in the first month after release showed that no birds left the population). The population model was coded in OpenBUGS software that uses Bayesian updating and allows modelling with a range of useful distributions (McCarthy 2007; Kéry & Schaub 2012). The model includes three steps: modelling fecundity, deriving survival rates, and simulating population dynamics based on the estimated vital rates.

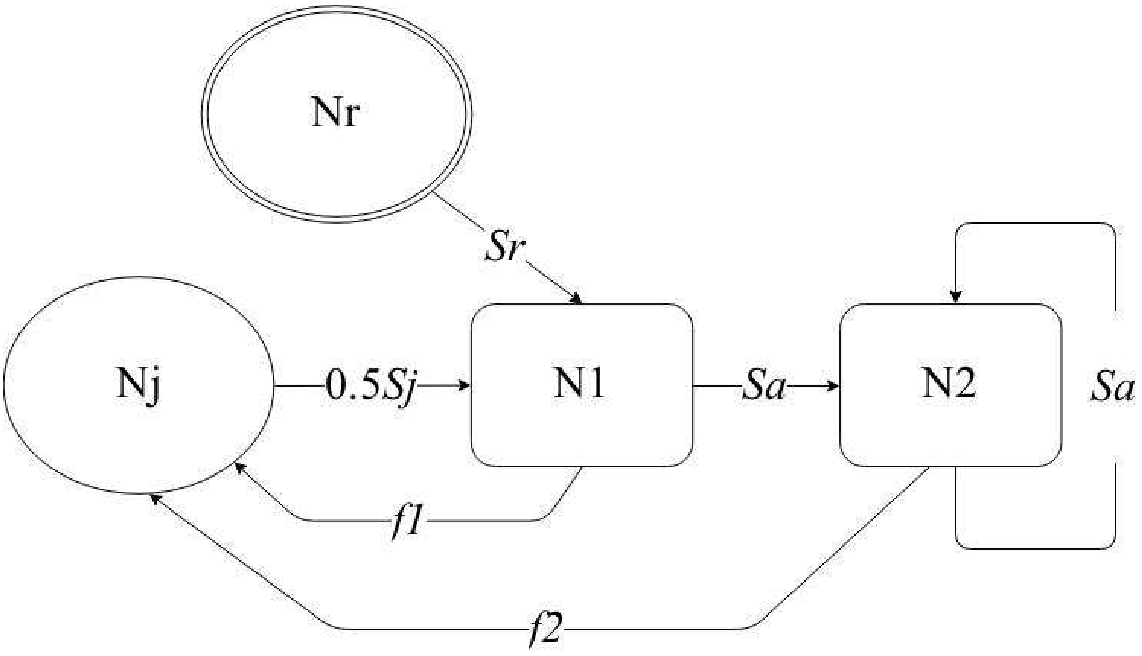

Figure 3.1. Female-only life-cycle graph of the hihi population in Bushy Park. The model has a pre-breeding census, meaning that the numbers of birds are counted just before breeding starts. Rectangles represent adult females of two age classes (N1, number of first-year adults; N2, number of older adults), and ellipses represent juveniles, which consist of juveniles fledged at Bushy Park (Nj) and translocated (or released) juveniles (Nr). The arrows represent the transition probabilities based on the vital rates (*Sr,* survival of translocated juveniles; *Sj,* juvenile survival; *Sa,* adult survival probability; *f1,* mean fecundity of first-year females; *f2,* mean fecundity of older females).

Step 1: Modelling fecundity. Fecundity (the number of fledglings per female per year) was taken to be Poisson distributed, and included a fixed age effect that was modelled on a logarithmic scale with informative priors. I assumed that mean fecundity of first-year females in Bushy Park would not be higher than that of first-year females from the source population (b_0f), thus the parameter b*fp*, the difference in mean fecundity between the Bushy Park and Tiri populations, was constrained to be ≤ 0. I did not have reproduction data for older females in Bushy Park. Therefore, I modelled fecundity of older birds completely based on the estimated age effect (b*fa*) at the source population, i.e. the difference in mean fecundity between older females and first-year females (see below estimation for Tiri). Linear models that were fitted to the fecundity data were

$$\log(f1) = b_0f + bfp$$

$$\log(f2) = \log(f1) + bfa$$

where $f1$ and $f2$ are mean fecundity of first-year females and older females in Bushy Park.

Step 2: Deriving survival estimates. I modelled survival in program MARK 7.2 using the Cormack-Jolly-Seber (CJS) model, considering alternative models included sex, age and post-release effects (Chapter 2). In order to account for estimation uncertainty in OpenBUGS, survival for each age class was modelled on a logit scale and parameters were randomly sampled from a normal distribution defined by the mean that was equal to averaged survival estimates and precision that was obtained based on standard error $(SE\ [logit(S)])^{-2}$. Averaged survival estimates with unconditional standard errors were used in order to allow for structural uncertainty (Chapter 2).

Step 3: Linking population dynamics to vital rates. Population dynamics were assumed to be determined by the following parameters: mean fecundity (number of fledglings per female per year) of first-year females, mean fecundity of older females, juvenile survival (probability of a fledgling surviving to the next breeding season) and annual adult survival. The probability of a fledgling being a female was assumed to be 0.5, which is supported by a large data set from the Tiri population.

The expected number E(N) of adult females next year in each age class can be represented mathematically as:

$$E(N1_{t+1}|N1_t, N2_t) = 0.5Sj(N1_t * f1 + N2_t * f2) + Nr * Sr$$

$$E(N2_{t+1}|N1_t, N2_t) = (N1_t + N2_t) * Sa.$$

The above equation is a deterministic version of the population model. To account for stochasticity uncertainty, I included demographic stochasticity in this model. The number of new-born juvenile females (Nj) each year was sampled from a Poisson distribution based on the expected number, which was based on the number of females in two age classes (N1 and N2), with an expected probability of being female set at 0.5:

$$Nj_t \sim Poisson\left(0.5 * (N1_t * f1 + N2_t * f2)\right).$$

The number of first-year hihi (N1) alive at first breeding was sampled from a binomial distribution based on the juvenile survival probability (*Sj*). I assumed birds to be juveniles until September.

$$N1_{t+1} \sim Binominal(Nj_t, Sj).$$

In the case of follow-up translocation, the number of translocated females (Nr) survived to the first breeding was sampled from a binomial distribution with probability of success equal to estimated survival and was added to the total number of juveniles.

$$N1_{t+1} + \left(Nr \sim Binominal(Nr, Sr)\right).$$

The number of older females (N2) the next breeding season was sampled from a binomial distribution based on the annual adult survival probability:

$$N2_{t+1} \sim Binominal(N2_t, Sa).$$

Extinction probability was estimated by using the function {step(-Nt)} in Open BUGS, which takes on the value 1 (extinct) if the variable Nt, in parenthesis, greater or equal to 0 and otherwise takes on 0 (persisting).

Carrying capacity. Estimating the carrying capacity for hihi in Bushy Park is not straightforward, first of all because this population has only started establishing and secondly because it is supported by supplementary feeding. The main idea of including carrying capacity in this study was to avoid unrealistic growth in the projections. I used elicitation of expert opinion of the Hihi Recovery Group (HRG) members to assess the carrying capacity at Bushy Park, taking into account the most likely management capacity as well as characteristics of the reserve. The process was initiated at the HRG meeting in 2014 and concluded via email, with five HRG members ultimately completing the questionnaires that were based on a modified Delphi method (MacMillan Marshall 2006). The average upper limit of the carrying capacity ranges given by these members was 189 individuals. For simplicity, I rounded this number to 200 and divided by 2, as sex ratio is assumed to be 1:1. In order to limit the population to 100 females I used function {min} in OpenBUGS and applied it to truncate the number of first-year females to:

$$N1_t = \min(K - N2_t, N1max_t),$$

where K = carrying capacity, $N1max_t$ = the expected number of first-year females at time *t*, and $N1_t$ = the truncated number of first-year females.

I ran the model with two sets of initial parameter values in order to check for convergence (McCarthy 2007), and it always converged to a posterior distribution within 1000 iterations. After burn-in of first 1000 iterations I ran each model 100000 times.

3.2.5 Modelling the impact of harvesting on the source population

In order to simulate the Tiritiri Matangi population I used a model that was previously developed in Microsoft Excel to guide harvesting from Tiritiri Matangi hihi population since 2005 (Armstrong & Ewen 2013). Its predictions have been tested during harvesting for translocations to Zealandia Wildlife Sanctuary, Ark in the Park and Maungatautari. Like the Bushy Park model described above, this model is female-only and incorporates uncertainties related to demographic stochasticity and parameter uncertainty. The model goes through several steps for each run of the model:

1. Number of females. Based on the re-sighting probability, it determines the number of juvenile and adult females that were missed in surveys. Based on the annual adult survival probability, it determines how many females survive the next 12 months.
2. Number of fledglings. Based on the probabilities of first-year and older females having ≥ one fledgling and the mean numbers of additional fledglings per female, it determines the total number of fledglings.
3. Number of juveniles and harvesting. Based on the monthly juvenile survival probabilities, it determines how many juveniles survived each month before and after harvesting.
4. Sexing. It determines how many juveniles that survived to breeding were females.

This model uses demographic parameters (survival and fecundity) that were estimated based on data collected from 1996 - 2004 (Thorogood et al. 2013; Table 3.1). These demographic rates have been stable so far and given accurate projections (Armstrong & Ewen 2013). The numbers of first-year birds and older birds were obtained during pre-breeding survey in September/October 2014.

The aim of this simulation was not to make long-term projections, but to model the probability distribution of the number of females after harvesting in September 2016. It is hard to project long-term dynamics of the Tiritiri Matangi hihi population because there will probably be further harvests, and the number of individuals harvested varies from year to year depending on circumstances. Since 2005 the number of birds removed annually to other conservation areas has varied between 0 and 72 (Parker 2013). Thus the mean number of female hihi in September 2016 serves as an indicator of the impact of the proposed follow-up translocation to Bush Park. Similar guidance has been frequently given for setting harvest rates in previous years; however, this is normally expressed as how many juveniles can be harvested such that we maintain an adult female population of about 70 (Armstrong & Ewen 2013).

Table 3.1. Demographic parameters for the Tiritiri Matangi hihi population based on the data from 1996 – 2004 (from Thorogood et al. 2013).

Demographic parameter	Estimate	SE
Juvenile survival	0.43	0.02
Annual adult survival	0.69	0.03
Fecundity of 1st-year female	2.27	0.23
Fecundity of older female	3.46	0.21
Sex ratio	1:1	

In order to project the alternative “translocation all juveniles” this model was rewritten in OpenBUGS and combined with Pushy Park population model (Appendix B). It allows me not only to incorporate the number of juveniles on Tiri in March 2015 to Bushy Park model, but also to account for uncertainty in this estimate.

3.3 RESULTS

3.3.1 Bushy Park population

Fecundity and survival. Monitoring over the 2014/15 breeding season showed that all four females known to survive to the breeding season attempted to nest. They had at least seven clutches (five of which were successful) and fledged at least 16 chicks. Two of these females had high productivity for first-year birds, they both produced six fledglings.

Uncertainty around fecundity estimates was quite large (Table 3.2). With uninformative (but constrained) priors, the posterior distribution for fecundity of first-year females had a mean of 2.53 (SE=0.40) fledglings per female, with a 95% credible interval (CRI) ranging from 1.70 – 3.28. Fecundity of older females had a mean of 3.71 (SE=0.71) and 95% CRI of 2.37 – 5.16. Such wide credible interval is expected as the sample size of first-year females was very small and I did not have Bushy Park data on older females. Survival estimates for the model in OpenBUGS were derived from those obtained in MARK (Table 3.3)

Table 3.2 Parameter estimates for the Bushy Park hihi population obtained from OpenBUGS. Fecundity parameters were modelled in OpenBUGS; survival parameters were first modelled in program MARK then used in the population model in OpenBUGS.

Parameters	Estimate	SE	95% CRI	
Annual adult survival	0.45	0.19	0.08	0.79
Prob. fledglings survive January – February	0.54	0.16	0.22	0.82
Prob. juvenile survive March – September	0.63	0.20	0.18	0.92
Prob. translocated female survive first month	0.78	0.08	0.61	0.90
Prob. translocated female survive April-September	0.35	0.12	0.14	0.59
Fecundity of first-year female	2.53	0.40	1.70	3.28
Fecundity of older female	3.71	0.71	2.37	5.16

Table 3.3 Monthly survival rates of female hihi in Bushy Park, estimated by model averaging in program MARK (see Chapter 2 for models considered).

Age class and sex	Estimate	SE	95% Confidence	
0-1 month post-release	0.79	0.08	0.60	0.90
1-6 months post-release	0.81	0.06	0.67	0.90
Adult	0.94	0.04	0.81	0.98
Fledglings	0.74	0.12	0.47	0.90
Juveniles	0.94	0.05	0.75	0.99

Population dynamics. I modelled population dynamics under four scenarios based on the results above: no translocation, translocation of 15 females in 2015, translocation of all available juveniles from the Tiritiri Matangi population in 2015, and translocation of 15 females in 2016. The means and medians show different trends over time – whereas the mean goes up over time, the median actually slowly declines (Figure 3.2). This occurs because the distributions of population size becomes progressively right skewed over time, meaning that lower numbers occur more often than higher ones, and the most likely population size (mode) is less than the mean.

Uncertainty around population size is quite large with very wide CRI (Figure 3.3). The median number of females in September 2024 is 2 with no translocation, 5 with translocation of 15 females in either 2015 or 2016, and 8 with all juvenile females being translocated in 2015 (38 juvenile females on average), with 95% CRI= 0 – 100 for each alternative. Although follow-up translocations increase population size, uncertainty about population size still remains very large. Taking into account both parameter uncertainty and demographic stochasticity, the probability of extinction within 10 years under alternative "no translocation" is 44% (Figure 3.4). The probability of extinction over 10 years decreases to 35% under alternative "translocation 15 in 2015", to 31% under alternative "translocation all in 2015" and to 34% under alternative "translocation 15 in 2016".

It is also interesting to compare the immediate benefits of releasing birds in the Bushy Park population. All translocations minimize the risk of extinction in September 2016. In contrast, the probability distribution of the alternative "no translocation" has a greater probability of zero females (Figure 3.5).

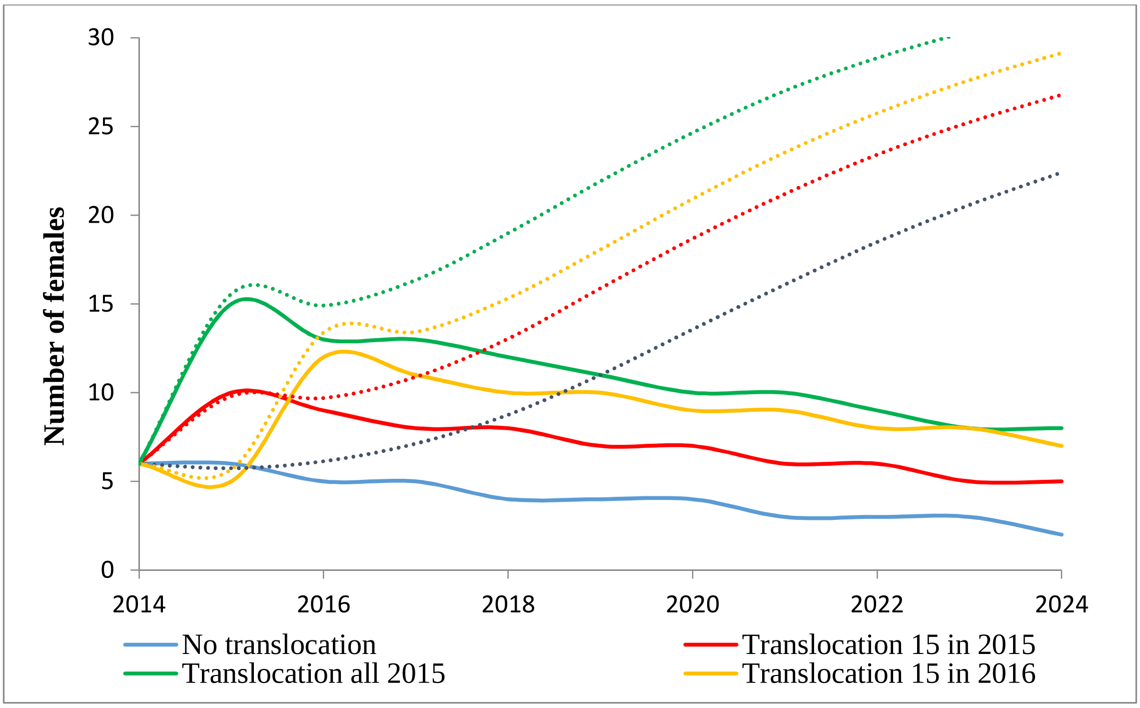

Figure 3.2 Projections of hihi population growth in Bushy Park over 10 years under four alternatives: no translocation, translocation of 15 females in March 2015, translocation of all juvenile females in March 2015, and translocation of 15 females in March 2016. Solid lines show median numbers of female, and dotted lines show mean numbers of females.

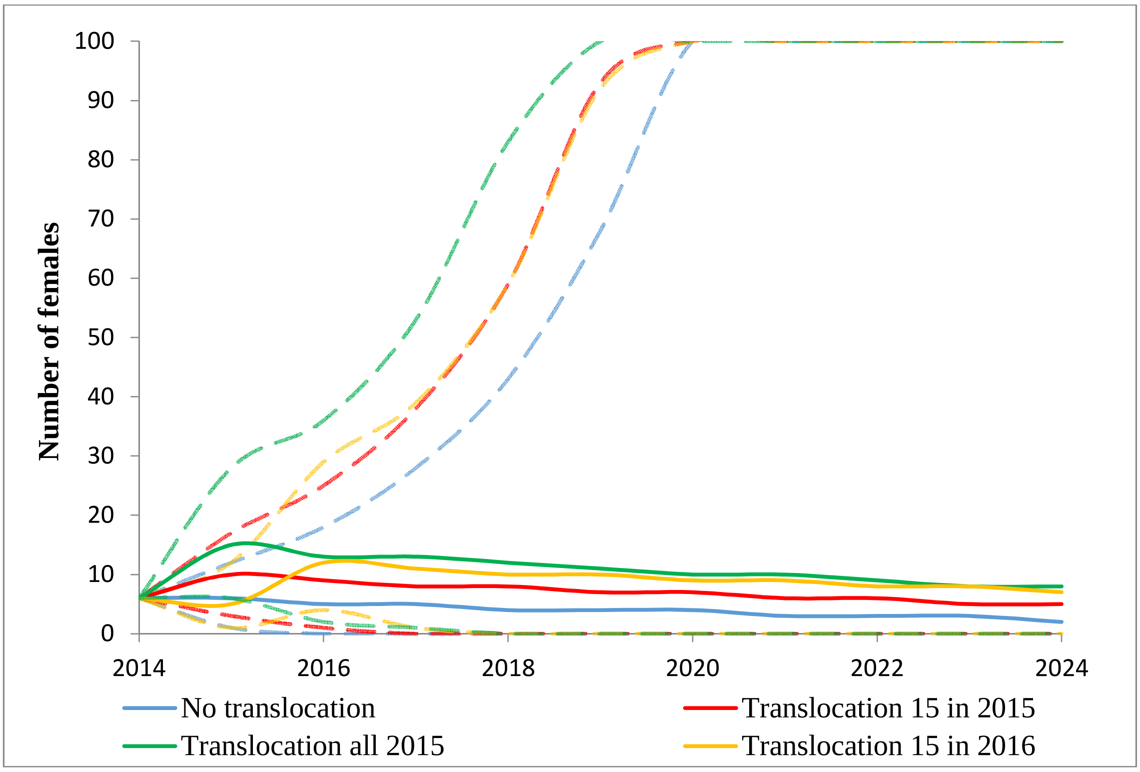

Figure 3.3 Projections of hihi population growth in Bushy Park over 10 years under four alternatives: no translocation, translocation of 15 females in March 2015, translocation of all juvenile females in March 2015, and translocation of 15 females in March 2016. Solid lines show median numbers of female, and dashed lines show 95% credible intervals.

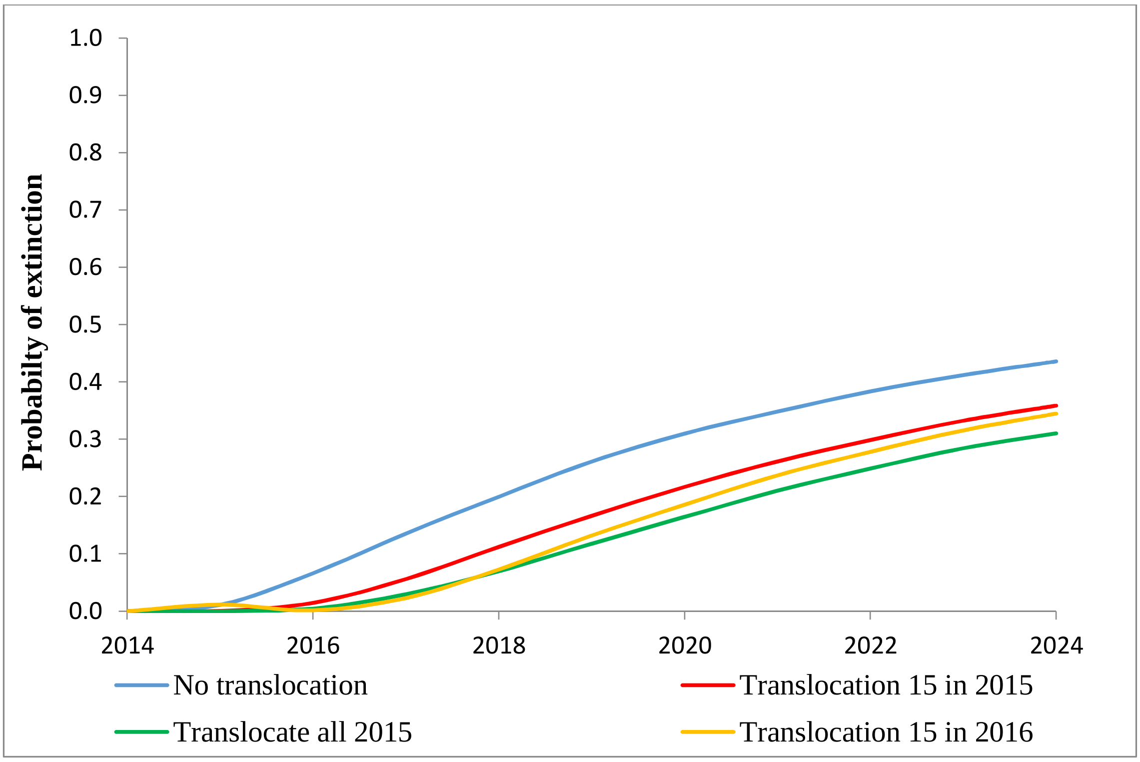

Figure 3.4 Mean probabilities of hihi population extinction in Bushy Park in ten years under four alternatives: no translocation, translocation of 15 females in March 2015, translocation of all juvenile females in March 2015, and translocation of 15 females in March 2016.

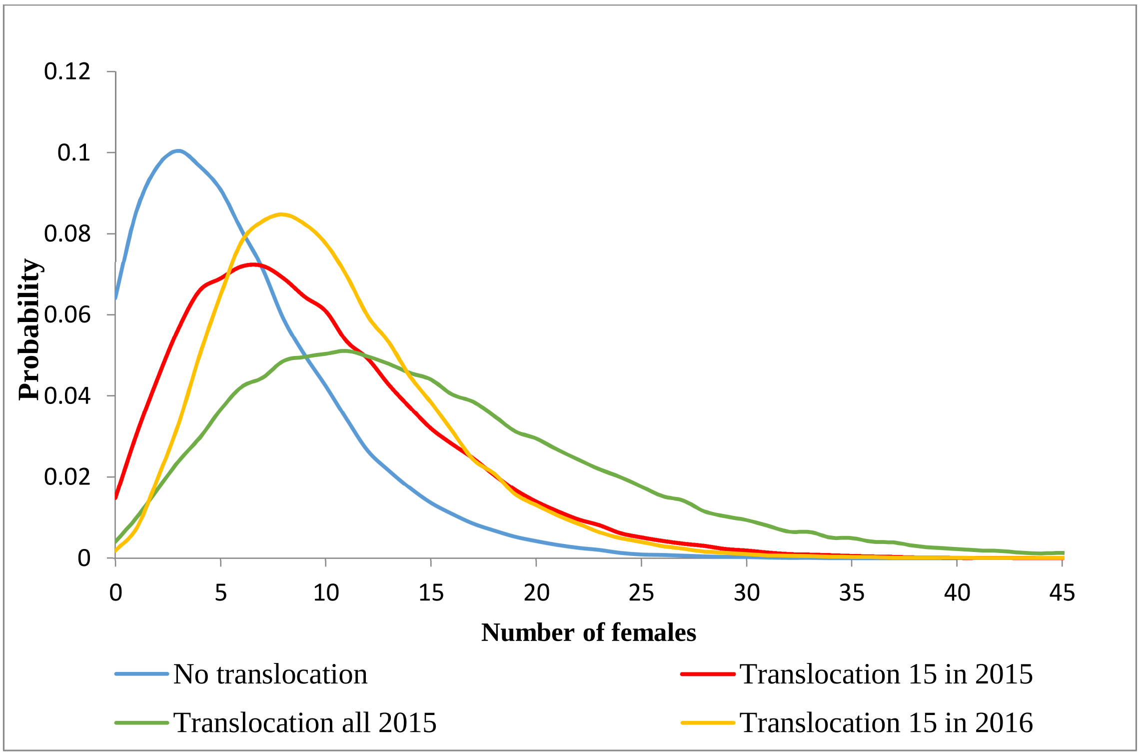

Figure 3.5 Probability distributions for number of female hihi in Bushy Park in September 2016, under four alternatives: no translocation, translocation of 15 females in March 2015, translocation of all juvenile females in March 2015 and translocation in March 2016.

3.3.2 Modelling the impact of harvesting on the source population.

The projections for the Tiritiri Matangi population started in September 2014 with 13 known first-year birds and 15 known older birds. As for Bushy Park, I modelled 4 scenarios: no harvest, harvest of 15 juvenile females in March 2015, harvest of all juveniles from the population in March 2015, and harvest of 15 juvenile females in March 2016. Modelling the number of female hihi in September 2016 indicates that Tiri population is expected to decline slightly from harvesting. Under the alternative “no translocation”, the mean number of females in 2016 is 68, with a 95% credible interval (CRI) of 42-103. Translocation decreases this number: to 55 (95% CRI = 31-88) for alternative “translocation 15 females in 2015”, to 34 (95% CRI = 19-52) under alternative “translocation all juveniles in 2015” and to 58 (95% PI = 33-93) under alternative “translocation15 in 2016”. The probability of extinction is close to 0 under all four scenarios (Figure 3.6).

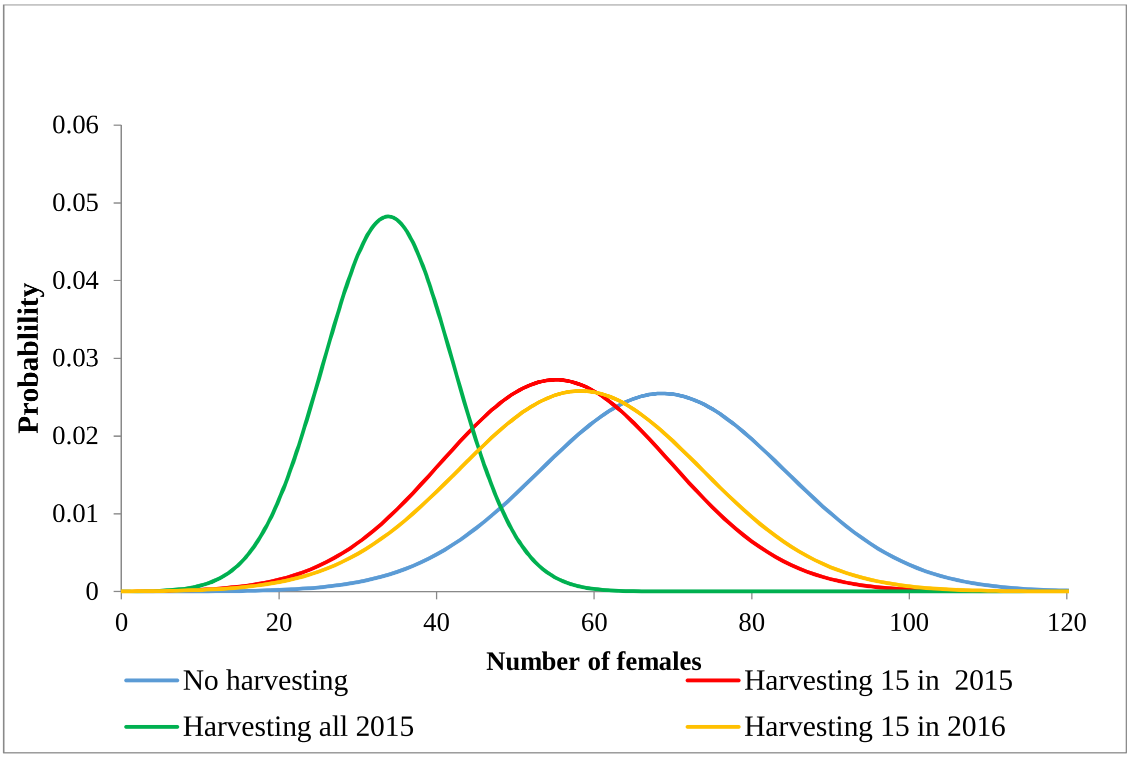

Figure 3.6 Probability distributions for number of female hihi on Tiritiri Matangi Island in September 2016, under four alternatives: no harvest, harvest of 15 females in March 2015, harvest of all juvenile females in March 2015, and harvest of 15 females in 2016.

3.4 DISCUSSION

The most obvious conclusion from modelling the Bushy Park population is that it shows a slight negative tendency no matter what management alternative is chosen, but with great uncertainty, as the 95% CRI for the number of females in 10 years ranges from 0 to 100 females (with 100 set up as a carrying capacity for Bushy Park) under each alternative. This means that I cannot claim with any confidence that this population is going to become extinct or survive the next 10 years. This uncertainty relates mainly to uncertainty in survival and fecundity estimates. This uncertainty is an integral part of reliable population projection (Boyce 1992), as each value within the range of uncertainty has to be considered as a possible true population parameter (Clark 2003).

Fecundity and survival. Vital rates are one of the most important parts of reliable population forecasting. Under- or over-estimated vital rates lead to significant changes in projections and as a result to inappropriate management decision. For example, underestimating the fecundity rate of translocated North Island robin on Tiritiri Matangi Island resulted in an unnecessary follow-up translocation (Armstrong & Ewen 2001). Further monitoring showed that the mean fecundity rate of robins was higher than that estimated based on one year of post-release data from 7 females (Armstrong & Ewen 2001). Sufficient sample size, consideration of post-release effects and post-release monitoring are important. However, use of available prior information may help to obtain reliable fecundity estimates, even based on one year of post-release data (Gedir et al. 2013). The fecundity estimate that I obtained was based on the 4 females that survived to the 2013/2014 breeding season. Moreover, 75% of the fledglings (12 of 16) were the offspring of two females. Consequently, I believe it was reasonable to make use of the prior information on fecundity of females from the Tiritiri Matangi Island population, allowing me to keep the mean fecundity of first-year females from Bushy Park within realistic limits. Prior information was also necessary to model fecundity of older females, as no data for Bushy Park were available yet. It was known that fecundity of hihi is age-specific, with first-year females starting nesting later and consequently producing fewer clutches (Armstrong et al. 2002; Armstrong et al. 2007; Low et al. 2007).

Simulation of the Bushy Park population. For modelling population growth and probability of extinction, I applied an integrated framework, meaning different types of data were analysed simultaneously (Abadi et al. 2010). However, my model was not fully an integrated model, as I modelled survival separately in program MARK. Kéry & Schaub (2012) suggest that as long as Goodness-of-fit test for integrated population models has not been developed yet, one should compute it for each data set. Modelling survival separately was also essential because it allowed incorporating model selection uncertainty, by model averaging. Bayesian model averaging is complex and under development (Barker & Link 2013). Moreover, it would probably require using multinomial likelihood, instead of state-space likelihood for survival, which is not so flexible (Kéry & Schaub 2012).

The combined model allowed me to reconcile uncertainty in parameter estimates with uncertainty in population growth prediction. On the one hand, parameter uncertainty incorporated into the model widens prediction intervals and reduces the confidence in a projection. On the other hand it is more likely to include true population parameter than one where uncertainty is ignored (Wade 2002; Clark 2003). This fact is especially important when the projections are used as a basis for making conservation management decisions, because managers always need to be aware of the risks that they undertake.

Modelling indicates that follow-up translocations would slightly reduce the probability of extinction, especially during the first three years when the mean probability of extinction is less than 5%. My model does not incorporate the Allee effect or genetic variation, so the reduced extinction probability can be explained only by decreasing demographic stochasticity and increasing sample size. There were no reasons to expect an Allee effect in hihi (D. Armstrong., personal comments, January 2015). However, follow-up translocation may have a long-term benefit of increasing genetic variations and decreasing inbreeding. The model that I used serves to predict the short-term persistence of the population, whereas loss of genetic variation and inbreeding would be relevant to longer-term modelling. These processes along with demographic stochasticity are important concerns for small translocated population (Keller et al. 2012). Brekke et al. (2010) found that inbreeding in the hihi population on Tiritiri Matangi results in a hatching failure and depressed male survival in early life stages. Although the genetic issue is important, Boyce (1992) claims that incorporating genetics in models will not affect the projection as much as incorporating demographic stochasticity.

Density dependence will also affect short- or long-term projections of population dynamics, depending on the habitat area available. The Bushy Park population has only been studied for a short time and at low density, so it is impossible to infer density dependence from that population. However, in other hihi populations some density dependence has been recently discovered: Ewen et al. (2011) found a slight reduction in fecundity as density increased on Tiritiri Matangi Island, and Chauvenet et al. (2012) found that recruitment declined with density on Kapiti Island. Ewen et al. (2012) also found that the proportion of females among recruits decreased as density increased. In my projections, I just applied a fixed population ceiling in order to avoid unrealistic population growth; however, more data are needed to better understand the process of density dependence on each site.

Studying the mechanisms underlying population dynamics, for example the effect of interspecific food competition and predation on vital rates, could potentially contribute to long-term projections in future. For example, the population of bellbirds *Anthornis melanura* probably increased after the supplementary food that was provided to hihi on Kapiti and Tiritiri Matangi Island. Now bellbirds make up a majority of birds feeding on the feeders. This growing bellbirds population can potentially displace hihi from natural nectar sources in the future. Another example is the most recent discovery of a rat incursion in Bushy Park. This suggests that the risk of non-native predator invasions should be incorporated in the population models, even if the reintroduction site is fenced and well monitored.

Impact of harvesting. I modelled the probability distribution for the number of females on Tiritiri Matangi in September 2016. Long-term projections for the Tiritiri Matangi population are impossible for a number of reasons. First of all, the annual harvest rate for this population is not constant and hard to predict: almost every year since 2005 (except 2006, 2012 and 2014) from 30 to 72 individuals have been removed to different conservation areas including Bushy Park (Parker 2013). Secondly, the translocated birds are predominantly juveniles, and most of the time harvesting takes place at the beginning of the year, making it difficult to model juvenile survival. Thirdly, little is known about carrying capacity of Tiritiri Matangi Island and the effect of density on demographic parameters in hihi populations. Beissinger and Westphal (1998) argue that models that do not incorporate density dependence yield unreliable extinction probabilities, which could be either under- or overestimated. Boyce (1992) also recommends including density dependence unless projections are short-term. Another concern about long-term projections for Tiritiri Matangi population is the sex ratio. The last survey in September 2014 revealed that Tiritiri Matangi population has become clearly male biased, with more than three males per female (J. Ewen pers. comm.), and although Ewen et al. (2011) found that male bias in the sex ratio has little effect on female survival and fecundity, this issue needs further investigation in order to understand how sex ratio will be changing in the future on Tiritiri Matangi Island and at different sites and how this should be factored into decisions about harvesting rates.

Conclusion. On the one hand, the population projections for Bushy Park indicate that follow-up translocation will not have a great impact on population viability. On the other hand, projections for Tiri also indicate that the proposed removal of females will not put that population at risk. The final management decision is expected to follow the recommendation of the Hihi Recovery Group (HRG) and depends on how they evaluate the benefits of follow-up translocation for Bushy Park population against the costs of harvesting for Tiri population. In cooperation with the HRG, I wished to decide whether the reduction in probability of extinction at Bushy Park from 45% to 35% is sufficient benefit to justify the losses incurred by the Tiritiri Matangi population and the cost of the follow-up translocation. In order to make appropriate management decision that will satisfy stakeholders I applied Structured Decision Making to this problem (Chapter 4).

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Chapter 4. Applying Structured Decision Making to the reintroduced hihi population in Bushy Park

4.1 INTRODUCTION

Reintroduction is a complex process, which requires cooperation between conservation managers and stakeholders to make management decisions before and after translocation (Converse et al. 2013a). Before translocation, managers need to decide what release site will be appropriate, how many individuals will be translocated, the origin of individuals, and what release method should be used. After release, managers need to decide what management to apply in order to increase population viability. Some of these decisions may be very complex and controversial. For example, in the 1990s reintroduction of the grey wolf *Canis lupus* generated a lot of debate between biologists, who argued that wolves were the last missing link in the Yellowstone National Park ecosystem, and adjacent landowners of the Park, who opposed the presence of a notorious predator near their settlements (Wilson 1997). Attempts to save another iconic species, the California condor *Gymnogyps californianus*, generated debate between conservation managers, scientists and environmental activists about whether or not to remove the remaining birds from the wild and start a captive breeding program (Alagona 2004). Difficulties about decision making also relate to our limited knowledge about system response, which means that the outcome of reintroduction is often uncertain. Thus the main question is how to make reasonable management decision in the face of uncertainties, and find a compromise between multiple opinions.

The framework for making rational decision about the numerous components of reintroduction is Structured Decision Making (SDM; Figure 4.1). SDM is a transparent process that guards decision makers against the potential pitfalls in the decisions that they are going to make.

Recently this approach has become more popular in natural resource management decisions (Gregory & Keeney 2002). However, there are only a few examples of SDM being applied in reintroduction biology. For example, Lauber and Knuth (1997) incorporated some elements of SDM by involving citizens in the decision making process about moose reintroduction to New York; Converse et al. (2013) used SDM to decide whether or not releasing captive-reared whooping crane chicks *Grus americana* would help to recover the non-migratory population in Florida, USA; and Ewen et al. (2014) applied SDM to guide management decisions about the supplementary feeding regime for hihi *Notiomystis cincta* on Kapiti Island, New Zealand.

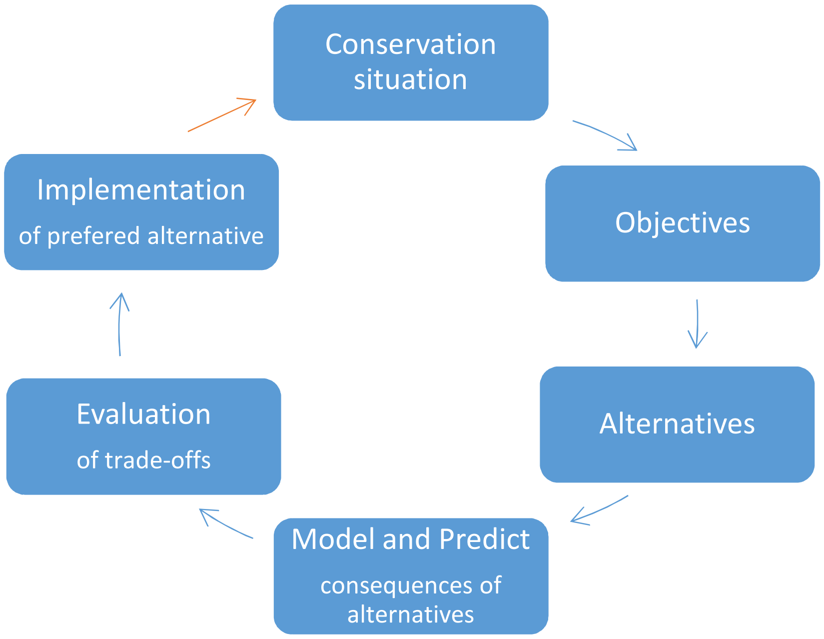

Figure 4.1 Structured decision making. The red arrow between the last and the first steps of SDM convert this scheme into an adaptive management cycle, which is a subset of SDM.

The rationale of SDM is to decompose a complex problem into smaller steps, focus on objectives while making decisions, overcome the common human errors in judgement, and involve stakeholders at each step of the decision making process (Figure 4.1). In this Chapter, I showed how SDM may be used to inform a decision about post-release management, and showed how the diverse interests of stakeholders may be incorporated in decision analysis.

4.2. APPLYING SDM TO POST-RELEASE MANAGEMENT

The success of a species' establishment after release can be evaluated based on the results of post-release monitoring, and further management can be adapted based on this new knowledge. Converse et al. (2013b) defined three main groups of management actions that are applied after release to increase population viability: 1) population reinforcement, which here I call follow-up translocations (Armstrong & Ewen 2001); 2) habitat management, which may include supplementary feeding, artificial nesting sites, predator and vegetation control; and 3) harvesting either for commercial or conservation purposes. SDM may be designed to provide an answer to the key question facing managers: what management action should be applied, considering the limited resources and uncertainties involved?

Problem definition and working groups. In the framework of SDM, the people who are interested in the outcome of the decision process are usually called *stakeholders* (Wilson & Arvai 2011). In reintroduction decisions in New Zealand, stakeholders are often species recovery groups that consist of Department of Conservation representatives, scientists, researchers, conservation managers and environmental activists.

The word "problem", in the frame of SDM, does not always mean something bad, but it applies to some complex situation that needs a decision. For example, one of the most common problems for reintroduction projects is to decide whether to release captive-reared or wild-caught individuals. Both may have their pros and cons that stakeholders need to evaluate. During this stage, the group of stakeholders discusses questions about who makes the decision, who should be involved, and when and how the decision will be made (Gregory et al. 2012).

Defining objectives. During this step, decision makers define what is important to them and how it could be measured (Gregory et al. 2012). Some problems in managing reintroduced populations will have few objectives, but most of them will have several. The important issue is to distinguish between means objectives and fundamental objectives (Wilson & Arvai 2011). For example, if "increase the number of populations of threatened animal X in country Z" is a fundamental objective, then "establish a self-sustaining population of threatened animal X in conservation area Y in country Z" would be a means objective. The establishment of a new population is important, because it would help to increase the overall number of populations in the country. However, means objectives could become fundamental if the problem statement is changed. For example, if one considers management of a particular reintroduced population in a particular territory, the objective "establish a self-sustaining population of threatened animal X in conservation area Y in country Z" becomes a fundamental for this particular project, and a means objective could be "increase the number of nesting sites for threatened animal X". In other words, means objectives tell us how we can achieve the fundamental objectives (Figure 4.2). Finally, fundamental objectives may be a part of another larger fundamental objective, like "increase biodiversity in country Z" (Gregory et al. 2012).

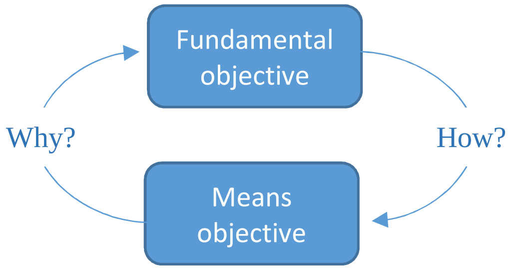

Figure 4.2 Difference between fundamental and means objectives. Adapted from Cochrane et al. (2011).

SDM requires not only consideration of a comprehensive set of objectives, but also developing the methods for measuring these objectives. Wilson and Arvai (2011) define three types of measurable attributes. First of all, there are *natural measures* that can be measured directly and expressed with real numbers, e.g. a project's cost or the number of individuals in the population over 10 years. If something cannot be measured directly, managers may use indirect or *proxy measures*, for example the number of successfully established reintroduced populations in a conservation area Y might be a measure of the quality of that area. If the results cannot be measured either directly or indirectly, the third type of measures is used – *constructed measures*. For example, in order to evaluate relationships between local residents and reintroduced animal X, managers may use surveys or polls.

Alternative management actions. This is the most creative step in SDM; it requires imagination and critical thinking. The best way to develop a good set of alternative management actions is to brainstorm the various ideas in a group of experts from different areas, and choose a set of appropriate alternative actions considering the constraints. In many cases, it means not only searching for good alternatives, but also creating new ones (Gregory et al. 2012). For example, in order to increase hatching success of reintroduced takahe *Porphyrio mantelli* on Mana Island, managers conducted a cross-fostering experiment with its closest relative, the pukeko *Porphyrio porphirio* (Bunin & Jamieson 1996). Alternatives for increasing the viability of reintroduced species may involve: continuing current management, providing supplementary food, conducting follow-up translocations, managing non-native predators, treating ectoparasites, and even removing the population from the current site. Sometimes, when the list of alternatives is large, it is useful to group alternatives into portfolios. For example, in order to increase the viability of hihi on Mokoia Island, one of the implemented alternatives was a combination of mite control and supplementary feeding (Armstrong et al. 2007). After identifying a range of alternatives, the list is then cut in order to account for constraints, because the feasibility of alternatives is limited by available resources such as funding, labour, source populations or knowledge.

Model and predict. During this step, managers predict the consequences of each alternative in a set according to chosen measurable attributes, using mathematical models based on the available data and expert opinion (subjective judgements). Mathematical models are very important as they provide information about future population dynamics that can be used as a basis for making decisions. Modelling a single-objective problem with no uncertainty is straightforward, but in reality post-release management involves many uncertainties that relate both to variability in the system and to lack of data. Thus modelling with uncertainties is essential during this step as it makes predictions more realistic.

It is also important to reduce uncertainty as much as possible. For example, managers use post-release monitoring to reduce uncertainty in vital rates estimates. Monitoring can be also designed to answer some specific questions like: what threatens the viability of the population? Are they limited in food? Are they vulnerable to predators? What individuals are more vulnerable e.g. males or females, fledglings or adults? If monitoring data on the target population are sparse, managers may incorporate into the model available information from other reintroduced populations (Parlato & Armstrong 2012; Gedir et al. 2013).

For clarity, collected information is presented in the form of consequence table, which combines objectives, alternatives and measurable attributes, and summarises information on how each objective performs under each management alternative. This information is given in terms of numbers and/or probabilities, and here I refer to these as *objective scores* ($S_{i,j}$).

Evaluating trade-offs. The perfect management action that does not have any drawbacks does not exist. Managers just need to weigh the benefits and costs of each management action and choose one where the cost/benefit balance is more acceptable for the particular situation. For example, the most common trade-off in reintroduction is evaluating the costs and benefits of follow-up translocation, especially when only one source population is available; thus the benefits of follow-up translocation are weighted against the costs of harvest. Evaluating the trade-off is complicated by uncertainty in the outcomes. Sometimes managers need to make a decision in a face of great uncertainty, because the problem needs immediate solving. For example, when only one pair of black robins *Petroica traversi* was left from the 7 birds reintroduced to Mangere Island, the conservation managers were forced to take a risky step. They conducted cross-fostering with Chatham tit *Petroica macrocephala chathamensis*, an action that probably saved black robins from extinction (Butler & Merton 1992).

Unfortunately, there is no a single recipe for identifying the best management action. However, several techniques help to deal with uncertainties, e.g. Simple Multi-Attribute Rating Technique (Box 4.2), stochastic dynamic programming, Bayesian belief networks, linked decisions, and simplifying the decision by reducing the number of alternatives and objectives (Gregory et al. 2012).

4.3 APPLYING SDM TO MANAGEMENT OF THE REINTRODUCED HIHI POPULATION IN BUSHY PARK

I used SDM to guide a decision about conservation management of the reintroduced population of hihi, an endangered New Zealand forest bird, in Bushy Park, a conservation reserve near Whanganui. In March 2013, 44 juvenile hihi were reintroduced to Bushy Park from Tiritiri Matangi Island. The population declined in the first year after reintroduction, with males surviving better than females. From the 21 reintroduced females, only four survived to the first breeding season and only two survived to the second (16 months). Monitoring over the first breeding season showed that the four females laid a total of at least 7 clutches, and fledged at least 16 chicks.

Problem definition. The main question was whether to conduct a follow-up translocation to reinforce the population, and if so, when.

The Bushy Park Trust proposed a follow-up translocation of an additional 15 females in August 2014. However, the proposed translocation was postponed until March 2015 by the Hihi Recovery Group (HRG) in order to collect more data on survival and reduce uncertainty before making a decision about whether to proceed with the translocation. The decision has been further complicated by detection of rats *Rattus sp.* inside the reserve in November 2014, meaning the follow-up translocation was contingent on successful eradication of the rats. The problem was to identify the best management action for maximize the viability of hihi in Bushy Park while also considering other objectives.

The New Zealand Department of Conservation (DOC) conservancies responsible for Bushy Park and Tiritiri Matangi are the decision makers, but their decision is expected to follow the recommendation of the HRG. The objectives and management alternatives for this analysis were elicited during the annual HRG meeting in April 2014 and were discussed later by e-mail. Evaluation of trade-offs and assigning scores of importance to the objectives was held by e-mail. The mailing list included eight HRG members: two from the DOC, four from community conservation groups (two from Bushy Park and two from Tiritiri Matangi Island) and two from universities.

Defining objectives. During the 2014 HRG meeting, four fundamental objectives were identified for the decision problem (Table 4.1). First of all, the HRG considered the impact of harvesting on the Tiritiri Matangi population, because this population plays a critical role in hihi conservation and is highly valued. Therefore, it was important to *Maximize the size of the source population.* As a measurable attribute for this objective, the projection of *Mean number of female hihi in September 2016* was chosen. The short-term projection was chosen because this population is frequently harvested for translocations, and therefore population size is hard to predict at longer time scales: since 2005 the number of birds removed annually to other conservation areas has varied between 0 and 72 (Parker 2013). Thus, the mean number of female hihi in September 2016 serves a measure of the impact of harvesting. Similar guidance has been frequently given for deciding the number harvested in previous years. However, this is normally expressed as how many juveniles can be harvested such that an adult female population is maintained of about 70 (Armstrong & Ewen 2013).

Table 4.1. Fundamental objectives and measurable attributes for hihi management in Bushy Park (BP).

Objectives	Measurable attributes	Preferred directions
Maximize the size of the source population	Mean number of female hihi in September 2016	Increase
Maximize the size of the BP population	Mean number of female hihi in 10 years	Increase
Minimize probability of extinction	Probability of extinction over 10 years	Decrease
Minimize costs to HRG	Amount of funds spent on the follow-up translocation project	Decrease

Secondly, the goal of this reintroduction project is to establish a population that will be viable. Thus the HRG also agreed on second objective – *Minimize probability of extinction of Bushy Park population* – and to use the extinction probability over 10 years as a measure of extinction risk.

Thirdly, it was also viewed as fundamentally important to have a large number of hihi at Bushy Park. This was treated as additional to its influence on extinction risk over 10 years, and was driven by a combined feeling of improved ecosystem services that hihi will provide to Bushy Park, an increase in human-hihi encounters, and greater long-term genetic viability. Therefore the HRG agreed on the third fundamental objective – *Maximize the size of the Bushy Park population* – with a measurable attribute *Mean number of female hihi after 10 years*. Objectives 2-3 were conflicting with objective 1, as increasing the Bushy Park population through translocation is expected to reduce the Tiritiri Matangi population.

The fourth objective was to *Minimize costs* to the HRG, as the translocation is partly funded by the HRG with funds that can be allocated elsewhere.

Alternative management actions. The set of alternatives was limited because, first of all, the intensity of management of the Bushy Park hihi population is unlikely to increase in the foreseeable future. Bushy Park is a community-based conservation project that has been managed by Bushy Park Homestead and Forest Trust since 1994, and is administered by a core of volunteers. Management of the reintroduced hihi population (as well as other conservation management at BP) relies on the collaboration of scientists and volunteers and is supported by donations and grants. Volunteers maintain five sugar-water feeders for hihi and monitor nest boxes during the breeding season. Secondly, the only suitable source for translocation is the Tiritiri Matangi population. This population has been used as a source for translocations since 2005, is intensively monitored, and is known to be able to sustain the harvests. The original remnant population on Little Barrier Island is also still occasionally used as a source population, but only to increase genetic diversity in populations believed to have a high probability of long-term persistence. In addition, Tiritiri Matangi is much more suitable for catching, holding and transporting birds than the less accessible and larger Little Barrier. Considering the objectives and above-mentioned constraints, the HRG developed two alternatives:

1. Continue current management (Status Quo)
2. Translocate 30 juvenile hihi (15 females and 15 males) to reinforce the Bushy Park population in March 2015 (follow-up translocation).

The unexpected rat invasion in Bushy Park in November 2014 increased the uncertainty in the decision-making process and necessitated the consideration of other alternatives. E-mail elicitation identified two additional alternatives: (3) delay the follow-up translocation to March 2016, and (4) remove hihi from Bushy Park. However, the last alternative was rejected during the discussion.

Model and predict. The Bushy Park population was monitored since reintroduction until the start of the second breeding season (September 2014), allowing estimation of adult and juvenile survival (see Chapter 2 for details) as well as reproduction. The Bushy Park and the Tiritiri Matangi populations were combined into one female-only model that incorporated uncertainty related to parameter estimations, model selection and demographic stochasticity. The model was coded in OpenBUGS (see Chapter 3 for details).

The modelling indicated great uncertainty about population outcomes under each management alternative (Table 4.2). Under the alternative “no translocation”, the Tiritiri Matangi population was predicted to increase to 68 females in September 2016, with 95%

credible interval (CRI) of 42–103. The mean number of female hihi in Bushy Park was predicted to decline to 22 (95% CRI = 0–100) after 10 years, with an extinction probability of 44%. This alternative, in contrast to both translocation alternatives, does not entail any additional financial costs. Both translocation alternatives are predicted to reduce the number of females on Tiritiri Matangi, to 55 (95% CRI = 31–88) under alternative "translocation 2015" and to 58 (95% CRI = 33–93) under alternative "translocation 2016". Both translocation alternatives are predicted to increase the number of females in BP in 10 years, with a mean of 26 (95% CRI = 0–100) females with 36% probability of extinction under both alternatives. Under all three alternatives, I assumed that rats were eradicated and did not affect hihi survival or reproduction.

In order to obtain objective weights (Box 4.1), I asked eight stakeholders, who participated in development of objectives and alternatives, to assign weights to each of the four fundamental objectives (Table 4.3). The information sent to stakeholders included the results from population modelling and brief information on how it was done. The two DOC representatives first approved the process of evaluation.

Box 4.1. **Objective weighting**. Many techniques help stakeholders to assign weights to objectives (see Hyde 2006 for summary). One of the most common is direct weighting, in which stakeholders express the importance of the objective with some number in a specified range. In assigning these weights, the stakeholders consider the estimated scores for the objectives under the different alternative actions. For example, for the first objective in this study, stakeholders considered the importance of decreasing the number of the Tiritiri Matangi females in 2016 from 68 to 55 due to harvesting.

To make this process easier for stakeholders, I first asked them to rank the objectives, with 1 given to the most important objective and then assign 100 to this objective. Other objectives were weighted relative to the most important one in a range between 0 and 100. Then, to obtain normalized weights that are required for SMART, I summed the weights and divided each value by the sum.

Evaluation of trade-offs. The SMART technique (Box 4.2) was used to evaluate the trade-offs between the alternatives. I added the lines of code for SMART to the population model (see Chapter 3), allowing me to incorporate model uncertainties in decision analysis. These new lines did the calculations shown in equations 3.1 and 3.2, giving a probability distribution for the final score under each alternative. In addition, I obtained distributions for the difference in the final scores between alternatives.

I first compared the final scores for the management alternatives using average stakeholder weights (Table 4.5; Figure 4.3). I then re-did the comparison using each of the eight sets of stakeholder weights to assess the sensitivity of the decision to these weights (Figure 4.4). Although objective scores were normalized, so they all are on the same scale between 1 and 0, the uncertainty allows values of the final score to be less than 0 or greater than 1.When average weights were used, the mean final score for alternative “no translocation” was slightly higher than that of the two translocation alternatives, but the distributions of these scores overlapped greatly (Table 4.5). All three distributions were multimodal with three peaks (Figure 4.3). This multimodality reflects the bimodal distribution of the number of females at Bushy Park after 10 years, with peaks at 0 and 100, and the approximately normal distribution for the number of females on Tiritiri Matangi.

The mean difference in the final scores between the alternatives “translocate 2015” and “no translocation” was slightly less than zero for each set of weights (Figure 4.4), meaning on average the translocation was not predicted to be beneficial. The differences in final scores between two translocation alternatives had its peak around 0 (Figure 4.4). This means that no alternative is clearly preferred. In addition, the distributions for these differences are wide and multimodal, reflecting the uncertainty in the population projections.

Box 4.2. **Simple Multi-Attribute Rating Technique (SMART).** The main idea of SMART (Barron & Barrett 1996) is to normalize objective scores so they are on the same scale (also known as standardisation), because it is difficult to compare, for example, number of birds and New Zealand Dollars. The most common method for normalization is to adjust objective scores based on their distance from the maximum and minimum value expected through all alternatives. This method transforms all objective scores to *normalized objective score*, so they all on the same scale, with 0 given to the worst outcome and 1 given to the best outcome:

$$N_{i,j} = \frac{S_{i,j} - \underset{i}{\min}\ (S_{i,j})}{\underset{i}{max}(S_{i,j}) - \underset{i}{min}(S_{i,j})}, \quad (3.1)$$

where $N_{i,j}$ is the *normalized objective score* under alternative i in relation to objective j, and $S_{i,j}$ refers to the original scores (Hyde 2006; Converse et al. 2013). For example, the normalized objective score for “Maximize size of the source population” (Table 4.2) under alternative “translocation 2016” is given by

$$N_{i,j} = \frac{58 - 55}{68 - 55} = 0.23.$$

SMART also requires assigning *weights* (Box 4.1), which are the measure of how stakeholders value fundamental objectives given the differences among alternatives.

Objective weights then are normalized, so they sum up to 1, to obtain *normalized weights* (W_j), and are then used to obtain the *final score* for each alternative (F_i), which is given by

$$F_i = \sum_j W_j * N_{i,j}. \quad (3.2)$$

The alternative with the highest final score is the preferred alternative. Sensitivity analysis allows understanding of how uncertainty in estimates or objective weights influences the decision.

Table 4.2 Consequence table summarising management options of hihi in Bushy Park (BP). The table illustrates the objective scores, i.e. performance of each alternative (translocation 2015, translocation 2016 and no translocation) on each objective. Objective one is measured by the mean number of female hihi on Tiritiri Matangi Island in September 2016, objective two is measured by the mean number of female hihi at BP after 10 years, objective three is measured as the probability of extinction over 10 years, and objective four is measured as the HRG contribution to the translocation in New Zealand Dollars. All alternatives assume that rats will be eradicated.

Objectives	Goal	Objective scores (mean with 95 % CRI)		
		Translocation 2015	Translocation 2016	No translocation
Maximize the size of the source population	Max	55.11(31-88)	58.37(33-93)	68.22(42-103)
Maximize the size of BP population	Max	26.91(0-100)	26.49(0-100)	22.22(0-100)
Minimize extinction of BP population	Min	36%	35%	44%
Minimize costs to HRG	Min	13,200	13,200	0

Table 4.3. Ranks and scores of fundamental objectives elicited from eight stakeholders, who were the Hihi Recovery Group members.

	Ranks							
Objectives	1	2	3	4	5	6	7	8
Maximize the size of the source population	1	3	1	2	1	1	1	3
Maximize the size of BP population	3	1	3	3	1	3	1	1
Minimize extinction risk of BP population	2	2	3	1	2	3	1	2
Cost	4	4	2	4	3	2	2	4
	Weights							
Maximize the size of the source population	100	50	100	90	100	100	100	60
Maximize the size of BP population	40	100	50	80	100	50	100	100
Minimize extinction risk of BP population	50	90	50	100	90	50	100	67
Cost	30	0	80	50	30	75	50	33

Table 4.4. Objective weights representing scores of importance, normalized so they sum to 1.

	Objective weights								Extreme weights		Averaged weights
Objectives	1	2	3	4	5	6	7	8	max	min	
Maximize the size of the source population	0.45	0.21	0.36	0.28	0.31	0.36	0.29	0.23	0.45	0.21	0.309
Maximize the size of BP population	0.18	0.42	0.18	0.25	0.31	0.18	0.29	0.38	0.42	0.18	0.274
Minimize extinction risk of BP population	0.23	0.38	0.18	0.31	0.28	0.18	0.29	0.26	0.38	0.18	0.264
Cost	0.14	0.00	0.29	0.16	0.09	0.27	0.14	0.13	0.29	0.00	0.154

Table 4.5. Final scores for each alternative obtained from the OpenBUGS model. Averaged objective weights were used.

Alternatives	Mean	SD	Median	95 % CRI	
No translocation	0.49	3.16	0.48	-3.03	6.67
Translocation 2015	0.55	3.23	0.32	-3.42	6.26
Translocation 2016	0.63	3.19	0.40	-3.37	6.35

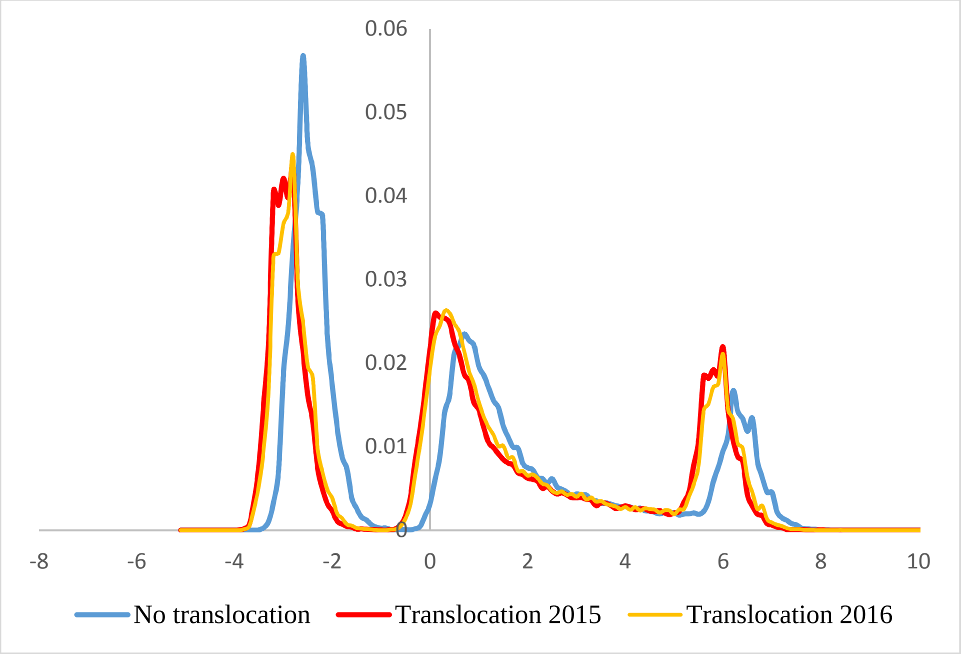

Figure 4.3. The probability distribution of the final score of three alternatives; averaged objective weights were used.

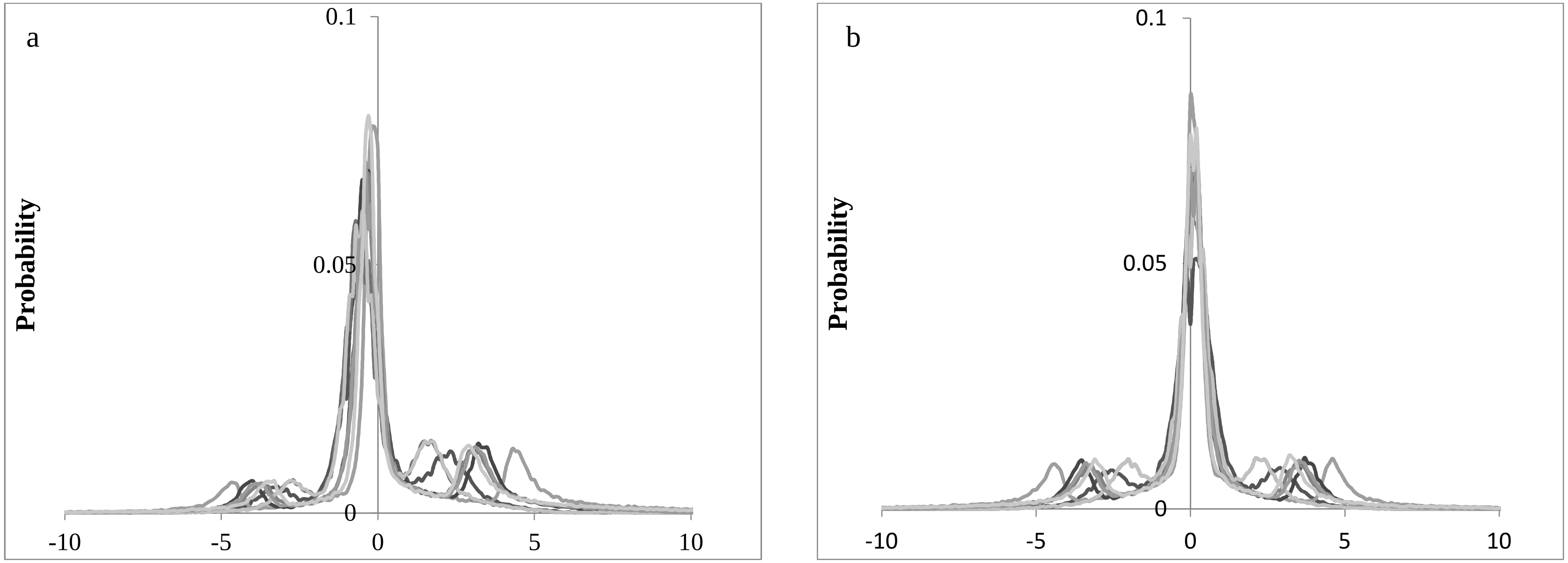

Figure 4.4. Eight probability distributions (corresponding to eight stakeholders) for the difference in final scores between alternatives: a) “Translocation 2015” and “No translocation”; and b) “Translocation 2016” and “Translocation 2015”.

4.4 DISCUSSION

The most obvious conclusion from this decision analysis is that none of the alternatives are clearly preferred. This mainly reflects the fact that the difference between the alternatives was not significant in terms of the Bushy Park objectives. However the question is “how decision makers will interpret this difference?”

This decision analysis combined four levels of uncertainty: the uncertainty in parameter estimations, uncertainty in model selection, demographic stochasticity, and the uncertainty in objective weights. As in many reintroduction projects, all projections in this analysis were uncertain, but they represent the level of certainty possible with the data available at this stage. The first three levels of uncertainty were described in detail in Chapters 2 and 3. The last level, uncertainty in objectives' weights, was the additional component for the current analysis.

There were several difficulties in interpreting the weights elicited from stakeholders. First of all, in this analysis, stakeholders directly assigned weights to the objectives. This means, they assigned the numbers from a predefined range from 0 to 100 to each of the objectives by comparing objective scores under each alternative and taking into account a range of uncertainties (Box 4.2). This method allows assigning all objective weights at once and can be held by e-mail in very short time. However, it has a big drawback – it does not guarantee that the assigned weights will match the requirements of the chosen decision analysis method. In other words, the assigned numbers may not reflect the actual trade-off between alternatives (Hobbs & Meier 1994). This problem is most acute when objective weighting is held by email, and the guidance on how to weigh objectives are not clear. In order to overcome misunderstanding between facilitator and stakeholder Pöyhönen and Hämäläinen (2001) recommend using more than one weighting technique, which helps “to check out possible inconsistencies and to increase the understanding of how the weights are interpreted.”

Secondly, I considered all sets of objective weights versus averaged weights. Other papers (e.g. Converse et al. 2013, Ewen et al. 2014) averaged objective weights between all stakeholders, but one may use average weights only when the assumption of a representative sample from a population is met. Although stakeholders assigned different scores of importance, the distributions of the final score of alternatives with different weights were very close (Figure 4.4).This indicates that stakeholders’ weights had minor role and the decision is not sensitive to objective weights, but instead is very sensitive to the uncertainties in the projections, which are large. By combining population models and SMART analysis in one OpenBUGS model, all levels of uncertainty were included. The OpenBUGS model made it possible to see how the simultaneous variation in parameter estimates influences the final decision. The distributions of final score obtained in this decision analysis were wide and multimodal, making the comparison of alternatives difficult to interpret. However, acknowledging this uncertainty is very important for decision makers, as it shows the variations in the possible outcomes and therefore the risks that accompany the decision.

Another important issue in assigning weights is the human factor. People, by their nature, use heuristics[1] while making decisions, and this can lead to a decision bias. For example, a *motivational heuristic* means that people place the most value on the objective in which they have a personal or business interest (Tani & Johnson 2013). Not surprisingly, in this study some stakeholders connected to Bushy Park valued the objective "maximize the size of Bushy Park population" higher than the objective "maximize the size of the source population" and vice versa. The other common heuristics in assigning weights are *recognition* and *representativeness*, whereby people put more value on objectives they are familiar with or that they understood better (Kahneman & Tversky 1974). The heuristic *take the last* also could take a place in this decision analysis. For example, if stakeholders valued the source population greater than the translocated population in the recent decision analysis, they might tend to do this again, even if the decision problem is different.

Finally, the information available to stakeholders about the Bushy Park and the Tiritiri Matangi populations probably also played an important role in objective evaluation. For example, Bushy Park representatives knew the fact that female fecundity was unusually high in Bushy Park (in comparison with other translocated populations). This information was not directly presented to the others, but was incorporated into projections. Although Bushy Park female fecundity estimates were high, the uncertainty around juvenile survival was large.

In SDM it is important to make sure that stakeholders understand the problem, objectives and consequences of the alternatives, possess the same information and, finally, have clear direction on how to assign their weights. For this study elicitation was held by email, but to overcome a common tendency and biases in decision making, it is better to work with stakeholders face-to-face, and conduct several rounds of eliciting scores of importance, to make sure all are on the same page. The outcome of this decision analysis will be reported

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[1] Mental shortcuts that involve focusing on one aspect of a complex problem and ignoring others.

at the next Annual Hihi Recovery Group Meeting and the second round of scores of importance elicitation will be conducted.

Conclusion. The type of decision described above, which is multi objective with uncertainty, is the most common in reintroduction and the most difficult to solve. Consequence tables, decision trees and other decision analysis techniques do not provide ready answers; they just provide stakeholders with information that can be used to guide the decision, which will depend on how they value the fundamental objectives and how they respond to uncertainties. SDM is intended to bring together knowledge of conservation managers, modellers and biologists to find the best solution to complex conservation problems. SDM allows different opinion and concerns to be expressed and discussed. Therefore, in decisions about post-release management it is important to use a logical structured approach. Although the outcome of this decision analysis remains uncertain, SDM brings data and decisions together by linking monitoring, modelling and decision analysis into a complex but single process, moreover it indicated what uncertainty affects the decision. The logical conclusion of this management process will be to link the last and the first step of SDM (Figure 4.1), and transform the linear process into a cycle by including adaptive learning in order to improve future management (Holling 1978, Walters 1986). For this reintroduction project, it would mean explicitly estimating the value of information from future monitoring, and then using the further monitoring data to resolve uncertainty about the viability of the hihi population at Bushy Park. Management can then be updated in response to a new understanding about this population.

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Chapter 5. General discussion

During the last three decades reintroduction biology has consisted of various types' scientific study with the general aim of improving reintroduction success (Seddon et al. 2007). One of the tools that help to improve the success of reintroduction projects is decision analysis, or Structured Decision Making (SDM). The effectiveness of the structured decision approach in reintroduction decisions was first applied to management of endangered whooping crane *Grus americana* in North America (Converse et al 2013). SDM is a very useful tool that allows managers to cooperate with key stakeholders in order to identify the optimal management action considering project objectives, constraints and available alternatives (Nichols & Armstrong 2012). It may combine together the components of monitoring, modelling and decision analysis, which have tended to be treated separately in reintroduction projects. This Masters project aimed to show how conservation managers can use SDM to guide management decision after initial reintroduction, when data are still limited and there are a lot of uncertainties around vital rates estimates. This research work showed that SDM is not a panacea that solves all the decision difficulties managers are faced to. However, if used wisely, it "provides a transparent and logical" basis for making reasonable decision (Nichols & Armstrong 2012). So what does it mean, "to use wisely"?

One of the most important issues in SDM is consistency, which means first defining the problem and fundamental objectives, then developing management alternatives and predicting consequences, and finally implementing the appropriate management actions. In this research, I illustrated these steps using the example of endangered hihi in Bushy Park. According to objectives and alternatives that were developed in cooperation with Hihi Recovery Group, I predicted the outcome of each considered management action and evaluated the trade-offs between them. In Chapter 2, I used post-release monitoring data to estimate hihi survival and fecundity. A number of models were built to predict survival of juveniles and adults, as well as distinguishing between age and post-release effects and assessing the difference between sexes. Then, in Chapter 3, I used survival and fecundity estimates to model population growth under three management alternatives. Finally, Hihi Recovery Group members assigned the values of importance to each of the four fundamental objectives, allowing the decision analysis in Chapter 4 that accounted for these objective weights.

"To use wisely" also means acknowledging the uncertainty that is an integral part of decision analysis. I gave a lot of attention to the different uncertainties in each of three classes of models that were built during this Masters project, which are *estimation*, *population modelling* and *decision analysis* (Converse et al 2013). Armstrong & Reynolds

(2012) recommend including both type of uncertainty, stochasticity and incertitude (imperfect knowledge), in models of reintroduced population.

Model selection uncertainty, which belongs to the group of incertitude, should be included in the parameter estimation based on the population models. Model selection uncertainty is essential when management decision is based on the projections, because different models may lead to different management decisions (Bell et al. 2013). Including model selection uncertainty can be done, for example, by model averaging (Burnham & Anderson 2002). I included model selection uncertainty in survival estimation model by applying model averaging in program MARK (Cooch & White 2006). Model averaging increased confidence intervals in parameter estimates. For example, the most parsimonious survival model suggested that males and females adult monthly survival rate was 0.96 with a 95% confidence interval ranging from 0.91 to 0.98, but after model averaging female monthly survival rate were estimated to be lower, 0.94, with the 95% ranging from 0.81 to 0.98 (Chapter 2). Model selection uncertainty can be also included in a population model, which means averaging the projections from several population models. However, this is not currently straightforward, as the approach requires integrated Bayesian population modelling (Abadi et al. 2010), and Bayesian model averaging is quite complex (Barker & Link 2014). Bayesian updating software such as OpenBUGS does not offer a model averaging option yet.

*Parameter estimation uncertainty***,** which is the second type of incertitude, should be accounted for in population models and in decisions. The easiest way to include parameter estimation uncertainty is by sampling random variables based on the mean and standard error of the parameter. Even standard spreadsheet applications such as Excel have functions that allow such sampling. I used OpenBUGS program to code population models (McCarthy 2007), allowing me to use a range of different distributions and to account for estimation uncertainties in survival, fecundity and objective weights (Chapter 3 and 4).

Demographic stochasticity, which is a type of stochasticity uncertainty that is essential for small populations, should be included in a population models for reintroduced populations, at least in their early stages. In contrast to other forms of uncertainty, it has been standard to include demographic stochasticity in population viability analysis for more than 20 years. I modelled demographic stochasticity in survival by sampling from binomial distribution and in fecundity by sampling from a Poisson distribution. The good thing about demographic stochasticity is that no data are required to model it (Armstrong & Reynolds 2012).

Projections of population growth are always uncertain, and the degree of uncertainty depends on how well the target species is studied and what data are available on vital rates, such as age and sex-specific survival, age-specific annual reproductive success, age and sex-specific dispersal and sex ratio. In the case of modelling the growth of reintroduced population, managers often use the vital rates of the source population or the vital rates of another available population (Holland et al. 2009), which increases uncertainty even more. However, this uncertainty, when incorporated into the model, along with inherent system variability, makes projections defensible and robust. Moreover, when all vital rates are modelled together the covariance among all parameters is accounted for, and projections become more reliable (Holland et al. 2009).

My decision analysis did not provide a straightforward answer, as the decision was sensitive to the high level of projection uncertainty. However, with this quantitative information, stakeholders are fully aware of the risks they are undertaking when choosing one or another management action.

My research made several valuable contributions in the field of modelling reintroduced population and evaluating the trade-off between management alternatives:

- **Understanding of post-release management decisions was improved by combining the source and the release populations in a single population model.** Dimond & Armstrong (2007) evaluated the costs of harvesting a source population of New Zealand robins, and this idea was developed further in my Master project. The population model built for this thesis integrated the source population and the released population models and made the modelling of different follow-up translocation scenarios precise, allowing understanding of how these two populations interact together in response to translocation. The model was coded in OpenBUGS, and incorporates the two types of uncertainty described above as well as allowing for parameter interactions. This demographic model is flexible enough that it can be easily updated when new monitoring data are collected. Moreover, it can be applied not only to other hihi translocations, but also to translocations of other threatened species, for which data are usually sparse and uncertainty is high.
- **Post-release data were improved by incorporating prior information to the model.** In Chapter 3, I showed how incomplete data can be supplemented by referring to some available information. Gedir at al. (2013), using the example of saddleback

population model for Bushy Park, showed that prior information from other translocated populations can reduce model uncertainty in projections, even when only one year of monitoring data are available. As I did not have data to estimate the fecundity of second-year females in Bushy Park, I used fecundity estimates of Tiritiri Matangi females as the upper bound for Bushy Park female fecundity.

- **Objective weights were incorporated in the population model with the tool for decision analysis.** In Chapter 4, the population model was combined with the model for decision analysis. Elicitation and interpretation of the objective weights were the most difficult and the most important part in this decision analysis. Stakeholders needed to decide how valuable both the released and the source population of hihi are, and therefore whether the estimated benefits of the follow-up translocation to the Bushy Park hihi population is worth the cost in terms of the impact on the Tiritiri Matangi population and the funds required. This is a common trade-off in reintroduction projects, and it is natural to value one population of the same species over another, for example, because genetic diversity of one population may be greater (Soulé 1985). However the level of risk to both populations also must be taken into account. On the one hand, projections indicated the follow-up translocation slightly reduced the risk of extinction of the Bushy Park population over 10 years, but on the other, hand the risk of extinction of the source population after harvesting was close to zero.

Finally, I would like to point out several issues that were raised by this Master project and lead to recommendations for further research on the management of the populations focused on:

- In order to aid future decision about management of the Bushy Park population it is essential to reduce uncertainty in vital rates. Additional monitoring data on the Bushy Park population should be collected in order to reduce uncertainty around juvenile survival and fecundity estimates. Updated vital rates can be easily incorporated into the developed population model and new projections should be used to adapt management.
- To obtain a more realistic carrying capacity for Bushy Park and for Tiritiri Matangi, the method for subjective judgment elicitation should be reviewed and elicitation

should be conducted between broader ranges of experts, as Delphi method has several drawbacks when sample size is small (P. Frost personal comments, January 5, 2015).

- Monitoring should be used to investigate whether population density has an effect on per capita growth in hihi populations. Studying the density dependence mechanisms in the Tiritiri Matangi population, as well as the Bushy park population in the future (if this population persists) will contribute to future management of these populations.
- Instead of computing survival rates separately in program MARK, it could be incorporated into the population model in OpenBUGS, and multimodal Bayesian inference applied once the methods for doing so become more accessible (Barker & Link 2014).
- The invasion of rats to Bushy Park suggests that the risk of predation should be included in the population model even the released site is fenced. It could be done, for example, by incorporating periodic invasions as “catastrophes”.

Conclusion. In the framework of Structured Decision Making, post-release management decisions are the result of group thinking and cooperation work of people from different areas, like scientists, conservation managers, modelers, executives and local community. Based on the available data and best knowledge, this group of interested people shares the risk of decision making together. Even when the outcome of reintroduction is uncertain, SDM is a process that allows people to understand this risk and make conscious decisions.

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Appendix A. Summary of hihi translocations

Table A.1. Summary of hihi translocation during the period 1980-2014. Locations in bold show present locations of hihi. Updated from the table in Department of Conservation 2005.

Location	Area (ha)	No. of Transfer	Total birds transferred	Year of transfer	Source for transfer	Notes	References
Hen Island	500	2	46	1980, 1981	Hauturu	Did not establish.	DOC 2005
Cuvier Island	195	2	65	1982, 1984	Hauturu	Did not establish	DOC 2005
Kapiti Island	1966	6	211	1983, 1985, 1990, 1991, 1992, 2002, 2010	Hauturu, Tiritiri Matangi, Mokoia, Mt Bruce	Population persists with constant management	DOC 2005, Parker 2013, Boyd & Castro 2000
Mokoia Island	135	1	40	1994	Hauturu	Population transferred to Kapiti	Armstrong et al. 2007
Tiritiri Matangi	220	2	57	1995, 2010	Hauturu	Population growth with constant management	DOC 2005
Mt Bruce	N/A	4	16	1995, 1999, 2001, 2002	N/A	Fluctuated between 3 and 12 birds	DOC 2005
Zealandia	225	3	70	2005, 2012	Tiritiri Matangi, Mt Bruce	Population persists with constant management.	Parker 2013
Ark in the Park	1100	3	110	2007, 2008	Tiritiri Matangi	Did not establish	Warneford 2009
Maungatautari	3,255	3	155	2009, 2010, 2011	Hauturu and Tiritiri	Needs further monitoring	Ewen et al. 2011
Bushy Park	87	1	44	2013	Tiritiri Matangi	Needs further monitoring	Parker 2013

Appendix B. OpenBUGS code for combined population model

#Following model combined models for the Tiritiri Matangi and the Bushy Park (BP) hihi populations and decision analysis. Three management alternatives are modelled simultaneously. Average objective weights are used in “Data” section.

```
Model {

######################################
# 1.Tiri population model
######################################

#################
#1.1. Fecundity
#################

# Priors for fecundity
log.mu1.Tiri ~ dnorm(log.mu1.mean, log.mu1.prec)        #  log mean fledged per 1-year female
log.age.Tiri ~ dnorm(log.age.mean, log.age.prec)        # difference in log mean for older vs 1-year
female
log.mu1.prec <- pow(log.mu1.sd,-2)
log.age.prec <- pow(log.age.sd,-2)
log(mu1.Tiri) <- log.mu1.Tiri
log(mu2.Tiri) <- log.mu1.Tiri+log.age.Tiri

################
#1.2. Survival
################

# Priors for survival
logit.surv.a ~ dnorm(logit.surv.a.mean, logit.surv.a.prec)# logit monthly survival for adults
logit.surv.j ~ dnorm(logit.surv.j.mean, logit.surv.j.prec)    # logit monthly survival for juveniles
logit.surv.a.prec <- pow(logit.surv.a.sd,-2)
logit.surv.j.prec <- pow(logit.surv.j.sd,-2)

# Calculations
surv.a.Tiri <-pow(exp(logit.surv.a)/(1+exp(logit.surv.a)),12)  # probability adult survives 12 months
surv.j.sep <-pow(exp(logit.surv.j)/(1+exp(logit.surv.j)),7)  # probability juvenile survives to September
surv.j.mar <- pow(surv.j.sep,exponent+(1-exponent)/7)  # probability juvenile survives to March
surv.j.marsep <- pow(surv.j.sep,(1-exponent)*6/7) # probability juvenile survives March-September

####################
#1.3. Population model
####################

# Look up initial numbers in each age class
logit.p ~ dnorm(logit.p.mean, logit.p.prec)  # logit probability of detection
logit.p.prec <- pow(logit.p.sd,-2)
logit(p) <- logit.p      # probability of detection
U1 ~ dnegbin(0.1,1) # prior for number undetected 1st-year females
U2 ~ dnegbin(0.1,1) # prior for number undetected older females
```

```
# for three different alternatives: 1 – No  harvest, 2 – harvest in 2015, 3 – harvest in 2016
for (a in 1:3) {
    F1.Tiri[a,1] <- nf1+U1         # number 1st-year females in Sep 2014
    F2.Tiri[a,1] <- nf2+U2         # number older females in Sep 2014
    nf1 ~ dbin(p,F1.Tiri[a,1])      # account for imperfect detection
    nf2 ~ dbin(p,F2.Tiri[a,1])
    F.Tiri[a,1] <- F1.Tiri[a,1]+F2.Tiri[a,1]          # total no. of females in September 2014

# run simulations until September 2024
for (i in 2:11) {
    JF.mu[a,i-1] <- 0.5*((mu1.Tiri*F1.Tiri[a,i-1])+(mu2.Tiri*F2.Tiri[a,i-1])) # expected no. female 
fledglings
    JF[a,i-1] ~ dpois(JF.mu[a,i-1])          # sample actual number of female fledglings
    J.March.before[a,i-1] ~ dbin(surv.j.mar, JF[a,i-1])  #sample no. juv fem in March before harvest
    J.March.after[a,i-1] <- J.March.before[a,i-1] - JF.trans*trans[a,i-1] #no. juv fem after harvest
    F1.Tiri[a,i] ~ dbin(surv.j.marsep, J.March.after[a,i-1])          #sample no. female juv next Sep
    F2.Tiri[a,i] ~ dbin(surv.a.Tiri, F.Tiri[a,i-1])         # sample number older females alive next Sep
    F.Tiri[a,i]<- F1.Tiri[a,i]+F2.Tiri[a,i]
        }
    }

##########################################
#2.Bushy Park population model
##########################################

##################
#2.1. Fecundity
##################
log.BP ~ dnorm(0,0.01) I(,0)                # effect of BP (i.e. BP - Tiri) on log fledglings per female
log(mu1.BP) <- log.mu1.Tiri+log.BP                    #mean no. fledglings per 1-year female at BP
log(mu2.BP) <- log.mu1.Tiri+log.BP+log.age.Tiri    #mean no. fledglings per older female at BP

# Model data for 4 females at BP (all 1st-year)
for (i in 1:n.fem) {
    fl[i] ~ dpois(mu1.BP)
    }

#################
#2.2. Survival
#################

Trans1 ~dnorm(1.33, 4.69)  # logit survival for 0-1 month post-release
Trans5 ~ dnorm(1.45, 7.06)   #  logit monthly survival for 1-6 month post-release
phi.a.mo ~ dnorm(2.70, 2.47)  # logit monthly survival for adults
phi.f.mo ~ dnorm (1.06, 2.74)  #logit monthly survival for fledglings (January-March)
phi.j.mo ~ dnorm(2.67, 1.56)   #logit monthly survival for juveniles (March-September)

# Calculations
phi.trans1 <- exp(Trans1)/(1+exp(Trans1))        # probability translocated bird survives first month
phi.trans6 <- phi.trans1*pow(exp(Trans5)/(1+exp(Trans5)),5)     # probability translocated bird 
survives to September
phi.a <-pow(exp(phi.a.mo)/(1+exp(phi.a.mo)),12)          # annual adult survival
phi.f <- pow(exp(phi.f.mo)/(1+exp(phi.f.mo)),2)
phi.j <- pow(exp(phi.j.mo)/(1+exp(phi.j.mo)),6)
phi.fj <- phi.f*phi.j         # probability juvenile survive March -September

#####################
#2.3. Population model
#####################
```

```
# for three different alternatives: 1 – No  translocation, 2 – translocation  in 2015, 3 – translocation in
2016
for (a in 1:3) {
# Look up initial numbers in each age class
      F.BP[a,1] <- F1.BP[a,1]+F2.BP[a,1]              # total no. of females in September 2014

# run simulations until September 2024
for (i in 2:11)     {
      F2.BP[a,i] ~ dbin(phi.a,F.BP[a,i-1])
      mutot[a,i-1] <- 0.5*(mu1.BP*F1.BP[a,i-1]+mu2.BP*F2.BP[a,i-1])       # expected no. female
fledglings
      FF.BP[a,i-1] ~ dpois(mutot[a,i-1])      # sample actual number of female fledglings
      F.trans[a,i] ~ dbin(phi.trans6, JF.trans) #sample no. translocated female juveniles in September
      F1.max.BP[a,i] ~ dbin(phi.fj,FF.BP[a,i-1]) # sample max number female recruits next year
      F1.BP[a,i] <- min(K-F2.BP[a,i], F1.max.BP[a,i]+F.trans[a,i]*trans[a,i-1])# no. 1st-year fem in Sep
      F.BP[a,i] <- F1.BP[a,i]+F2.BP[a,i]    # total number of females next year
      PE.BP[a,i] <- step(-F.BP[a,i]) # probability that population extinct next year
            }
      }

########################################
#3. Structured Decision Making
########################################
#for three different alternatives
for (a in 1:3) {
# Outcomes from population models for BP and Tiri
      obj[a,1] <- F.Tiri[a,3]           # number females on Tiri Sep 2016
      obj[a,2] <- F.BP[a,11]           # number females at BP Sep 2024
      obj[a,3] <- step(obj[a,2]-1) # 1 if BP pop persists, 0 otherwise

# Normalise and multiply by weights
for (i in 1:4) {
      obj.norm[a,i] <- (obj[a,i]-min[i])/(max[i]-min[i])  #normalise objective scores by equation (3.1)
      score[a,i] <- weight[i]*obj.norm[a,i]   # multiply by weights by equation (3.2)
            }
      sum.score[a] <- sum(score[a,])        # add together objective scores by equation (3.2)
      }

      benefit.trans.2015 <- sum.score[2]-sum.score[1]      # whether 2015 trans better than no trans
      benefit.2016 <-  sum.score[3]-sum.score[2]            # whether 2016 trans better than 2015 trans

}

#Data
list(
#Tiri
#survival
logit.surv.a.mean=3.4428, logit.surv.a.sd=0.1361,logit.surv.j.mean=2.05, logit.surv.j.sd=0.062,
exponent=0.425,
#fecundity
log.mu1.mean=1.025, log.mu1.sd=0.1064, log.age.mean=0.3786, log.age.sd=0.1064,
logit.p.mean=2.11, logit.p.sd=0.58,
#number of female in each age class
nf1=13, nf2=15,
#BP
n.fem=4, #number of females
fl=c(6,6,3,0), #fledglings per female
K=100, #carrying capacity
#number of female in each age class
```

```
F1.BP=structure(.Data=c(
4,NA,NA,NA,NA,NA,NA,NA,NA,NA,NA,
4,NA,NA,NA,NA,NA,NA,NA,NA,NA,NA,
4,NA,NA,NA,NA,NA,NA,NA,NA,NA,NA
),.Dim = c(3, 11)),
F2.BP=structure(.Data=c(
2,NA,NA,NA,NA,NA,NA,NA,NA,NA,NA,
2,NA,NA,NA,NA,NA,NA,NA,NA,NA,NA,
2,NA,NA,NA,NA,NA,NA,NA,NA,NA,NA
),.Dim = c(3, 11)),

#Translocation
JF.trans=15,     # assume 15 females translocated if translocation done
trans=structure(.Data=c(
0,0,0,0,0,0,0,0,0,0,        # no translocation
1,0,0,0,0,0,0,0,0,0,        # translocation in 2015
0,1,0,0,0,0,0,0,0,0         # translocation in 2016
),.Dim = c(3, 10)),

#Structured Decision Making
weight=c(0.309,0.274,0.264,0.154), #average weights
# min and max objective scores from the consequences table
min=c(55.11,22.22,0.56,-13200),
max=c(68.22,26.91,0.65,0),
# cost matrix
obj=structure(.Data=c(
NA,NA,NA,0,                 # no translocation
NA,NA,NA,-13200,            # translocation in 2015
NA,NA,NA,-13200             # translocation in 2016
),.Dim = c(3, 4)))

#Inits
list(
#Tiri
U1=5,U2=5,log.age.Tiri=0.35,
J.March.before=structure(.Data=c(
35,50,50,50,50,50,50,50,50,50,
35,50,50,50,50,50,50,50,50,50,
35,50,50,50,50,50,50,50,50,50
),.Dim = c(3, 10)),

#BP
log.BP=-1, F1.max.BP=structure(.Data=c(
NA,3,3,3,3,3,3,3,3,3,3,
NA,3,3,3,3,3,3,3,3,3,3,
NA,3,3,3,3,3,3,3,3,3,3
),.Dim = c(3, 11)),
F2.BP=structure(.Data=c(
NA,3,3,3,3,3,3,3,3,3,3,
NA,3,3,3,3,3,3,3,3,3,3,
NA,3,3,3,3,3,3,3,3,3,3
),.Dim = c(3, 11)),
FF.BP=structure(.Data=c(
3,3,3,3,3,3,3,3,3,3,
3,3,3,3,3,3,3,3,3,3,
3,3,3,3,3,3,3,3,3,3
),.Dim = c(3, 10))
)
```