Conservation translocations and monitoring of kiwi

A thesis
submitted in partial fulfilment
of the requirements for the Degree of
Doctor of Philosophy

at
Lincoln University
by
Peter Jahn

Lincoln University

2021

Publications and manuscripts

Kiwi translocation review: are we releasing enough birds and to the right places?

Peter Jahn [1], E Fernando Cagua [2], Laura E Molles [1,3], James G Ross [1], Jennifer M Germano [4]

1. Department of Pest Management and Conservation, Lincoln University, PO Box 85084, Lincoln 7647, New Zealand
2. Centre for Integrative Ecology, University of Canterbury, PB 4800, Christchurch 8140, New Zealand
3. Verum Group, PO Box 29-415, Christchurch 8024, New Zealand
4. Department of Conservation, Whakatū / Nelson Office, PB 5, Nelson 7042, New Zealand

PJ, LM, JR, and JG all contributed to the conceptualising of the study; PJ and FC designed the methodology and undertook the data analysis. PJ wrote the manuscript and all other authors reviewed and contributed text. Data curation and visualisation were undertaken by PJ.

New Zealand Journal of Ecology 46(1): 3454. Manuscript accepted on 7 June 2021 and published online on 13 September 2021.

Post-translocation movements and ranging behaviour of roroa–great spotted kiwi

Peter Jahn [1], James G Ross [1], Vanessa Mander [1], Laura E Molles [1,2]

1. Department of Pest Management and Conservation, Lincoln University, PO Box 85084, Lincoln 7647, New Zealand
2. Verum Group, PO Box 29-415, Christchurch 8024, New Zealand

PJ, JR, and LM all contributed to the conceptualising of the study; PJ and VM collected and curated the data. PJ designed the methodology and undertook data analysis and visualisation. LM acquired funding. PJ wrote the manuscript and all other authors reviewed and contributed text.

Manuscript re-submitted on 15 February 2022 for publication in *Notornis*.

Acoustic monitoring and occupancy analysis: cost-effective tools in reintroduction programmes for roroa–great spotted kiwi

Peter Jahn [1], James G Ross [1], Darryl I MacKenzie[2], Laura E Molles [1,3]

1. Department of Pest Management and Conservation, Lincoln University, PO Box 85084, Lincoln 7647, New Zealand
2. Proteus Wildlife Research Consultants, PO Box 7, Outram 9062, New Zealand
3. Verum Group, PO Box 29-415, Christchurch 8024, New Zealand

PJ, JR, and LM all contributed to the conceptualising of the study; PJ, DM, and LM designed the methodology. PJ and LM curated the data. LM acquired funding and processed the data in Kaleidoscope. PJ undertook data collection, data analysis, and visualisation. PJ wrote the manuscript and all other authors reviewed and contributed text.

New Zealand Journal of Ecology 46(1): 3466. Manuscript accepted on 12 October 2021 and published online on 24 December 2021.

Thesis abstract

Conservation translocations
and monitoring of kiwi

by

Peter Jahn

Translocations of kiwi (*Apteryx* spp.) are one of the most common and growing types of conservation translocations in New Zealand. However, their outcomes remain mostly unpublished, which does not allow for sharing of lessons learnt from past developments. We reviewed 102 kiwi translocations from the 19th century until 2018 and identified factors affecting their outcome. North Island brown kiwi (*A. mantelli*) was the most translocated species, but the highest impact of translocations on the improvement of conservation status was for the rarest taxa: little spotted kiwi (*A. owenii*), rowi (*A. rowi*), and Haast tokoeka (*A. australis* 'Haast'). Translocations are typically used for creating secure populations and, more recently, for ecosystem restoration and meta-population management. We developed a set of criteria to evaluate the outcome of introductions and reintroductions based on demographic parameters alongside current recommendations on the genetic make-up of translocated populations. Based on these criteria, only a few translocated populations can be considered successful in the medium–long term: 15+ years following the release of a genetically diverse population (40+ unrelated individuals). Most historical translocations failed or require further genetic and habitat management. However, the majority of kiwi translocations have occurred over the last two decades and, while several populations have successfully established, for most of them, it is too soon to assess their medium-long term outcome. An analysis of factors affecting translocation outcomes revealed that, despite ongoing predator control, populations at small, unfenced sites on the mainland suffer from dispersal and predation, which has negative demographic and genetic consequences. Releases to larger mainland sites and predator-free areas have increased survival times, indicating higher chances for a positive translocation outcome. Moreover, translocated wild-caught and captive-sourced birds survived longer than birds from the Operation Nest Egg (ONE) programme, particularly at sites that were not predator-free. We highlight the need for genetic considerations in the planning and adaptive management of proposed and existing translocated populations. Specifically, we suggest that differences in kiwi survival, based on the type of released birds and release site's area size and predator status, should be considered during translocation planning.

The kiwi translocation review identified significant inconsistencies and often insufficiency of post-translocation monitoring. We demonstrate the utility of post-translocation monitoring methods in a recent translocation case study: a reintroduction of roroa–great spotted kiwi (*A. maxima*) in the Nina Valley, Lake Summer Forest Park. In 2015, eight wild-caught adults were translocated from the Hawdon Valley, Arthur's Pass National Park, following the release of ten ONE subadults between 2011–13. We tracked the translocated kiwi by radio telemetry between 2015–17 to monitor post-release survival, dispersal, and ranging behaviour. Dispersal was highly variable among the released wild birds. The straight-line distance from the release site to the last recorded location ranged between 0.5–10.3 km. Based on the dynamic Brownian bridge movement model, seven of the wild birds survived, remained in the Nina Valley, and covered up to 1700 ha (95% utilisation distribution). Releasing the wild birds had no measurable impact on the ranging behaviour of previously released subadults.

Additionally, we used occupancy modelling to analyse passive acoustic monitoring data (PAM) from the Nina and Hawdon valleys to monitor changes in distribution and growth of the translocated population and the impacts of the translocation for the source population. We analysed data from two survey years 2012–13 and 2017–18, being before-and-after the 2015 translocation. Occupancy estimates increased significantly at both study areas, despite the translocation of approximately 20% of known territorial adults (four pairs) from the Hawdon to the Nina. Moreover, at least three out of four territories, where adult birds were removed, were re-occupied by new pairs within 2.5 years. Site occupancy increased in the Nina from 0.20 (SE 0.10) to 0.72 (0.10), and in the Hawdon from 0.63 (0.10) to 0.95 (0.04). Detectability varied significantly between study areas and was influenced by the length of survey night, breeding/non-breeding season, and wind speed. The differences between the naïve and estimated occupancy values underscore the benefits of occupancy modelling for measuring response to conservation management. This study demonstrates the utility of PAM in monitoring translocation outcomes: tracking changes in occupancy and local distribution and assessing impacts on the source population following the birds' removal for translocation.

Keywords: conservation translocation, reintroduction, reinforcement, translocation outcome, translocation success, post-release effects, post-release survival, home range, dispersal, radio telemetry, passive acoustic monitoring, bioacoustics, occupancy analysis, occupancy modelling, *Apteryx*, *Apteryx maxima*, *Apteryx haastii*

Acknowledgements

I want to thank my supervisors, Laura, James, and Adrian for their support and encouragement, both professional and personal. I am grateful for their guidance, advice, and expertise. I am also very thankful to my external co-supervisor Jen and co-authors, Fernando, Vanessa, and Darryl. This thesis was hugely influenced by their ideas and contributions. Thanks to all people who accompanied me in the field and provided data.

I also want to thank Lincoln University, Kiwis for Kiwi, and the Ornithological Society of New Zealand. Without their scholarships and grants, this research would not be possible. Additionally, my gratitude goes to the Department of Conservation *Te Papa Atawhai* for their support both during the field work and putting together the kiwi translocation review.

Most importantly, I want to thank to my family and friends, who supported me during these years. V první řadě děkuji mé mamince Marii a celé rodině, že při mně stáli a podporovali mě. Gracias a Fer para siempre estar a mi lado, para su ayuda y todo. Thanks to everyone who was part of this stretch of my life journey.

Table of Contents

List of Tables

List of Figures

Chapter 1
General introduction

1.1 Conservation translocations

Conservation translocations are one of the major tools in species recovery management worldwide with a huge increase in popularity in the last few decades (Bajomi et al. 2010). Animals and plants are deliberately moved by humans from one area to another in order to achieve a conservation benefit for a population, species, or the entire ecosystem (IUCN/SSC 2013). A whole spectrum of conservation translocations exists, ranging from population restorations (reintroduction or reinforcement/supplementation of extant populations) to conservation introductions of species outside their historical range (assisted colonisation or ecological replacement of extinct taxa) (Seddon 2010; Seddon et al. 2012). However, translocations are also carried out for other reasons outside this spectrum, such as biological control, rehabilitation releases (IUCN/SSC 2013), conservation advocacy (Parker 2008), or a rapidly growing field of salvage/mitigation driven translocations (Germano et al. 2015; Nally & Adams 2015).

Reintroductions, one of the main and growing types of conservation translocations, aim at returning species into their historical ranges, from which they were previously extirpated (Armstrong et al. 2019). Causes of local extinctions vary widely for different species, ranging from habitat loss, disease, human persecution, and predation by introduced pests. For a successful reintroduction, these original causes of decline need to be identified and removed or sufficiently eliminated before the release of animals can take place (IUCN/SSC 2013). However, the evaluation of translocation success is more complicated as there are no widely accepted and applied criteria on what constitutes a successful translocation project (Fischer & Lindenmayer 2000; Chauvenet et al. 2013; Robert et al. 2015). Therefore, the success of a translocation largely depends on meeting the objectives that are specific to each project (Armstrong et al. 2019).

Post-translocation monitoring is a crucial tool to evaluate the reintroduction outcome during three different post-release phases: population establishment, growth, and self-regulation of a population at carrying capacity (Sarrazin 2007).

Robust data analysis from case studies based on post-translocation monitoring allows refinement of translocation protocols, guidelines for translocation planning, and increase the likelihood of positive translocation outcomes (Armstrong & Seddon 2008; Nichols & Armstrong 2012; Parker et al. 2013). There are numerous calls in the literature to improve standards of the monitoring and reporting of translocation outcomes (Sutherland et al. 2010; Miller et al. 2014; Gitzen et al. 2016). Appropriate monitoring and reporting from other projects provide necessary information and guidance for conservation practitioners, who often rely on anecdotes rather than robust scientific evidence (Sutherland et al. 2004; Adams & Sandbrook 2013; Sutherland & Wordley 2017).

1.2 Translocations in New Zealand

New Zealand's unique biodiversity has been seriously hit by a suite of introduced invasive predators since humans' arrival and particularly European settlers in the 19th century. Many indigenous species have been driven to extinction due to predation, competition, over-harvesting, and habitat loss (Wilson 2004). These pressures coupled with climate cause gradual decline of many other native species and ecosystems, ultimately leading to biodiversity crisis (Department of Conservation 2020a). The wave of local extinctions and the spread of exotic mammalian predators have been addressed by conservation efforts, particularly translocations, which appeared as early as the end of the 19th century (Nally & Adams 2015; Armstrong et al. 2019).

In the last decade of the 19th century, translocations of native wildlife were pioneered by Richard Henry, a caretaker on Resolution Island in Dusky Sound, Fiordland. Henry transferred hundreds of flightless birds, mainly kākāpō (*Strigops habroptilus*) and two species of kiwi (*Apteryx* spp.), to offshore islands from nearby areas of the mainland Fiordland in an attempt to secure them from invasive predators, mainly stoats (*Mustela erminea*) (Colbourne 2005). However, stoats soon arrived at Resolution Island and subsequently the other nearby islands in Dusky Sound, where kiwi and kākāpō were released, which compromised the chances of the newly-established populations surviving (Hill & Hill 1987). All translocated kākāpō and little spotted kiwi (*A. owenii*) eventually died out; however, introduced tokoeka (*A. australis*) still persist on three Dusky Sound islands: Long, Indian, and Parrot

Island (Colbourne 2005). Thus, although most of the early translocations were unsuccessful, Richard Henry did set up a pathway that has been widely followed by conservation practitioners since then (Colbourne & Robertson 2000; Armstrong et al. 2019).

From the 1960s, the number of translocations in New Zealand, particularly birds, rapidly increased (Miskelly & Powlesland 2013; Nally & Adams 2015). Despite many failures, several translocations resulted in significant successes in averting the extinction of tīeke–saddleback from the North (*Philesturnus rufusater*) and the South Island (*P. carunculatus*), black robin (*Petroica traversi*), or kākāpō (Jones & Merton 2012). Encouraged by translocation success stories, numbers of translocation projects and translocated taxa, including kiwi, kept increasing in the following decades (Miskelly & Powlesland 2013). Traditionally, most translocations were carried out by the Department of Conservation (DOC) and its predecessor, the New Zealand Wildlife Service. However, in the last few decades, community groups and iwi/hapū have had an increasing role in initiating and leading wildlife translocations (Parker 2008; Cromarty & Alderson 2013). Increasing interest in ecosystem restoration projects and associated translocations for advocacy purposes by local communities create a possible conflict of interest between species recovery and advocacy (Nally & Adams 2015). Such situations call for well-informed policies that would regulate the translocation approval process and best practice guidance to maximise the likelihood of positive translocation outcomes.

1.3 Translocations of kiwi

Kiwi are some of the most commonly translocated New Zealand birds (Cromarty & Alderson 2013). Kiwi are iconic birds to New Zealand and taonga/treasure for Māori, hence the large cultural importance of the species. Currently, there are five species of kiwi formally described separated into multiple subspecies or evolutionarily significant units (Shepherd et al. 2012; Weir et al. 2016; Undin et al. 2021). Three kiwi species are threatened: tokoeka, rowi (*A. rowi*), and roroa–great spotted kiwi (*A. maxima*, formerly known as *A. haastii*, Shepherd et al. 2021). The two most translocated species, little spotted kiwi and brown kiwi (*A. mantelli*) are classified as ‘at risk’ (Colbourne 2005; Robertson et al. 2017). Given their conservation status and role as a national identity symbol, kiwi receive large public attention and funding. Therefore, the number of community-led initiatives to reintroduce kiwi, often for ecosystem restoration and advocacy purposes, has been steadily increasing (Cromarty & Alderson 2013; Miskelly & Powlesland 2013). Still, there is an insufficient strategic direction for kiwi translocations and often inadequate awareness of translocation complexity and requirements among translocation proponents and managers (Germano et al. 2018).

Kiwi popularity has led to an increasing number of translocations; however, they have not been rigorously reviewed in the scientific or grey literature. There is a review of bird translocations in New Zealand by Miskelly and Powlesland (2013), conservation introductions in Australasia by Seddon et al. (2015), and an overview of kiwi translocations to islands by Colbourne (2005). Nevertheless, these works lack necessary detail on individual projects that is crucial to inform future directions in translocation use. The current kiwi recovery plan has identified the need to review kiwi translocations to determine success rates and factors influencing translocation outcome to assist the development of a national kiwi translocation strategy (Germano et al. 2018). Additionally, inconsistent monitoring and reporting on individual kiwi translocations' progress highlight the need for a clear policy mandating standardised monitoring to measure effectiveness and provide evidence for meeting translocation objectives (Nally & Adams 2015). Furthermore, reporting and publications of information on individual translocation projects, such as data on demographics, habitat, and detail on management actions, will allow sharing lessons learnt and improve the use of translocations in conservation management in general (Moro et al. 2015).

Addressing the gaps in knowledge on kiwi translocations requires a thorough translocation review, as concluded in the current kiwi recovery plan (Germano et al. 2018). The review needs to be based on detailed information about translocations, particularly translocation objectives, project area attributes, information on released birds, detail on post-release monitoring, together with applied management interventions. To address the widespread inconsistencies in post-translocation monitoring, a detailed description of monitoring methods demonstrated in real-world case studies should be published to inform project managers. Ideally, the monitoring methods would be suitable for a range of terrains, population densities, and all kiwi taxa. Therefore, case studies of kiwi species found in a rugged landscape and at low densities, such as roroa or tokoeka, provide an opportunity to demonstrate the universal utility of post-translocation monitoring methods in situations with higher demands on labour and challenging access.

1.4 Thesis structure

This thesis consists of three scientific articles that focus on kiwi translocations and monitoring translocated populations of roroa. Each chapter represents a standalone piece of research, including a review of literature and a discussion of the results. Therefore, in this section, I outline research questions and how the chapters link together.

In Chapter 2, I critically review information on all kiwi translocations and attempt to identify factors influencing translocation outcome, which addresses one of the research priorities of the current Kiwi Recovery Plan (Germano et al. 2018). This review comprises all available information on kiwi translocations and a literature review of reintroduction biology in the context of kiwi conservation management. It builds upon previously published translocation reviews and summaries by Miskelly and Powlesland (2013), Colbourne (2005), Colbourne and Robertson (2000), Atkinson (1990), and McHalick (1998). However, it substantially expands the information presented in these works with data obtained from project managers, the Department of Conservation's registers and databases, and publicly available historical records. As such, the review is based on information from 102 kiwi translocations between 1863–2018, and so it includes several translocations that did not appear in previous reviews or summaries.

The main research objectives of Chapter 2 are an assessment of translocation effort for each kiwi taxon, identification of trends in the use of kiwi translocations, evaluation of outcomes of translocations, and analysis of factors influencing translocation outcome. The translocation effort assessment is based on the numbers of translocated birds and translocation projects, including the temporal and geographical context. To analyse trends in kiwi translocations, it addresses the evolution of main translocation objectives. Additionally, it investigates the changes in the compositions of translocated birds based on their origin: wild-caught, captivity, or the Operation Nest Egg (ONE) – releasing subadults hatched in captivity from eggs collected in the wild (Colbourne et al. 2005; Gillies & McClellan 2013). Chapter 2 also summarises changes in the type of a translocation project area based on the predator status and if the project area was an island, unfenced mainland site, or fenced sanctuary. Subsequently, it classifies translocations into outcome categories while trying to avoid the risk of misclassifying early population establishment and initial growth as a long-term translocation success (Seddon 1999; Robert et al. 2015; Seddon 2015). Finally, Chapter 2 summarises monitoring type and effort and an analysis of post-translocation survival for likely factors influencing translocation outcome. Specifically, the analysis addresses the impact of the source type of translocated birds (wild-caught, captive, ONE) and the predator status and size of the project area. These findings have implications for the planning of future translocations and adaptive management of existing translocated populations. They also highlight the need for robust and standardised monitoring of translocated populations to allow adaptive management and sharing lessons learnt.

In Chapters 3 and 4, I focus on a recent translocation case study, a reintroduction of roroa–great spotted kiwi to the Nina Valley, Lake Sumner Forest Park, to demonstrate the utility of post-translocation monitoring methods.

Roroa is endemic to the South Island of New Zealand, mainly inhabiting montane beech and podocarp forest (McLennan & McCann 2002). Roroa is classified as a nationally vulnerable species (Robertson et al. 2017). The ‘nationally vulnerable’ status has been in place since 2008 (Hitchmough 2013; Robertson et al. 2013) when it worsened from ‘gradual decline’, which was last assigned in the 2005 assessment (Hitchmough et al. 2007). The current population (2018 estimate) is approximately 14 000 individuals (Germano et al. 2018), 2000 less than ten years prior (Holzapfel et al. 2008). The roroa range originally spanned across the whole north-west of the South Island but has shrunk by at least 30% since the Europeans’ arrival, mainly due to stoat predation of juveniles (McLennan & McCann 2002). The present roroa range is separated into Arthur’s Pass–Hurunui, Paparoa Range, Westport, and the north-west Nelson regions. However, there appear to be only two biological and genetically distinct populations: inland (Arthur’s Pass–Hurunui) and western (Paparoa Range, Westport, the north-west Nelson) (Taylor et al. 2021).

The Nina Valley translocation is the only roroa reintroduction of the inland population. It is part of a wider ecosystem restoration project and is motivated by efforts to restore roroa’s former distribution (Holzapfel et al. 2008; Hulsman et al. 2010; Morrison & Yong 2014). The releases took place between 2011–15, following only two other reintroductions of roroa’s western population to Lake Rotoiti, Nelson Lakes NP (2004), and the Flora Valley, Kahurangi NP (2010) (Miskelly &

Powlesland 2013). Unlike Lake Rotoiti or the Flora Valley, where only wild-caught birds were initially released (Gasson 2005; Toy & Toy 2020), the Nina population was founded by ONE birds (2011–13) and wild-caught birds were released a few years later (2015). Therefore, it was unclear whether released roroa in the Nina Valley would behave similarly to the other roroa translocations. Hence, the Nina translocation provided a unique opportunity to study post-release survival, dispersal, and population establishment. Moreover, various monitoring methods used in this study will inform future releases and adaptive management.

Chapter 3 investigates the population establishment phase of the translocated population focusing on post-release effects, such as dispersal and reduced survival following the release (Tavecchia et al. 2009; Armstrong & Reynolds 2012). Although this phase is not indicative of the overall translocation outcome by itself, successful population establishment in the project area and the magnitude of post-release effects are crucial for long-term reintroduction success (Armstrong et al. 2017). The assessment of post-release effects also enables estimating the sufficient overall number of released animals to establish a genetically viable population (Weeks et al. 2015) and thus informs adaptive management of the translocated population (Pacioni et al. 2020).

Radio telemetry was the primary tool for monitoring the survival and dispersal of roroa in the two years following the release of wild-caught adults in the Nina Valley in 2015. By triangulation and homing of the radio-tagged birds (Neill & Jansen 2014; Colbourne et al. 2020), it was possible to establish an approximate dispersal path of the birds post-release. Subsequently, it enabled an estimation of the utilisation distribution (Kranstauber et al. 2012) to indicate an area the roroa used during the dispersal. Additionally, Chapter 3 focuses on changes in the ranging behaviour of the roroa before-and-after the translocation of the wild-caught birds. Several other studies report on seasonal (McLennan & McCann 1991; Keye et al. 2011; Jahn et al. 2013) or inter-seasonal home ranges of roroa (Toy & Toy 2020), but the effect of translocation on home range size have not been studied in a kiwi species yet.

Monitoring translocated populations by radio telemetry is case-specific but common during the first year post-release (Colbourne et al. 2020). However, radio-tracking most translocated birds for more than five years is rare (Toy & Toy 2020). Because for long-lived species, such as kiwi, it would be premature to declare translocation success after such a short period (Seddon 2015), it is essential to keep monitoring translocated populations by other means.

Chapter 4 explores how the data from increasingly popular passive acoustic monitoring (PAM) (Teixeira et al. 2019) can be effectively used in long-term monitoring programmes for roroa and other kiwi species. A large amount of data that PAM can easily collect allows an estimation of detection probability (Gu & Swihart 2004) and true site occupancy (MacKenzie et al. 2002). Therefore, occupancy analysis provides a useful tool to monitor temporal population changes (MacKenzie et al. 2005) by incorporating covariates affecting detectability and occupancy (MacKenzie et al. 2018).

Acoustics-based occupancy analysis is used to monitor the translocated population of roroa in the Nina Valley following the cessation of radio telemetry monitoring. This approach allowed an assessment of population establishment in the project area and provided sufficient baseline data for future periodic acoustic surveys to evaluate long-term translocation success. Chapter 4 also investigates the impacts of birds' removal on the source population to avoid future overharvesting or other detrimental effects (Dimond & Armstrong 2007). Occupancy analysis based on repeated acoustics surveys is used to assess changes in the source population and to determine if and how fast the vacated territories would be re-occupied. Therefore, investigation of occupancy analysis' utility can expand the toolkit of monitoring methods for kiwi translocations projects, particularly during the growth and self-regulation phases post-release. Acoustics-based occupancy analysis can assist with the evaluation of long-term translocation outcomes and contribute to the advancement of reintroduction biology in the context of kiwi conservation management.

Chapter 2

Kiwi translocation review: are we releasing enough birds and to the right places?

2.1 Introduction

Wildlife translocations, together with pest eradications, are perceived as achievements of conservation management in New Zealand. These management tools have built upon years of lessons from initial “trial-and-error” efforts (Brichieri-Colombi & Moehrenschlager 2016). Kiwi management over the past hundred years illustrates this evolution of New Zealand conservation, and its understanding helps us to advance the use of reintroduction biology in practical management.

Around the turn of the 19th century, kiwi translocations were reactive measures to save populations from the imminent threat of predation (Armstrong et al. 2015) or habitat loss due to logging in the second half of the 20th century (Colbourne 2005). Early transfers between mainland sites were mostly unsuccessful as most or all birds died or dispersed (Saunders 1995; Miskelly & Powlesland 2013), but many early translocated island populations have persisted until the present (Colbourne & Robertson 2000). Advances in pest eradications on islands (Armstrong & McLean 1995), landscape-scale predator control (Saunders & Norton 2001), and fenced sanctuaries have allowed successful releases of kiwi and other wildlife valued by communities throughout New Zealand (Burns et al. 2012; Smuts-Kennedy & Parker 2013; Innes et al. 2015a). Those advances in predator control techniques, mainly trapping and poisoning, have led to the creation of larger ecosystem restoration areas (Saunders & Norton 2001; Innes et al. 2019), which also enabled effective management of remnant kiwi populations *in situ*.

Kiwi are the most translocated bird species in New Zealand (Cromarty & Alderson 2013; Miskelly & Powlesland 2013) and the number of new projects and released birds are steadily increasing. Originally, all translocated kiwi populations were sourced from existing wild populations. However, in 1995, Operation Nest Egg (ONE), a new method of intensive management of kiwi, was developed to source birds to supplement existing populations or to establish new populations. Operation Nest Egg involves removing kiwi eggs from the wild, incubating and hatching chicks in a captive facility, and keeping them in a predator-free environment (crèche) until they reach a safe size (typically 0.8–1.2 kg) to fend off stoats (*Mustela erminea*), before releasing them back into the wild (Colbourne et al. 2005; Gillies & McClellan 2013). This method has been widely used for several kiwi species (Innes et al. 2015b). Initial successes of these developments and increased involvement of community groups, iwi and hapū, resulted in a boom of kiwi translocations over the last two decades. As the number of translocations grew, so did the translocation objectives. The objectives vary: creating secure populations and establishing kōhanga sites/source populations to supply wild kiwi for releases elsewhere (Innes et al. 2016), meta-population management (Robertson et al. 2019b), ecosystem restoration projects (Innes et al. 2019), and mitigation transfers (Colbourne & Robertson 2000).

There is a lack of information on kiwi translocations and their outcomes, which makes it difficult to assess overall translocation effectiveness, address possible issues, improve translocations as a tool, and identify the role that translocations play in kiwi recovery management. Currently, there is no database of all kiwi translocations, and the existing registers lack sufficient detail to fully document the scope of translocations and objectives for which they were carried out. Outcomes of translocations remain mostly unpublished and are often unreported, especially for those that failed or had uncertain outcomes (Fischer & Lindenmayer 2000; Miller et al. 2014; Gitzen et al. 2016).

A thorough assessment of translocations is critical for the refinement of this management tool (Brichieri-Colombi & Moehrenschlager 2016) to maximise the probability of positive outcomes (Batson et al. 2015). Here we review the information from kiwi translocations with a focus on the last four decades. Specifically, we assess the level of translocation effort for each taxon and the evolution of translocation objectives over time. Additionally, we evaluate translocation outcomes and investigate contributing factors, such as source type of released birds and predator status at the release site. Finally, we identify issues with the past and current practice and provide recommendations addressing the issues faced by the translocation projects in accordance with the latest Kiwi Recovery Plan (Germano et al. 2018).

2.2 Methods

2.2.1 Definitions and terminology

Translocations are intentional movements of wildlife by humans from one place to another and are often used in conservation management (Seddon 2010; Seddon et al. 2014). To qualify as a 'conservation translocation', the aim of the movement should be to produce a measurable conservation benefit at a population, species, or ecosystem level. Other types of translocations are carried out solely to benefit the translocated individuals, such as mitigation translocations and rehabilitation releases, or for aesthetic reasons/enjoyment by landowners (IUCN/SSC 2013). In recent decades, translocations have been also carried out for conservation advocacy and community interest (Parker 2008).

We separate conservation translocations into 'conservation introductions' – releases of species or particular taxa outside their historical range, and 'population restorations' – releases within their range: an area in which the taxon naturally occurs or was known to occur in the past (IUCN/SSC 2013). Additionally, we discern between two types of population restorations. First, a 'reintroduction' happens where a species has become locally extinct. Second, a 'reinforcement' (alternatively 'supplementation') happens when a population of the species is still present at the release site (Seddon et al. 2012). We also differentiate between two types of conservation introductions based on their purpose. First, 'assisted colonisation' aims to create a secure population to avoid species extinction. Second, 'ecological replacement' allows the translocated species to fulfil an ecological function of a previously extinct taxon (Seddon et al. 2012). Translocations can consist of a single release or can take place over several years and consist of multiple releases until the desired number of translocated animals is achieved (Griffith et al. 1989).

2.2.2 Data collection

We collated a dataset of 102 kiwi translocation projects that occurred between 1863 and 2018 from a large variety of sources, which included: the Department of Conservation's translocation register, the Zoo and Aquarium Association's brown kiwi studbook, published reviews and summaries (Atkinson 1990; McHalick 1998; Colbourne & Robertson 2000; Colbourne 2005; Miskelly & Powlesland 2013), websites "Reintroduction Projects in New Zealand" (www.massey.ac.nz/~darmstro/nz_projects.htm) and “Avian Reintroduction & Translocation” (www.lpzoosites.org/artd/; both accessed in July 2020), journal articles, published and unpublished reports, translocation proposals, management plans, monitoring data and various relevant information provided by project managers and associates. Historical translocations were typically poorly documented, or the documents were difficult to locate and access. For that reason, we generally focused on the 76 translocations occurring in the last four decades (1979–2018), with an emphasis on the 60 translocations from the last two decades (1999–2018), for which post-release monitoring data and supplementary information were more available.

We reviewed all conservation translocations but also included mitigation translocations, rehabilitation releases (if these were supplementary to a wider kiwi conservation project), and other known historical translocations. Some of these projects extend beyond the conservation translocation spectrum (Seddon 2010; IUCN/SSC 2013), but we included them in the dataset to have a complete picture of the use of kiwi translocations in New Zealand. Specifically, we included all translocations of wild birds, birds from captive facilities, and intended permanent releases of juveniles and subadults from the ONE programme, provided that these birds were released at a location different to their source site. In contrast, temporary translocations of ONE birds to a crèche site, ONE releases to the original source population, transfers to captivity, and other releases not intended to found a permanent kiwi population were excluded from our review. Releases of different kiwi species to a single site were treated as separate translocation projects. Also, translocations of the same species into the same site were treated as separate projects if there were more than 15 years between the releases and therefore were unlikely a part of the same series of releases.

To enable a thorough analysis of the translocation information, we recorded various parameters for each translocation and, where possible, for each release event within a translocation. The parameters included: taxon; year; release site and its area size and type (mainland, island, fenced sanctuary); source site; founder source type (wild, captive, ONE); number of released birds; post-release survival; how many birds dispersed and attempted to breed; type of monitoring (radio telemetry, call counts, recapture); length of monitoring; presence and type of introduced mammalian predator control; translocation objective; and lead entity (DOC, community group, iwi/hapū). For translocations to islands and fenced sanctuaries, the whole area of an island or a sanctuary was considered as the translocation project area. At mainland unfenced sites, the project area was equivalent to the size of the area under predator control, in which the released birds were expected to settle. At sites with no predator control, the size of a reserve or another designated protected area where the kiwi were released, was considered as the project area. Where additional predator control buffer zones were created around the project area, such zones were not deemed part of the project area, as their main purpose was to reduce the probability of predator incursions rather than for the birds to settle in them.

The geographic scope of original species ranges and kiwi taxonomy were based on the current state of knowledge as described in Weir et al. (2016), Shepherd et al. (2012), and Germano et al. (2018). All taxa or evolutionarily significant units of brown kiwi (*A. mantelli* – Northland, Coromandel, Western, Eastern) and tokoeka (*A. australis* – Haast, North Fiordland, South Fiordland, Rakiura) were analysed separately. Great spotted kiwi/roroa (*A. haastii*), little spotted kiwi (*A. owenii*), and rowi (*A. rowi*) were each treated as a single taxon unit because they have been managed as such. All translocations to islands, where kiwi had not been recorded before, were considered as introductions, similar to Seddon et al. (2015). Only early translocations of tokoeka to Resolution Island (1895–1898), little spotted kiwi to Cooper Island (1903), and brown kiwi to Te Hauturu-o-Toi/Little Barrier Island (1903–1919) were considered reinforcements due to presumed naturally-occurring populations still there at the time (Henry 1895; Palma 1991; Colbourne 2005).

2.2.3 Assessing translocation outcomes

Evaluating translocation outcome is challenging because there is no scientific consensus on what constitutes translocation success (Chauvenet et al. 2013; Robert et al. 2015). Success criteria generally depend on the set objectives and type (e.g., reintroduction vs reinforcement) of each particular translocation (Armstrong et al. 2019). Moreover, most reported outcome evaluations focus on the establishment phase of a translocated population. However, even though successful establishment and growth are necessary for a population to be viable in the long-term, they do not predict the ultimate translocation outcome (Robert et al. 2015). Therefore, to accurately evaluate translocations, we need to distinguish between three post-release phases: population establishment, growth, and self-regulation (Sarrazin 2007; Armstrong & Seddon 2008).

Assessing and comparing the outcomes of kiwi translocations is further challenged by kiwi life-history and ecology. Several decades may be required before a translocated population reaches the regulation phase and so be considered as successfully persisting for the long term, over multiple generations. All kiwi species are long-lived with a possible life expectancy exceeding 50 years for most taxa (Heather & Robertson 2015). Kiwi are irregular breeders and require 1–5 years before they reach sexual maturity, with variation among the taxa (Heather & Robertson 2015). All kiwi are sexually dimorphic, which allows for relatively simple sexing of adults in the field, but it is not possible to reliably assign age once mature (McLennan & McCann 1993; Robertson & Colbourne 2017). These characteristics make it difficult for long-term monitoring of translocated populations consisting of unmarked individuals.

We based our assessment of translocation outcomes on the assumption that the primary objective of most translocations is to establish or restore a population with a high probability of persistence (Converse & Armstrong 2016). Such populations require sustained population growth (Armstrong & Reynolds 2012) and an adequate number of founders to minimise the loss of genetic diversity (Weeks et al. 2015). This objective aligns with the main recovery goals for kiwi management: grow populations of all species by at least 2% per year and maintain their genetic diversity (Germano et al. 2018). To achieve sufficient retention of genetic diversity, at least 40 unrelated individuals of brown kiwi need to be released initially (Weiser et al. 2013; Weiser 2014). This number of starters (initially released birds) is expected to maximise the probability of persistence and avoid genetic deterioration of introduced populations (Weeks et al. 2015; Frankham et al. 2017). The target of releasing 40 individuals to a project area with a carrying capacity for at least 100 pairs was adopted for all kiwi taxa by the official Department of Conservation guidelines (Sporle 2013; Robertson & Colbourne 2017; Department of Conservation 2018), and we included it among the criteria for a successful introduction or reintroduction in this review. We excluded reinforcements from the outcome assessment given that it was not usually possible to attribute the contribution of the released individuals to population growth or genetic diversity.

Assessment of the translocation outcome is based on the point in time when the assessment was made, and hence it may change subsequently (Wolf et al. 1996; Seddon 1999). We set the minimum assessment timeframe of 15 years to allow a sufficient period for the population to start growing following the post-release effects and acclimation period of suppressed growth rate (Converse & Armstrong 2016). This window leaves out populations in the establishment phase, but allows assessment of projects in the growth and regulation phases. Armstrong and Reynolds (2012) and Robert et al. (2015) argue that the ultimate evaluation of translocation success should happen only once an introduced population reaches carrying capacity. Nonetheless, populations at different sites and of different taxa reach carrying capacity at different times, and so, for practical reasons and for the ability to compare the projects, we chose an intermediate duration of 15 years. This 'in progress' period was expected to provide enough time for the released juveniles or subadults of all taxa to mature and breed, as well as for their offspring to start breeding. This timeframe provides a reasonable indicator of success in the population growth phase without requiring data across the decades that may be necessary to reach the regulation phase at all sites.

2.2.4 Translocation outcome definitions

We divided translocation outcomes of introductions and reintroduction projects into five categories, which are similar to the categories used in reviews by Miskelly and Powlesland (2013) and Brichieri-Colombi and Moehrenschlager (2016):

1. **Likely successful**: a successfully established population growing at least 2% on average per annum (finite rate of increase) over a minimum of 15 years, after 40+ individuals were released. Also, self-regulating populations of 100+ pairs founded by 40+ individuals. Further immigrants likely need to be periodically added or immigration facilitated for maintaining genetic diversity.
2. **Requiring further management**: an established population 15+ years after the latest release with less than 40 released birds in total, which will likely require more major releases or ongoing genetic management. Also, populations below the 2% average annual growth rate and kōhanga sites, more than 15 years since the latest release.
3. **In progress**: a population less than 15 years since 40+ birds were released or a population with less than 40 birds released within the last 15 years.
4. **Unsuccessful**: a population showing signs of decline below 50% of the initially released birds at 15+ years post the latest release, or in cases where released birds were removed from the project area.

5. **Not assessed**: a reinforcement translocation or an introduced hybrid population of individuals originating from different kiwi species.

2.2.5 Data analysis

Project-level analysis

First, we carried out an exploratory analysis of translocation projects and how they evolved. Specifically, to assess the trends in numbers of new translocations in each decade, we plotted individual projects using decadal bins. Similarly, to display trends in the type of release sites over time, we binned the numbers of all translocated birds in one-year intervals. To explore translocation effort for each taxon over the last 40 years, we plotted the numbers of translocated birds and translocation projects and compared the number of translocated birds in this period with 2018 populations estimates (Germano et al. 2018). To inspect the geographic distribution of translocations, we plotted central points of project areas onto a map of New Zealand. Because some projects areas were either close to each other or overlapping, we plotted the points with a jitter factor of 0.01 arc degrees. Subsequently, we compared source types of translocated birds by displaying numbers of released individuals for each species based on their origin (wild-caught, captive, ONE). Then, we evaluated the habitat type and size of project areas. To assess the evolution of motivations for translocations, we compared primary and secondary objectives for periods before-and-after 1989 to highlight the shift of dominant objectives in the last three decades. Finally, we estimated proportions of translocations led by different entities and based on reporting in the literature.

In the following step, we assessed outcomes of translocations and assigned them into the categories, as mentioned above. Following this assessment, we carried out a sensitivity analysis to categorise translocations with adjusted criteria using three different values for numbers of released birds (35, 40, 50) and three different assessment timeframes (10, 15, 20 years). This step enabled us to evaluate how variation in the criteria threshold affects the number of likely successful translocations.

Individual-level analysis

Subsequently, we explored how different characteristics of translocation projects affect the survival of released birds. Initially, we assessed differences in the magnitude of post-release effects, specifically mortality and dispersal. Then, we modelled mean survival times based on translocation characteristics. We used a set of Bayesian time-to-event generalised mixed effect models. The response variable of the model was the survival time of the released birds in the project area. We used a Bayesian approach for three main reasons. First, Bayesian models have been shown to be better suited to deal with small sample sizes similar to those in our dataset, and therefore ideal for modelling translocated populations (Chauvenet et al. 2015). Second, the flexibility of Bayesian models allowed incorporating random effects in a time-to-event regression framework. Third, and most importantly, this approach enabled us to better understand the uncertainties of the model estimates.

To fit the model, we used the R package brms 2.14.4 (Bürkner 2017). We used a normal prior ($\mu = 0$, $\sigma = 5$) for all the population-level effects and an exponential prior ($\lambda = 1$) for our response variable. We used four independent Markov chains (Hamiltonian Monte Carlo algorithm) with 5000 iterations each. Trace plots of all Markov chains suggested model convergence and Gelman and Rubin's potential scale reduction factor $\hat{R}$ for all model estimates was below 1.002.

Data on post-translocation survival in the project areas was available in most cases for at least a subset of the released birds in projects since 1979. We also included rare cases of known survival from incidental reports before 1979 (six projects) and reports of local extinctions from failed translocations. The exact survival time of translocated birds was usually unavailable, and therefore our response variable was structured as a range of minimum and maximum possible survival times. The range of possible survival times was determined for 41% of the translocated birds. Often the time ranges were large, but they still provided valuable information about the differences between the translocation characteristics.

For those birds from all taxa for which we were unable to determine maximum possible survival time, we used a value of 50 years, in line with the estimated life expectancy of several kiwi species (Heather & Robertson 2015). For instance, the survival information of 10% of the birds was available only for the full first 12 months post-release after which monitoring stopped, or data were unavailable. Therefore, the exact survival time for these individuals could be anywhere between 1–50 years. These ranges were included as censoring intervals in our time-to-event model. We modelled the survival ranges using a log-normal family distribution. This distribution assumes a constant hazard rate and is commonly used to model time-to-event data when the rate of the event peaks at intermediate levels of the expected lifetime.

In the model, we included five variables as predictors of the mean survival time of translocated birds, two of them as fixed effects and three as random effects. As fixed effects we included the factors where we more expected their potential impact on management decisions: 1) the source type of the translocated birds (either wild, ONE, or captive), 2) the predator status (either predator-free, predator-managed by trapping/poisoning/both, or predator-present and not managed). We separated sites with predator management into two groups based on area size under predator control delimited by the median area for mainland unfenced sites. As random effects we included: 1) the kiwi taxon, 2) the translocation project, and 3) whether the birds were part of an introduction/reintroduction or a reinforcement translocation. Although this last predictor could have been also included as a fixed effect, we decided to model it as a random effect because later releases of an introduction/reintroduction project may resemble a reinforcement project and hence it was not always possible to unequivocally categorise individual releases. Including the translocation type as a random effect allowed us to account for the variability around this factor without the challenges of interpreting model estimates for each of the two translocation types. Subsequently, we tested for differences within our explanatory variables and presented the strength of evidence based on posterior probability. At mainland unfenced sites, birds were considered to have survived if they remained in the project area to act as population founders. Birds dispersing outside the project area were assumed to be unlikely to breed successfully due to limited or no predator control. However, birds that dispersed and subsequently returned to the project area, or birds that were brought back by managers and then stayed, were considered to have survived.

2.3 Results

2.3.1 Number of translocations and translocated birds over time

We identified 102 translocation projects that occurred between 1863 and 2018 (Fig. 2.1). It was not always possible to establish the numbers of released birds, particularly for the historical translocations. However, we were able to determine that at least 2572 birds were translocated through the end of 2018 (Fig. 2.2). After the initial wave of translocations in Fiordland at the turn of the 19th century and releases to Kapiti and Te Hauturu-o-Toi/Little Barrier islands in the early 20th century, there was little such activity until the 1970s. From then on, the number of new projects has generally been increasing every decade. The highest increase of translocation events and the number of translocated birds occurred during the last two decades, with 76% of all translocated kiwi released in the last 15 years (2004–2018).

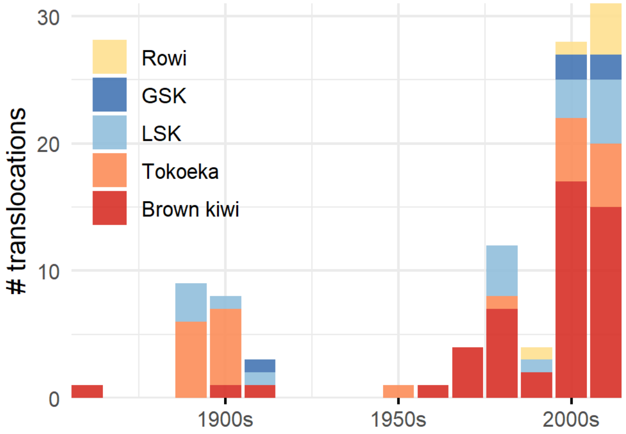

Figure 2.1 Numbers of kiwi translocation projects between 1860–2018 (n = 102). Project numbers increased rapidly in the last two decades. Projects were assigned to the decade in which they started. First recorded translocation was in 1863, the last in December 2018. Each colour represents a different kiwi species: rowi, great spotted kiwi (GSK), little spotted kiwi (LSK), tokoeka, and brown kiwi. Projects marked as rowi in the 1990s and 2000s show translocations of LSK/rowi hybrids, while in the 2010s only translocations of rowi occurred.

In the period 1979–2018, for which we have the most information, 76 translocation projects were carried out. These projects took place over five years on average (range: 1–17 years), and in many cases, releases are likely to continue. For the same period, we recorded at least 817 separate release events (releases on consecutive days were counted as a single event). The median number of release events per translocation was four (1–73), with the overall length and number of releases increasing markedly in the last two decades. The median number of released individuals per project was 25 birds (1–169, n = 74; two projects with missing and incomplete data were not included). The median number of released birds for introduction and reintroduction projects was 29 (2–169, *n* = 53), while for reinforcements it was 11 (1–114, *n* = 21). The sex ratio of adult males to females was 1.1:1, the reporting on age classes was highly inconsistent among the projects.

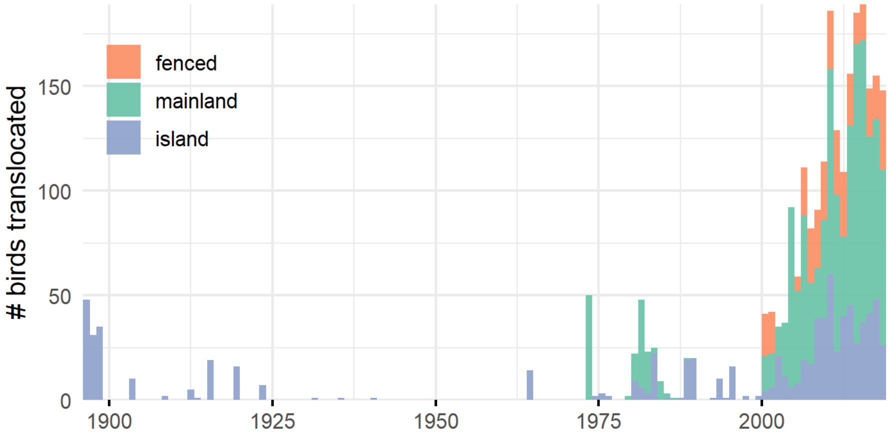

Figure 2.2 Numbers of translocated birds for different release site types between 1895–2018 (*n* = 2572). Numbers of translocated birds show a steady increase since 2000. Each colour represents a different type of release site: fenced sanctuaries (ring- or peninsula-fenced), unfenced mainland sites, and islands. Information about numbers of released birds from several translocations before the 1990s may be incomplete or missing; numbers of birds released before 1895 are unknown.

In this period, most translocation projects (55%) were of brown kiwi, followed by little spotted kiwi (17%), tokoeka (14%), great spotted kiwi (5%), rowi (5%), and hybrids of rowi and little spotted kiwi (3%) (Fig. 2.3). Proportions of translocated birds relative to the current (2018) total population estimates differed considerably among kiwi taxa (Fig. 2.4). The rarest taxa, rowi and Haast tokoeka, together with little spotted kiwi, had the highest percentage of translocated birds in the last four decades. All brown kiwi taxa also had substantial proportions of translocated birds relative to their total populations, while for more populous South and Stewart Island taxa less than one per cent of their extant populations were translocated birds.

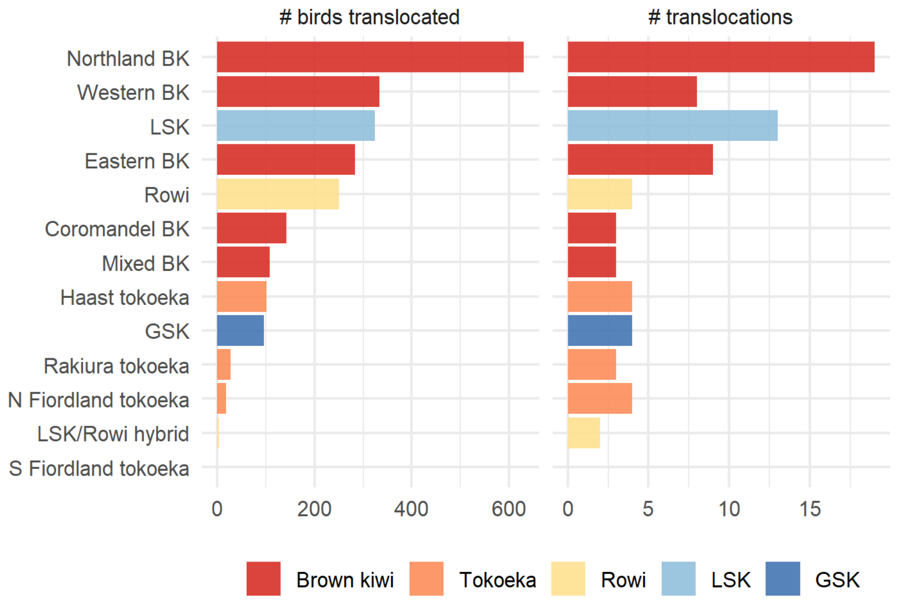

Figure 2.3 Numbers of translocated birds and translocation projects by kiwi taxon between 1979–2018. Brown kiwi, particularly the Northland taxon, were most translocated taxa, whereas the populous South and Stewart Island taxa were least translocated. Each bar represents either a different species: little spotted kiwi (LSK), rowi, great spotted kiwi (GSK), or a taxon/evolutionarily significant unit of brown kiwi (BK): Northland, Western, Eastern, Coromandel, and mixed brown kiwi of various origin, or tokoeka: Haast, Rakiura, North Fiordland, and South Fiordland. No South Fiordland tokoeka releases were recorded during this period. Each colour represents a different kiwi species, LSK/rowi hybrids are displayed in the same colour as rowi.

2.3.2 The spatial pattern of translocations

The geographic distribution of kiwi translocation projects was unevenly spread across New Zealand. Since the 1860s, most of the release sites (63%) were in the North Island, or the surrounding offshore islands. Only 33% and 4% of all translocations were to South and Stewart Island sites, respectively (Fig. 2.5). Most of the release sites (86%, *n* = 87) were unique to a single kiwi translocation project. However, 12 release sites received birds as part of two or three different translocation projects, with seven of those receiving at least two different species.

Resolution, Long, and Anchor islands in Dusky Sound, Fiordland, had both tokoeka and little spotted kiwi released on them in the late 19th century. Even though little spotted kiwi had not persisted at any of those sites, the species was re-released on Anchor Island in 2015. Kapiti and Te Hauturu-o-Toi/Little Barrier islands both received three kiwi species in the early 20th century. At Kapiti Island, little spotted, brown kiwi, and tokoeka were released, but only little spotted kiwi thrived there, while just a small population of likely brown kiwi and tokoeka hybrids remain on the island (Colbourne 2005). At Te Hauturu-o-Toi, brown, great spotted kiwi, and probably a single tokoeka were released, but only brown kiwi persisted. Mana Island first held a small hybrid population of rowi and little spotted kiwi, which was later removed, before the island received rowi two decades later. Among the recent projects, only Cape (Kidnappers) Sanctuary has two kiwi species – brown and little spotted kiwi, which are held separately. Brown kiwi were introduced to Mokoia Island for the second time after a previously failed attempt. The remaining four island projects (Motukawanui, Tiritiri Matangi, Red Mercury, and Ulva) released kiwi to supplement previously introduced populations.

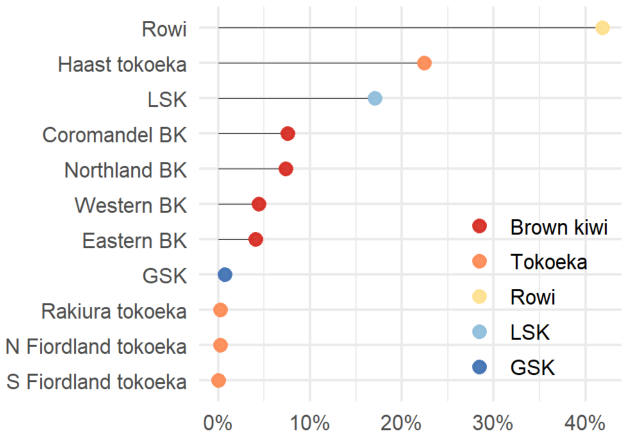

Figure 2.4 Percentages of translocated birds between 1979–2018 relative to the 2018 population estimates for each kiwi taxon (Germano et al. 2018) indicating translocation effort. For the three rarest taxa, the share of translocated individuals relative to their overall populations was the highest, whereas, for the populous South and Stewart Island taxa, the share of translocated individuals was less than 1%. Taxa as in Figure 2.3; hybrids and mixed provenance brown kiwi translocations are not included.

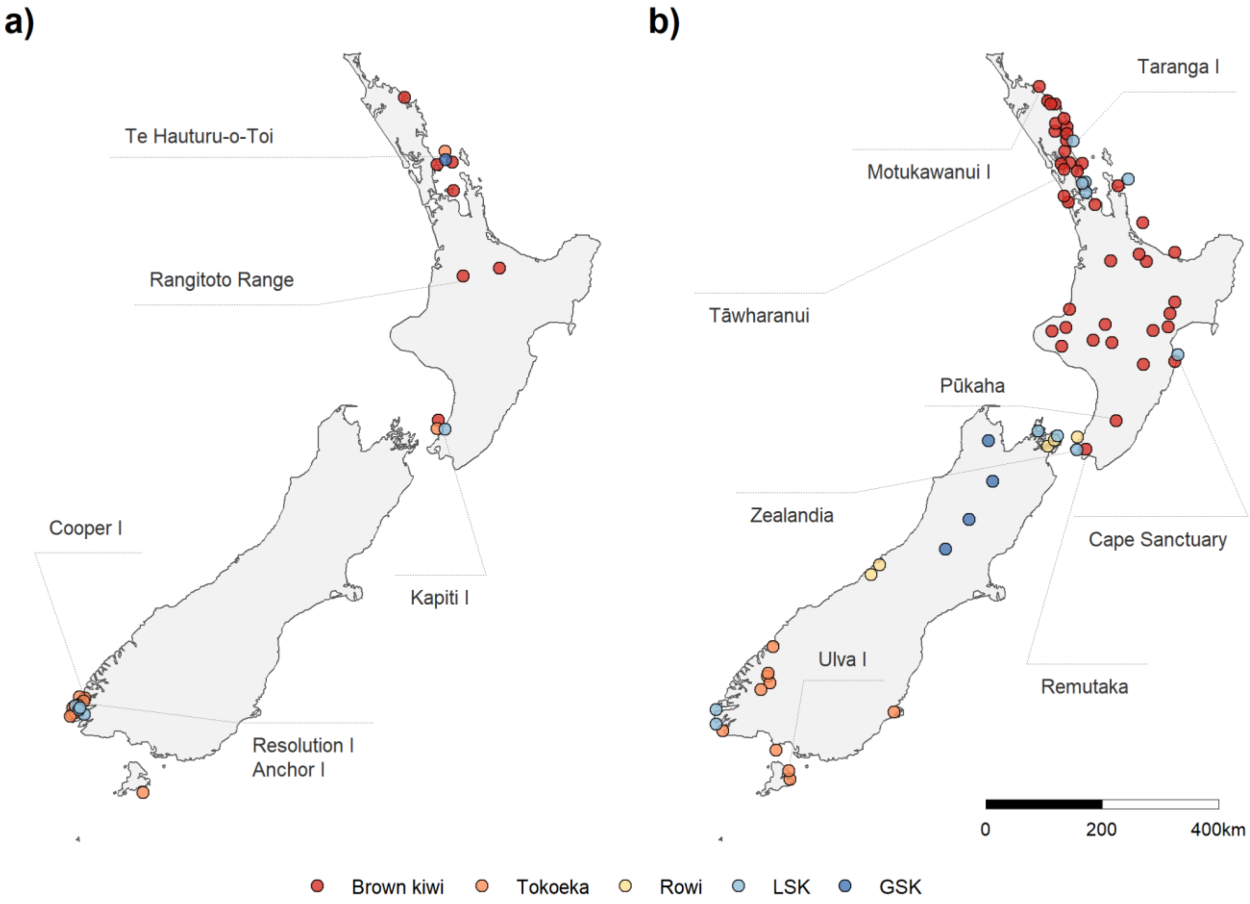

Figure 2.5 Sites of kiwi translocation projects for (a) historical (1863–1978) and (b) recent (1979–2018) periods. Each colour represents a different kiwi species: brown kiwi, tokoeka, rowi, little spotted kiwi (LSK), and great spotted kiwi (GSK). Translocations of LSK/rowi hybrids are grouped with translocations of rowi. Sites where more than one translocation project of the same species occurred, are shown only as one point. Sites where more than one species of kiwi were released (e.g., Kapiti Island, Te Hauturu-o-Toi/Little Barrier Island, or Cape Sanctuary) have points for each species displayed with a jitter factor of 0.01 arc degrees.

2.3.3 Translocation type

Most kiwi translocations (58%) were introductions to places where the kiwi taxa were not known to occur previously, such as on offshore islands, or to areas on the mainland outside of their historical range. Reintroductions (within the historical range) accounted for only 19% of all translocation projects. Finally, 24% of all translocations were reinforcements of either naturally existing populations, or, in four cases, populations that were previously established by earlier introductions (see above).

Conservation introductions of kiwi outside their historical range have been the dominant group of translocations. These introductions were mostly carried out either as assisted colonisations to islands, where kiwi were not previously recorded. In other cases, they were ecological replacements of extinct kiwi taxa, such as the introduction of brown kiwi to Pūkaha (2003) and the Remutakas (2006), a range previously occupied by extinct North Island rowi (Weir et al. 2016).

Reintroduction attempts first appear in the late 1970s/early 1980s in the Auckland region, with most of them occurring after 2000. Similarly, reinforcements of existing populations were mostly carried out after 2000, except for three early translocations to Resolution, Cooper, and Te Hauturu-o-Toi/Little Barrier islands, as mentioned earlier. However, despite the recent increase of reintroductions and reinforcements, conservation introductions remained the largest group representing 40% of kiwi translocations in the last 20 years (1999–2018), followed by reinforcements and reintroductions with 35% and 25%, respectively.

2.3.4 Source of translocated birds

Before 1995, only wild-caught kiwi were translocated. With the introduction of the ONE programme in 1995 and releases of brown kiwi from captivity, the number of translocations relying only on wild-caught animals has declined markedly. Only 35% of translocations comprised exclusively of wild-caught animals in the last two decades. The share of translocation projects releasing exclusively ONE sourced birds in the same period increased to 25%, and only 5% (three minor reinforcement translocations) comprised birds solely from captivity. The remaining 35% of translocation projects consisted of a mix of birds sourced from a combination of wild, ONE, or captive populations. The shift towards ONE releases is most apparent for brown kiwi, rowi, and Haast tokoeka (translocations comprised exclusively of ONE birds) (Fig. 2.6). Translocations of little spotted kiwi, Fiordland tokoeka, and Rakiura tokoeka were all sourced solely from wild-caught birds. Great spotted kiwi translocations were predominantly sourced from wild-caught birds.

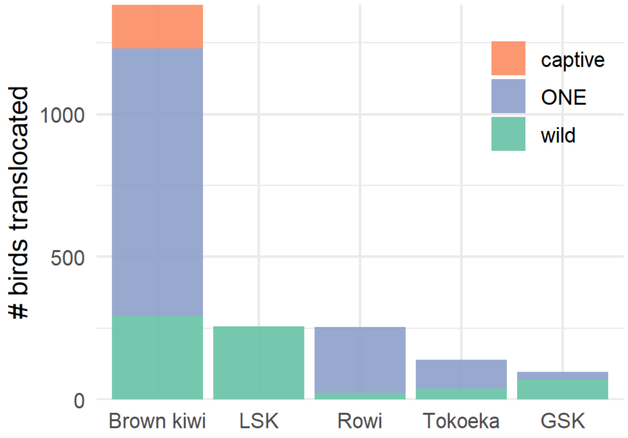

Figure 2.6 Numbers of translocated birds by source type between 1995–2018, following the introduction of the Operation Nest Egg (ONE) programme and releases from captivity. Most translocated brown kiwi, rowi, and tokoeka (Haast tokoeka) are sourced from the ONE programme. Before 1995, all translocated kiwi were wild-caught birds. Each bar represents a different kiwi species: brown kiwi, little spotted kiwi (LSK), rowi, tokoeka, and great spotted kiwi (GSK).

2.3.5 Habitat

Most kiwi translocation projects were to islands (55%), particularly during the early translocation period. There were no known cases of introductions or reintroductions to the mainland until the 1970s (Colbourne & Robertson 2000). Overall, 36% of translocations were to open mainland sites and the other 9% to ring- or peninsula-fenced sanctuaries. Translocations to the mainland became more frequent after 2000. Between 1999–2018, the share of translocations to islands dropped to 33%, while the share of translocations to mainland sites and fenced sanctuaries increased to 52% and 15%, respectively.

The median translocation project area was 800 ha (8–20 887, *n* = 100). Nevertheless, there were substantial differences in area among the translocation projects and, in several cases, the area size changed between individual releases for a single project. In the period 1979–2018, the median area of island projects was 218 ha (16–1509, *n* = 31), and fenced sanctuaries 450 ha (124–3363, *n* = 9), while mainland unfenced sites had a median area of 3000 ha (41–20 000, *n* = 35).

Mainland sites substantially differed in size based on translocation type. Reintroduction and introduction project sites on the mainland had a median area of 1350 ha (41–19 000, *n* = 19), whereas reinforcement sites stretched over a median area of 3750 ha (210–20 000, *n* = 16).

Nearly half of all the release sites (49%) were considered free from the main predators targeting kiwi (stoat *Mustela erminea*, ferret *M. furo*, dog *Canis familiaris*, cat *Felis catus*) at the time of the releases, while 18% had one or more of these predators present and no predator control. These included sites with cats as the only main kiwi predator. The remaining 33% of sites had some level of sustained predator control management. The first translocation projects with ongoing predator control appeared only in the last two decades. Before 1999, 43% of translocations (*n* = 42) were to sites with no predator control (the last of these were in the 1980s), while 57% of translocations were to presumed predator-free sites, although anecdotal evidence suggests that some of them were invaded by predators soon after the kiwi releases (Colbourne 2005). Between 1999–2018, 43% of translocations (*n* = 60) were to predator-free sites, and 57% had some kind of predator control; 17% relied on trapping, 35% on a combination of trapping and poisoning (mostly by 1080/sodium fluoroacetate), and 5% relied solely on aerial poisoning as a means of control.

2.3.6 Translocation objective

Historical records and the available literature indicate that early kiwi translocations were predominantly driven by emergency/mitigation efforts and the establishment of secure populations due to expanding ranges of invasive predators, particularly around the turn of the 19th century. In the 1970s and 1980s, mitigation and emergency transfers were predominant due to the loss of habitat in Northland, mainly driven by logging of native forest. Translocations for meta-population and genetic management appeared in the 1980s for little spotted kiwi and have continued since. In the last three decades, the range of translocation objectives has diversified (Fig. 2.7). Ecosystem restoration and establishment of secure populations were typical objectives across all taxa while conservation advocacy appeared mainly among brown kiwi translocations. The rarest kiwi taxa, Haast tokoeka and rowi, were translocated chiefly to establish secure populations and to serve, eventually, as kōhanga/source sites.

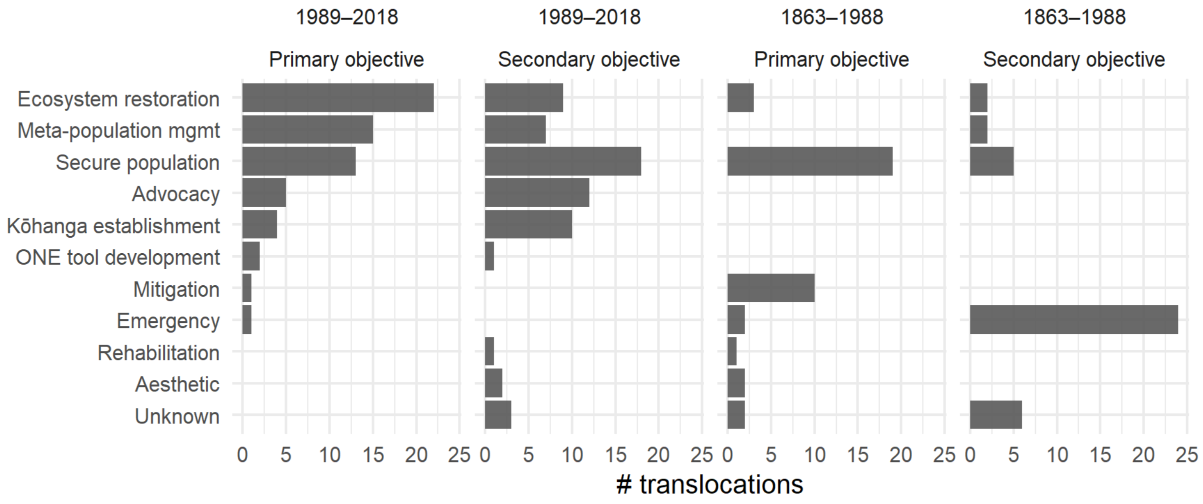

Figure 2.7 Primary and secondary objectives for all kiwi translocations between 1863–2018. The displayed split between the periods highlights the substantial shift of dominant objectives in the last three decades. Only overall translocation objectives were listed; some releases within a translocation project were carried out for multiple objectives.

2.3.7 Translocation lead and reporting

Translocations have been initiated and led by multiple entities. The Department of Conservation (DOC) has played the dominant role in leading kiwi translocations since it was formed in 1987. In the last two decades, DOC or regional councils led 38% of translocation projects. In contrast, community groups and iwi/hapū led 25% and 3% of the projects, respectively. The remaining 33% were joint projects between DOC and either community groups or iwi/hapū. The information about earlier translocations is incomplete, but records suggest that pre-1987 translocations were also led by a variety of entities: The New Zealand Wildlife Service (a predecessor of DOC), community groups, reserve caretakers, and private initiatives.

Specific kiwi translocation projects are rarely reported in the scientific literature. Out of 76 translocations between 1979–2018, reports from only nine projects (12%) appear in scientific journals: four little spotted kiwi translocations (Jolly & Colbourne 1991; Colbourne & Robertson 1997), four brown kiwi projects (MacMillan 1990; McLennan & Potter 1992; Smuts-Kennedy & Parker 2013), and one great spotted kiwi project (Toy & Toy 2020). Information on a further 47 projects (62%)

exists in reports, some published and publicly available, mainly through the DOC website, but most project reports remain solely as internal documents. For 20 projects (26%), reports either do not exist or were unavailable. Information about these projects is usually kept only as internal records. Recently, information about some projects has appeared on social media or in news reports. However, the focus of such information is mainly on release events, rather than long-term monitoring, or summarising lessons learnt.

2.3.8 Translocation outcome

We assessed translocation outcomes of 75 out of 102 translocations. The remaining 27 projects were not assessed as they were either reinforcements (24) or introductions resulting in hybrid populations (3). Out of the 75 assessed introductions or reintroductions, only 19 are known to have released at least 40 birds by 2018. However, only four projects had released at least 40 birds by 2003, so that at least 15 years had passed before the assessment date. Three of these failed and only one site, Zealandia, had a population increasing at a rate of more than 2% annually on average and therefore can be considered as a likely successful introduction/reintroduction as of 2018 (Fig. 2.8).

The sensitivity analysis categorising translocations revealed a similar pattern. Only 1–3 translocations were likely successful, based on different criteria values. When we reduced the assessment timeframe to ten years, three more sites received 40+ birds from 2004 to 2008. However, only one, Tāwharanui, could be considered likely successful due to its sustained population growth at this stage. When we increased the assessment timeframe to 20 years, no translocation would fit into the likely successful category. To identify translocations that were close to achieving a sufficiently genetically diverse founder population, we reduced the threshold number of released birds from 40 to 35. Only one project site received between 35–39 birds by 2003 and none in the period 2004–2008. Taranga (Hen) Island, received 38 birds and has shown a more than 2% average annual population increase so would fit into the likely successful category with these less restrictive criteria. When we increased the minimum number of released birds to 50, only two project sites, Resolution Island (LSK) and the Rangitoto Range, received 50+ birds by 2008. Nevertheless, both these introductions failed.

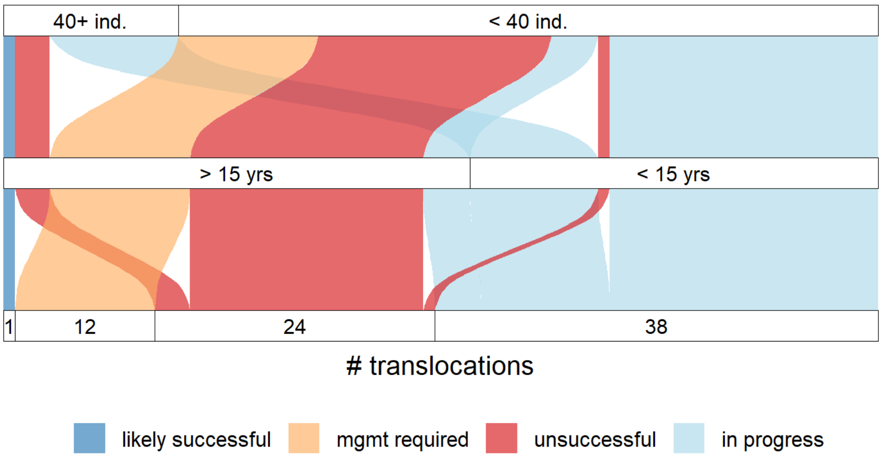

Figure 2.8 Outcomes of introductions and reintroductions (*n* = 75) between 1863–2018. Most recent kiwi introductions and reintroductions are still in progress, and most previous projects (before 2003) are either unsuccessful or require further management. Each colour represents an outcome category: likely successful, requiring further management, unsuccessful, and in progress. The top horizontal bar separates the translocations between projects releasing the recommended 40+ individuals and projects that have released less than 40 birds. The middle bar separates the translocations which were either finished or had released 40+ individuals more than 15 years ago (by 2003) and projects that are either ongoing or released 40+ individuals after 2003. The bottom bar shows the totals in each outcome category. Three introductions resulting in extant hybrid populations were not assessed, nor were 24 reinforcement translocations.

Most assessed translocations (56 out of 75; 74.7%) released fewer than 40 individuals by 2018 and are therefore either classified as requiring further management, unsuccessful, or in progress. Here, we included eight projects where the number of transferred birds is not available or is uncertain, but is likely less than 40 individuals. We classified 12 established translocated populations with less than 40 released birds as requiring further management. Among these projects is little spotted kiwi on Kapiti Island, where it is unlikely that more birds will be added as there is no known wild population available for harvest. Also, most in progress projects (23 out of 38; 60.5%) released fewer than 40 birds by 2018. The remaining

15 in progress projects released 40+ birds after 2003, less than 15 years before our assessment was made. Previously translocated populations, which recently received additional birds by reinforcement, were also classified in progress, such as Motukawanui, Tiritiri Matangi, Red Mercury, and Ulva islands (Appendix 1).

Failed introduction or reintroduction projects, in which birds were subsequently removed, or where the population did not persist, were categorised as unsuccessful. In these instances, some individuals may still survive in the project area, but the population is considered functionally extinct. One third of the assessed translocations were unsuccessful projects (24 out of 75; 32%), but only three of these occurred in the last two decades. These recent unsuccessful attempts were projects where the translocated birds were removed following fatalities due to either starvation or predator incursion. The remaining 21 unsuccessful translocations were mostly early island releases from the late 19th/early 20th century or mitigation/emergency translocations from the 1970s and 1980s. Likely causes of failure in the first group were a combination of small founder populations, predator invasions, and insufficient habitat size (Colbourne 2005; Frankham et al. 2017). Failed translocations in the 1970s and 1980s to unfenced mainland sites suffered mainly from the lack of predator control, dispersal, and/or insufficient initial population size (MacMillan 1990; McLennan et al. 1996; Colbourne & Robertson 2000).

2.3.9 Monitoring of population establishment

A lack of detailed post-release monitoring data severely limited the evaluation of post-translocation survival and dispersal. No such data were collected, or data were unavailable, for kiwi translocations prior to 1979. Between 1979–2018, post-translocation monitoring information was available for 75% of all translocations (n = 76). Information from monitoring databases and survey reports was supplemented by reports on incidental deaths or sightings of released animals. Fewer projects had available information about survival longer than one-year post-release. Specifically, survival information was available for at least a subset of released birds for one full year (70% of projects), three years (46%), and five years (30%) post-release.

In the last two decades (1999–2018), radio telemetry was the primary type of post-release monitoring (72% of all projects, n = 60). Call counts and other acoustic monitoring were the second most common (13%), followed by recapture surveys (8%). Information for the remaining 7% of the projects was unavailable, or no monitoring was carried out. Many projects, particularly the longer running ones, used various types of monitoring methods at different stages of the project or for different releases. In many cases, radio telemetry was replaced with acoustic recorders or call count surveys. Infrequent recapture surveys with trained dogs or playback calls tended to be used mostly on islands where other types of monitoring were deemed impractical.

2.3.10 Post-release effects – mortality and dispersal

Post-translocation survival in the release area was affected by a combination of mortality and dispersal of released birds. Between 1999–2018, 47 out of 60 (78.3%) translocation projects collected some information about post-release survival and mortality, including its causes. Out of the 47 projects, 72% recorded dead birds among successfully released individuals (for all but one project, these deaths were within three years post-release). Despite the effort by managers to identify the causes of death, 45% of projects with survival/mortality information (n = 47) reported kiwi mortality of unknown or uncertain causes (Table 2.1). Misadventure, as a cause of death, particularly by drowning and falls, was recorded at 40% of the projects. Although misadventure and unknown causes of death appeared at all site types, rates of predation varied widely between unfenced mainland sites on the one hand, and islands and fenced sanctuaries on the other. Sixty-five per cent of mainland projects (15 out of 23 projects with information on mortality) recorded predation of released kiwi. The birds were mainly depredated by mustelids – predominantly stoats, but also ferrets, which were confirmed at 30% (7) of mainland sites. Predation by dogs was also an issue at 30% (7) of sites.

In contrast, no island translocations reported predation of released kiwi between 1999–2018. However, stoat incursions occur repeatedly on islands near the mainland, such as in the case of Pomona Island at Lake Manapouri. In one case in 1982, previously released little spotted kiwi were removed from Maud Island to avert predation by newly-invading stoats (Colbourne 2005). The situation was similar at fenced sanctuaries. Only one fenced sanctuary, which did not eradicate stoats, reported stoat predation. Two other fenced sanctuaries reported rare stoat incursions but no associated kiwi mortality.

Table 2.1 Reported causes of kiwi mortality in translocation projects between 1999–2018. The table shows the numbers of projects where different causes of mortality occurred irrespective of their frequency, monitoring length, or effort. Predation was reported as the main cause of death at unfenced mainland sites despite predator control measures. We included reported incidents up to 12 years post-release to capture rare events, such as ferret or dog predation, affecting the long-term outcome of translocations. Misadventure includes injuries, falls, and drownings.

reported cause of mortality	**mainland**	**fenced**	**island**	**all sites**
predation all	**15**	**1**		**16**
by mustelids (stoat, ferret, weasel)	**13**	**1**		**14**
by ferret	**7**			**7**
by dog	**7**			**7**
by cat	**1**			**1**
car strike	3			**3**
misadventure	10	5	4	**19**
malnutrition	3	1	3	**7**
disease	2	4	1	**7**
transmitter entanglement	1	1	2	**4**
unknown	13	3	5	**21**
no mortality	4	1	8	**13**
projects with information	**23**	**8**	**16**	**47**
projects without information	8	1	4	13
all projects	31	9	20	60

Dispersal of released birds beyond the project area was a significant issue influencing population establishment, particularly at the unfenced mainland sites. There were no reports of kiwi dispersing from islands, and there was only one known report of a kiwi dispersing from a peninsula-fenced sanctuary, likely through a gap between the beach end of the fence and the waterline. Dispersal from protected project areas appears to be a frequent issue at unfenced mainland sites. Between 1999–2018, 64% of unfenced mainland projects (*n* = 22) with available information reported dispersal of at least some individuals to outside the project area within the first-year post-release. Sites with an area less than 3000 ha (median area of mainland sites) had one dispersed bird per three known surviving birds by the end of the first-year post-release. In contrast, sites of 3,000+ ha had one dispersed bird per 24 known surviving birds. At most mainland projects with reported post-release dispersal (at least eight out of 11), some or all wandering kiwi were brought back to the project area by managers. Several of the dispersing birds were brought back repeatedly.

2.3.11 Survival Model

The model estimating survival time of translocated kiwi showed considerable uncertainty in the model estimates due to the lack of precision in survival information and the small sample size within categories. Nevertheless, there were some clear patterns in the data (Fig. 2.9), such as evidence (posterior probability 0.92) for longer survival of birds released into predator-free sites compared to large sites (3000+ ha) with sustained predator control. We also found evidence for longer survival times of birds in predator-free sites compared to small sites (< 3000 ha) with predator control and unmanaged sites (both posterior probabilities > 0.99). Note, however, that the survival data from unmanaged sites came entirely from earlier translocations of only wild birds with no releases of ONE or captive birds. We found substantial evidence (posterior probability 0.82) for longer mean survival times in large sites than small sites with ongoing predator control.

Additionally, the post-release survival of wild and captive birds was higher than that of ONE birds (both posterior probabilities > 0.99) but there was no difference between captive and wild-caught birds. The expected annual survival rates of the released birds are shown in Table 2.2. There was no difference in mean survival times of birds released as part of reinforcement translocations compared to introductions and reintroductions. Similarly, there were no substantial differences among the mean survival times of various kiwi taxa. Nevertheless, we found substantial differences among individual translocation projects, which indicate the existence of unmeasured factors at the project level that may explain variances in estimated mean survival time. These factors may include the quality and type of predator control, the abundance and suite of predators in the project area and the surrounding landscape, the age of released birds, or other factors not included in our model.

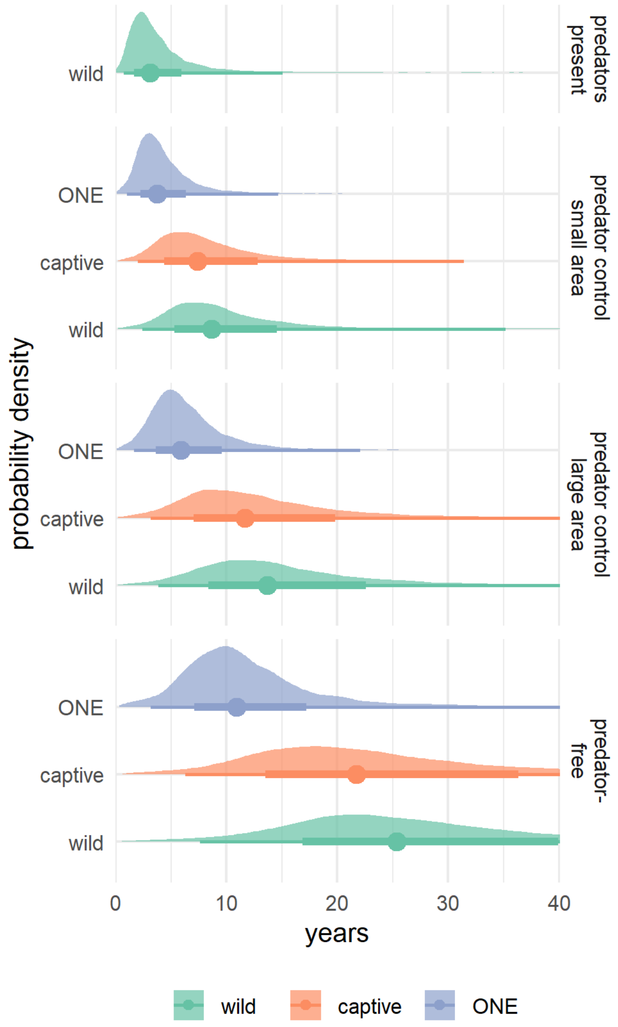

Figure 2.9 Density distributions of the expected mean survival times of translocated birds for different types of predator status of the project area. The dot represents the median and error bars display 66% and 95% credible intervals. Predator-free sites where wild-caught or captive birds were released exhibit the longest expected survival times, whereas sites where predators were present and uncontrolled show the shortest survival times. The defining line between small and large areas with predator control was size 3000 ha – the median area of mainland unfenced sites. Different colours represent different bird source types: wild-caught birds, captive facilities, and birds from the Operation Nest Egg (ONE) programme. Only wild-caught birds are shown for unmanaged sites with predators present as there were no recorded releases of captive or ONE birds to such sites.

Table 2.2 Expected annual survival rates (%) and expected median survival times for different source types of translocated birds – wild-caught birds, captive facilities, birds from the Operation Nest Egg (ONE) programme – and different types of predator status of the project area (see Fig. 2.9 for more detail). Values represent a median survival rate (%) with 95% credible intervals in brackets assuming constant hazard over time. A value of zero was assigned when it was not possible to calculate the annual survival rate due to an expected survival time of < 1 year.

source type	predator-free	predator control large area	predator control small area	predators present
wild	96 (87–99) *25.3 years*	93 (74–98) *13.6 years*	88 (59–97) *8.7 years*	68 (0–93) *3.1 years*
captive	95 (84–99) *21.7 years*	91 (68–98) *11.7 years*	86 (50–97) *7.4 years*	—
ONE	91 (68–98) *10.9 years*	83 (39–95) *5.8 years*	73 (1–93) *3.7 years*	—

2.3.12 Breeding of the released birds

Information on breeding by released birds within the dataset was scarce. Between 1999–2018, only 33% of all translocation projects (*n* = 60) reported breeding of the released birds within five years post-release. This information is mostly limited to signs of breeding attempts or successful hatching of chicks, and information about offspring recruitment into the breeding population is usually unavailable. During the first-year post-release, 75% of the projects (15 out of 20) reported breeding attempts among some of the released birds. This rose to 95% of the projects (19 out of 20) reporting breeding birds within three years post-release, which was mostly attributed to ONE birds reaching sexual maturity. During the first-year post-release, only 6% of reported breeders (*n* = 112) were ONE birds, while the share of wild and captive birds was 79% and 15%, respectively.

2.4 Discussion

We reviewed kiwi translocations since the 19th century, summarised information on translocated birds and translocation projects and evaluated their outcomes. Even though only a small number of translocations can be considered likely successful in establishing a self-sustaining population as of 2018, and for many translocations it is too early to make an assessment, we identified factors affecting the survival of released birds, which contribute to the translocation outcome.

The dataset of kiwi translocation projects we reviewed is more extensive than in previously published reviews and summaries (Atkinson 1990; McHalick 1998; Colbourne & Robertson 2000; Colbourne 2005; Cromarty & Alderson 2013; Miskelly & Powlesland 2013; Seddon et al. 2015). We are confident that the dataset contains an overwhelming majority of all kiwi translocation projects. There were uncertainties about several historical translocations, particularly to Kawau, Kapiti, Horomamae/Owen, and Motukiekie islands (Jolly & Daugherty 2002; Colbourne & Robertson 2004; Colbourne 2005), but despite the lack of or conflicting information about the exact year of releases or numbers of translocated birds, they appeared sufficiently credible to be included.

We encountered uncertainties about the historical ranges of kiwi taxa and the presence or absence of conspecifics at the time of the release, which affected the categorisation of projects as introduction (including assisted colonisation and ecological replacement), reintroduction, and reinforcement (Seddon 2010). Managers often classified kiwi translocations as reintroductions, but we re-interpreted some as assisted colonisations or ecological replacements of previously extinct taxa, based on former distributions of kiwi (Shepherd et al. 2012; Weir et al. 2016; Germano et al. 2018). We assumed that most kiwi translocations to islands were introductions, similar to Seddon et al. (2015), although kiwi naturally occurred on several large islands in the proximity of the mainland, such as D'Urville (Jolly & Daugherty 2002), Secretary, Resolution, and presumably Cooper Island (Henry 1895; Colbourne 2005) or Te Hauturu-o-Toi/Little Barrier Island (Palma 1991). Hence, historical translocations of tokoeka to Resolution, little spotted kiwi to Cooper Island, and brown kiwi to Te Hauturu-o-Toi were considered reinforcements.

2.4.1 Translocation effort and conservation status

Over the last four decades, birds from the North Island taxa (brown and little spotted kiwi) were disproportionately more frequently translocated compared to the South and Stewart Island taxa with large overall population sizes. Only the rarest taxa from the South Island, rowi and Haast tokoeka, had markedly higher proportions of translocated birds compared to their populations due to intensive conservation management and small population size (Fig. 2.4). A similar pattern in general kiwi management was identified by Innes et al. (2015b). In their assessment, great spotted kiwi, Fiordland and Rakiura tokoeka receive the least conservation management, and their overall populations are declining. In contrast, the North Island taxa, rowi, and Haast tokoeka have higher proportions of the population under some management regime and are generally increasing.

The kiwi taxa that receive relatively little management, including only a few or no translocations, have moved to more threatened conservation status in recent years, whereas those with more management, including more translocations, generally improved their conservation status. Brown kiwi is no longer considered threatened as its status improved between 2012–2016 to at risk/declining (Robertson et al. 2017). Similarly, little spotted kiwi, which has been repeatedly listed nationally as at risk/recovering, improved its conservation status on the IUCN Red List from vulnerable to near threatened category between 2004–2008 (BirdLife International 2016). The status of rowi improved from nationally critical to nationally vulnerable between 2012–2016, whereas Haast tokoeka remained nationally critical due to very low numbers and ongoing recruitment failure. In contrast, the status of great spotted kiwi and North Fiordland tokoeka worsened from gradual decline in 2005 to nationally vulnerable in 2008 and have remained at that level (Hitchmough et al. 2007; Miskelly et al. 2008; Robertson et al. 2017). Rakiura and South Fiordland tokoeka were considered nationally vulnerable, but both were reclassified as nationally endangered between 2008–2016 (Robertson et al. 2013; Robertson et al. 2017).

Even though a higher translocation effort may indicate an improvement in the conservation status of kiwi taxa, it is likely related to their respective population sizes. For example, the rare kiwi taxa would not survive or be recovering without translocations (Seddon et al. 2015). Little spotted kiwi currently persist only in translocated populations and the major range expansions of little spotted kiwi, Haast tokoeka, and rowi can be attributed solely to management by translocations. In contrast, the population increase of brown kiwi taxa is mainly attributed to improved and expanded *in situ* management through predator control, while translocations played only a minor role in the species' recovery (Innes et al. 2015b). Similarly, the most populous South and Stewart Island kiwi taxa do not appear to be declining because of the low translocation effort, but because of the lack of landscape-scale predator control across most of their current range (Innes et al. 2016). Therefore, the use of translocations in reversing population decline and improving conservation status appears to have a substantial impact only in the rarest kiwi taxa: Haast tokoeka, rowi, and little spotted kiwi.

2.4.2 Translocation objectives

Establishing secure populations was a common translocation objective for most kiwi taxa over the last two decades. Similarly, translocations carried out as part of meta-population management or for establishing kōhanga/source populations for future translocations were relatively common in little spotted kiwi, rowi, and Haast tokoeka – the three most management-dependent taxa with the smallest populations. Given the improving conservation status of brown kiwi and little spotted kiwi, these taxa have been translocated in the last 20 years for a wider set of objectives, such as ecosystem restoration and conservation advocacy (Innes et al. 2015a; Nally & Adams 2015). These translocations have not been driven primarily by species-oriented conservation but instead by attempts to restore native ecological communities through pest eradication or suppression, alongside reintroductions of a broad suite of native species (Saunders & Norton 2001; Miskelly 2009). In some instances, advocacy and public engagement were the main objectives for translocations, which may create a possible conflict of interest between species recovery and public demand (Nally & Adams 2015).

2.4.3 The outcome of introduction and reintroductions

Improving outcomes of translocations is an important element of kiwi recovery management (Germano et al. 2018) as well as of continued public support for this conservation tool (Jachowski et al. 2016). The number of populations initiated through translocations is increasing, which contributes to one of the main kiwi recovery goals: restoring former distributions of all kiwi taxa. Translocations also expand beyond the former distributions, by creating new populations on islands, or through ecological replacements of kiwi taxa that are now extinct. However, it is not immediately evident if these newly established populations will grow and maintain sufficient genetic diversity, and therefore if they contribute to the other two kiwi recovery goals (growing populations of all taxa and maintaining their genetic diversity), which are crucial for population persistence (Converse & Armstrong 2016; Nelson et al. 2019).

Kiwi translocations have been reported as successful in internal transfer reports and scientific literature, usually based only on initial survival of released birds and/or signs of breeding (Colbourne & Robertson 1997; Miskelly & Powlesland 2013; Smuts-Kennedy & Parker 2013) unless there was an apparent failure of kiwi to establish at the release area (MacMillan 1990; McLennan & Potter 1992). However, this perceived success does not necessarily lead to a self-sustaining population (Wolf et al. 1998; Brichieri-Colombi & Moehrenschlager 2016). Also, those assessments generally did not consider the genetic make-up of newly-established populations and the consequences of small founder numbers, which have been highlighted only in the last two decades (Briskie & Mackintosh 2004; Groombridge et al. 2012; Jamieson & Lacy 2012; Taylor et al. 2017).

Based on our categorisation, kiwi translocations appear to have a lower-than-expected success rate. Only one out of 75 introductions and reintroductions can be considered likely successful at the 15-year mark with the *caveat* that this population was, unavoidably, sourced from a previously severely bottlenecked population, which may have negative implications in the future (Ramstad et al. 2013). The number of likely successful translocations increases to two if we reduce the in progress period from 15 to 10 years, and three if we also reduce the required number of released birds from 40 to 35. In contrast, one third of all assessed projects (24) were unsuccessful. However, nearly half of these projects failed more than 100 years ago and only three failed in the last two decades. Half of all assessed projects (38) were still in progress, and we categorised the remaining 12 translocations as requiring further management. All translocations in this group were previously considered successful due to achievement of population establishment and growth (Colbourne & Robertson 1997; Miskelly & Powlesland 2013). However, several of these populations were established with a small number of individuals (5–38, two projects unknown), and it has become apparent that they may suffer from inbreeding depression despite successful establishment and initially strong population growth (Taylor et al. 2017). Several of these populations have been recently supplemented by reinforcement translocations to improve their genetic diversity (Robertson et al. 2019a, 2019c) and therefore, we considered them as projects in progress. These meta-population management efforts appear to be an exception for rare (little spotted kiwi) or endangered taxa (Rakiura tokoeka). The need for meta-population management in little spotted kiwi is amplified by the fact that the entire known population of the species likely descends only from a few birds translocated to Kapiti Island more than a century ago and most recent populations were established with less than 40 individuals (Taylor et al. 2017). Most management plans for other kiwi taxa do not explicitly address the issue of the genetic health of previously established populations with low founder numbers.

Nearly half of all kiwi translocations (50) commenced, and 76% of all translocated birds were released after 2003. Thirty-eight per cent of these projects were reinforcements, which were not assessed similarly to introductions and reintroductions. Reinforcements were carried out for a large variety of objectives, from mitigation/emergency transfers to meta-population management, and the numbers of released birds varied widely (1–100+) and are not directly comparable between each other (Fischer & Lindenmayer 2000). Fifty-eight per cent of the translocations started after 2003 were introductions and reintroductions that we categorised as in progress; only one project has definitely failed, because of ferret predation. Several other introductions and reintroductions that are in progress suffer from population declines due to high mortality and dispersal and/or lack clear plans to release the recommended 40 individuals for a genetically diverse and self-sustainable population; these projects are on a trajectory towards the requiring further management or unsuccessful categories after the 15-year benchmark we used for assessment. In contrast, some translocations with higher numbers of released birds to pest-free sites or larger sites with intensive predator control already show signs of positive population growth, the essential precursor to long-term translocation success (Armstrong & Reynolds 2012).

2.4.4 Population establishment

An overall translocation outcome depends on multiple factors, but a prerequisite of a successful translocation project is the establishment of the population at the release area. Despite identifying only one translocation, or possibly up to three, that are considered likely successful at the time of our assessment, we attempted to determine factors that contribute to translocation success in the future across all translocation types. In particular, we focused on the post-release effects and factors affecting post-translocation survival at the release site, which determine the prospects of population establishment (Tavecchia et al. 2009).

Survival modelling clearly shows differences in mean survival time of translocated birds based on the source type of released birds as well as predator status at the release site, despite wide credible intervals caused by the lack of long-term monitoring data. Information on the exact time of death or dispersal for more of the released kiwi would have allowed more robust predictions. Wild-caught birds and birds from captivity appear to survive for much longer than birds from the ONE programme. Kiwi from captive facilities showed similar mean survival times to wild birds, which is surprising in light of reviews identifying translocations of captive animals as less successful for many species (Griffith et al. 1989; Fischer & Lindenmayer 2000). However, other studies (Miskelly & Powlesland 2013; Brichieri-Colombi & Moehrenschlager 2016) showed that releases of wild and captive animals have similar outcomes. Lower median survival rates of translocated ONE birds were consistent with lower survival rates of ONE subadults and juveniles released back to their source populations, compared to survival rates of wild adults of various kiwi taxa in those populations (Robertson et al. 2011; Robertson & de Monchy 2012). However, in those populations the wild adults were resident birds with established territories and therefore the situation is not directly comparable to translocations of both wild and ONE birds into unoccupied areas. The shorter mean survival time of ONE birds was likely caused by higher rates of mortality related to misadventure, but also a higher susceptibility to predation due to the young age of most ONE birds at the time of their release (McLennan et al. 1996; Robertson et al. 2011).

Mean survival times for birds released to predator-free environments were substantially higher than to areas with ongoing predator control and large areas performed better than smaller project sites. Predator-free sites were generally islands (Colbourne 2005; Bellingham et al. 2010) or fenced sanctuaries, where the main predators of kiwi were eradicated and where necessary measures against reinvasion have been in place (Burns et al. 2012; Innes et al. 2015a). Sites with ongoing or no predator control were mainly unfenced mainland sites with a range of kiwi predators at varying densities. Two-thirds of the project sites with ongoing predator control still reported issues with predation of the released birds, mostly by mustelids. Given many unknown causes of mortality, the real proportion of mainland translocation projects affected by predation is likely to be higher. The list of reported predators is similar to other studies on kiwi survival (McLennan et al. 1996; Basse et al. 1999; Innes et al. 2010).

Post-translocation dispersal is very likely to be another major factor affecting population establishment at unfenced mainland sites, as has been found in other studies (Brichieri-Colombi & Moehrenschlager 2016; Berger-Tal et al. 2019). While predator-free sites benefited from natural or artificial barriers limiting incursions of predators, these barriers also prevented kiwi from dispersing outside the project areas. Both the reduced risk of predation and reduced dispersal out of predator-free sites likely contributed to markedly higher survival times than at unfenced mainland sites. Smaller unfenced mainland areas were presumably more affected by dispersal than large sites, which allowed post-release dispersal of more birds to occur within the limits of the project areas.

2.4.5 Genetic considerations

The current official guidance for kiwi introductions and reintroductions is to release 40 individuals to establish a founding population (Sporle 2013; Robertson & Colbourne 2017). This recommendation is based on the modelling of retention of rare alleles for brown kiwi and estimates of the founder numbers required to preserve the genetic diversity essential for population viability (Weiser et al. 2013; Weiser 2014) and maintenance of its adaptive potential (Weeks et al. 2015). However, there are several other requirements that need to be met to maintain the desired level of genetic diversity, including 90% post-release survival, no dispersal, a high proportion of breeders among released individuals (> 90%), a predator-free environment, and a sufficient carrying capacity of the project area to allow for population growth. Furthermore, given the lower survival rates of translocated juvenile and subadult kiwi, more young birds should be released to retain the same level of allelic diversity within a new population than within a population established by releasing adults alone (Weiser 2014).

The reality of translocation often diverges from these requirements. At three predator-free sites, where populations were established by wild-caught and mainly adult little spotted kiwi, the estimated proportion of effective founders (released birds that successfully produced offspring) within the total numbers of translocated birds was highly variable (40–81%). Such low proportions of successful breeders suggest a significant loss of the source population's genetic diversity within the new populations (Ramstad et al. 2013), which is likely common among translocations of other kiwi taxa as well. Similarly, given the lower mean survival times at sites with predator control and where populations were established by ONE birds, we could infer that the number of released birds needs to be much higher than the suggested 40 birds to maintain the desired level of genetic diversity.

2.4.6 Recommendations

Advances in translocation practice, such as implementing a regulatory framework (Nally & Adams 2015) and providing official guidance (Robertson & Colbourne 2017), have helped to increase prospects of recent translocations. Still, the increasing popularity of kiwi translocations brings challenges relating not only to the sufficient size of founder populations but also in the continuation of habitat management, which is critical for the achievement of long-term recovery objectives (Grant et al. 2019). A successful translocation has been traditionally understood to be the establishment of a self-sustaining population (Griffith et al. 1989; Fischer & Lindenmayer 2000). However, long-term habitat management in terms of predator control (Stadtmann & Seddon 2020) and management of genetic diversity may be necessary for long-term persistence (Hayward & Slotow 2016). Cryptic inbreeding depression in isolated bottlenecked populations or ongoing predator incursions may pose significant threats to the persistence of translocated populations. Therefore, continued monitoring of translocated populations and adaptive management are crucial.

From the available data, it is apparent that each introduction/reintroduction project needs to plan for a specific number of released birds based on a combination of factors, such as predator status of the project area, its size, the age structure and source(s) of released birds, and dispersal probability. Managers need to account for the loss of genetic diversity caused by the differences in survival rates, causes of mortality, or the carrying capacity, and the boundary type of the project area (Department of Conservation 2018). Left unchecked, the genetic consequences of small founder size of translocated populations have the potential to reduce the fitness of birds, increase rates of hatching failure, and decrease the overall population viability (Briskie & Mackintosh 2004; Heber & Briskie 2010; Frankham et al. 2017). Such reduced viability of a newly established population has already been observed in little spotted kiwi on Long Island (Taylor et al. 2017), but it is likely more widespread (Ramstad et al. 2013). This issue underscores the importance of genetic monitoring and management to address the risk of genetic drift and inbreeding depression in translocated populations (Weeks et al. 2015; Biebach et al. 2016; Frankham et al. 2019). Long-term genetic monitoring will also enable assessment of the effectiveness of reinforcements, to identify whether the translocated birds successfully contribute to the gene pool of the resident wild population (Fischer & Lindenmayer 2000).

At mainland unfenced sites, sufficient space for post-release dispersal or natural dispersal of future juveniles and subadults needs to be available to reduce losses of released birds and their offspring. If no additional habitat is available to enlarge the project area, managers may need to follow up with periodic releases to compensate for the loss of dispersed individuals and facilitate necessary gene flow between populations (Brown et al. 2015; Richardson et al. 2015; Gitzen et al. 2016). Nearly all of the mainland unfenced sites where kiwi were established in the last two decades were smaller than the recommended 10 000 ha. Smaller areas may not provide sufficient habitat for natural juvenile dispersal and compromise population viability due to higher rates of emigration and predator incursions (Basse & McLennan 2003; Westbrooke 2007; Brown et al. 2015). Between 1999–2018, more than half of the unfenced mainland projects established a population at an area smaller than 1500 ha, which resulted in substantial rates of dispersal outside the project area. However, even when kiwi were released to larger areas, there were instances of release burrows being located near the boundaries of the project area, which diminished the buffer zone effect for dispersal. Currently proposed translocation guidelines recommend a project area size with a carrying capacity of at least 100 pairs to accommodate population growth and enable maintenance of genetic diversity. This approach acknowledges variability in kiwi densities for different taxa and different habitats (Department of Conservation 2018), but the habitat availability for post-release and juvenile dispersal needs to be considered as well.

Similarly, habitat quality, including predator status, is a crucial factor for the outcome of translocations (Sheean et al. 2012; Brichieri-Colombi & Moehrenschlager 2016; Stadtmann & Seddon 2020). Small, unfenced areas may not provide sufficient habitat quality for population persistence even with intensive predator management (Brown et al. 2015). Extensive buffer zones with predator control around unfenced project sites may reduce predator incursions and associated kiwi mortality. Notably, control measures for wide-ranging ferrets in the areas surrounding project sites may be necessary to reduce deaths within translocated populations. Long-term habitat suitability and the propensity for stochastic events should also be considered when selecting sites for kiwi translocations. Instances of droughts on Tiritiri Matangi and Red Mercury islands, leading to kiwi mortality and the reduction of carrying capacity (Robertson et al. 2019a, 2019c), demonstrate the need for careful assessment of long-term viability of such kiwi populations at isolated sites with changeable habitat quality.

Translocated populations must be adequately monitored, and the translocation outcomes reported. Post-release effects play an important role in population establishment and should be addressed by managers to reduce mortality and dispersal, particularly of ONE birds. Appropriate release strategies, such as releasing ONE birds in larger groups and outside of winter and autumn months, may significantly increase survival, as demonstrated in rowi (Batson et al. 2015). However, the most suitable season for release may differ among kiwi taxa (Robertson & Colbourne 2017). Modelling of survival times suggests that releases of ONE birds are best suited for predator-free sites to maximise chances of these birds to settle, mature, and breed successfully. Sufficiently long and standardised post-release monitoring (Sutherland et al. 2010) will facilitate addressing possible issues, inform subsequent adaptive management, and allow for planning of additional releases, if needed, to supplement the founder population (McCarthy et al. 2012). Reporting of demographic data from monitoring, together with details on habitat variables and applied management tools, is essential to further increase the effectiveness of future translocations (Moro et al. 2015; Gitzen et al. 2016).

2.5 Conclusions

We reviewed 102 kiwi translocations from the 19th century until the present to assist with the refinement of the translocation process and guidelines (Batson et al. 2015), and to inform the upcoming kiwi translocation strategy (Germano et al. 2018). Translocations of kiwi have grown in the last two decades, both in numbers of translocated birds and the number of new translocation projects. There has been a marked increase of projects focusing on ecosystem restoration by releasing less threatened taxa of kiwi in the proximity of urban areas and a shift from the government towards community groups undertaking the translocations (Nally & Adams 2015). During the same period, establishment and reinforcement of kiwi populations on the mainland became the dominant type of kiwi translocations, and releases to fenced sanctuaries were pioneered. However, releases to predator-free islands increased as well, mainly in the last ten years.

Because kiwi are long-lived, most recent kiwi translocations are considered as in progress. However, our analysis shows that despite their growing popularity, and improvement in their planning and management, kiwi translocations face similar issues as translocation projects of other species worldwide (Berger-Tal et al. 2019). Unfenced mainland project sites suffer from predation, which highlights the common overestimation of the efficacy of predator control measures and overall habitat suitability in translocation proposals, particularly at smaller sites, which are affected the most. An overestimation of habitat suitability also occurs at predator-free sites. Notably, islands may suffer from droughts causing a reduction in habitat carrying capacity and increased risk of translocation failure due to small population size (Robert et al. 2015). Behavioural issues, such as post-release dispersal and learning incidents, exacerbate adverse post-release effects, particularly at unfenced mainland sites and in releases of subadult and juvenile ONE birds. Such issues cause substantial differences in mean survival times and thus differences in the probability of translocation failure among various projects, based on the source type of the released birds, predator status, and size of the project area.

In our framework for the evaluation of introduction/reintroduction outcomes, we focused on the recommended number of released birds as one of the metrics for likely translocation success. Current guidance is to release 40 unrelated starters to become the founders of a new population (Robertson & Colbourne 2017). However, differences in expected survival times and time until breeding occurs revealed the need for project-specific adjustments of the number of released birds to achieve similar targets for the retention of genetic diversity. For projects where higher rates of mortality, dispersal, and delayed breeding is expected, substantially more released birds will be required. Consequently, past or recent translocations with lower numbers of effective founders will likely need further releases or periodic meta-population transfers to achieve genetically robust and thriving populations (Weeks et al. 2011) that contribute to the kiwi recovery goals.

Adaptive management based on appropriate demographic and genetic monitoring will facilitate the long-term persistence of the translocated populations. Information stemming from such management and monitoring will further inform best practice for kiwi translocations and broader applications of reintroduction biology.

2.6 Acknowledgements

We are thankful to all data providers and people who collected the data in the field: members of community groups, tangata whenua, and staff of the Department of Conservation. We also thank the Kiwi Recovery Group members, particularly Jess Scrimgeour, Hugh Robertson, Rogan Colbourne, and Isabel Castro, for their input and feedback, and to Kiwis for Kiwi and the Zoo and Aquarium Association for the assistance with obtaining and interpreting the data. We are thankful to two anonymous reviewers for their insightful comments to improve the manuscript. PJ was supported by a Lincoln University Doctoral Scholarship.

Chapter 3

Post-translocation movements and ranging behaviour of roroa–great spotted kiwi

3.1 Introduction

Kiwi (*Apteryx* spp.) are flightless ground-dwelling birds endemic to the three main islands of New Zealand, and most of them are threatened with extinction (Robertson et al. 2021). Roroa (great spotted kiwi, *A. maxima*, previously known as *A. haastii*, Shepherd et al. 2021) is native to the north-western part of the South Island, New Zealand, with a range currently separated into four known subpopulations: i) Arthur's Pass, ii) Paparoa Range, iii) Westport, and iv) the north-west Nelson region. Roroa population size is estimated to have decreased from approximately 16 000 to 14 000 individuals during 2008–2018, and is likely still declining (Holzapfel et al. 2008; Germano et al. 2018). Until recently, much of the extant roroa population received little or no regular management for invasive predators, which are considered the primary driver of population decline (Innes et al. 2015b). Consequently, roroa's conservation status is 'Nationally Vulnerable' (Robertson et al. 2021).

Several management actions have been adopted to address the ongoing kiwi population decline and reduce the threat of extinction. Kiwi conservation management focuses mostly on either suppression or elimination of invasive predators, i.e. mustelids (Mustelidae), common brushtail possum (*Trichosurus vulpecula*), and feral cats (*Felis catus*), mainly through trapping and poisoning using aerial 1080 (sodium fluoroacetate); and advocacy and avoidance training to mitigate predation by dogs (*Canis familiaris*) (Robertson et al. 2011). Another management regime involves head-starting chicks under the Operation Nest Egg (ONE) programme. This approach consists of removing eggs from the wild, hatching them in captive facilities, and keeping the young kiwi in a predator-free environment until they reach a size at which they can fend off stoats (*Mustela erminea*), their main predator, before they are released back to the wild (Colbourne et al. 2005; Gillies & McClellan 2013). Subadults from ONE programmes are either returned to their source population or released elsewhere to establish or reinforce an existing kiwi population.

Kiwi translocations have become an increasingly popular tool in the conservation management of all kiwi species (Miskelly & Powlesland 2013; Jahn et al. 2022a). To date, translocations have played a relatively minor role in roroa management compared to the other kiwi species. Roroa transfers involve mainly wild-caught birds, whereas releases of ONE subadults dominate in North Island brown kiwi (*A. mantelli*), rowi (*A. rowi*), and tokoeka (*A. australis*) management (Jahn et al. 2022a). The first documented translocation of roroa was a 1915 release of 19 birds onto Te Hauturu-o-Toi/Little Barrier Island, but despite initial population establishment, this introduction failed, likely within 15 years post-release (Oliver 1955; Colbourne 2005). There were no other attempts to establish new populations until the 21st century. Wild-caught roroa from the north-west Nelson and Westport populations were reintroduced in 2004 to Lake Rotoiti, Nelson Lakes National Park, and in 2010 to the Flora Valley, Kahurangi National Park (Gasson 2005; Toy & Toy 2020). Following these initial efforts, ONE subadults from the Arthur's Pass population were reintroduced in 2011 to the Nina Valley. The Nina Valley is part in Lake Sumner Forest Park, Conservation Area Nina Doubtful Rivers, and Lewis Pass Scenic Reserve.

The Nina Valley reintroduction project was similar to the Rotoiti and Flora translocations focusing on ecosystem restoration and being driven by attempts to restore the former species distribution (Holzapfel et al. 2008; Hulsman et al. 2010; Morrison & Yong 2014). The project was initiated by the Hurunui College Nina Valley Restoration Group in co-operation with the Department of Conservation (DOC). During 2011–13, ten ONE subadults initially sourced as eggs taken from the Hawdon Valley, Arthur's Pass National Park, were released to the Nina Valley to re-establish a roroa population. Subsequently, eight wild-caught adults from the Hawdon Valley were translocated to the Nina Valley in April 2015 to expand the initial founder group. The birds were released at several sites in the central part of the Nina Valley, within the 1600 ha trapped area that stretches alongside the Nina River.

The Nina translocation was the first – and to-date only – roroa reintroduction within the Arthur's Pass population. Therefore, it was vital to monitor the birds' post-release behaviour, to inform the planning of future releases, and provide information for potential management interventions. Lessons on post-release dispersal and territory establishment were available from Lake Rotoiti (Gasson 2005) and intensive monitoring was underway in the Flora Valley (Toy & Toy 2020), but it was not clear if the same behaviours would occur in the genetically distinct Arthur's Pass population (Taylor et al. 2021). We intensively monitored the translocated population in the Nina Valley to understand the released birds' dispersal pattern and identify where and when they established home ranges. Based on these data, and monitoring data from the source population in the Hawdon Valley prior to this translocation, we were able to address the following research questions:

- What were the dispersal paths and distances moved of wild-caught adult roroa following the translocation?
- What were the changes in the home range size of adults before-and-after the translocation?
- Were there any changes to the ranging behaviour of the previously translocated ONE subadults following the release of wild-caught adults into the same general area?

3.2 Methods

3.2.1 Study areas

The translocation of roroa was carried out from the Hawdon Valley (42°57' S, 171°45' E), Arthur's Pass National Park, to the Nina Valley (42°28' S, 172°19' E) near Lake Sumner Forest Park. Both valleys are within the historical range of roroa (Taylor et al. 2021). They are 70 km apart, east of the main divide near Arthur's Pass and Lewis Pass, respectively, indicating similar climate characteristics. The floor of the Hawdon Valley lies at 570–780 m a.s.l. and is surrounded by mountain peaks 1400–1930 m a.s.l. The floor of the Nina Valley lies at 610–860 m a.s.l. and is surrounded by mountains 1500–1780 m a.s.l. River terraces and steep slopes in both valleys are covered by native montane beech forest until the bush line at about 1300 m. The dominant tree species are mountain beech (*Fuscospora cliffortioides*), silver beech (*Lophozonia menziesii*), with red beech (*F. fusca*) at lower altitudes (Read & O'Donnell 1987; Blakely et al. 2008).

3.2.2 Translocation and monitoring

Eight wild-caught birds, four males and four females were translocated to the Nina Valley in April 2015. These birds were part of a roroa monitoring programme in the Hawdon Valley for up to five years before the translocation. All were of unknown age but were confirmed to be breeding pairs by radio telemetry monitoring. The birds were captured and transported to the Nina Valley according to the best practice guidance (Morrison & Yong 2014). The pairs were placed in pre-determined release burrows 800–900 m apart (closer only if separated by the Nina River), outside of known roroa territories, to mimic natural territorial structure. One pair was placed together in one large burrow while three other pairs had males and females placed in separate nearby burrows to allow paired individuals to stay in close contact. The birds were kept in the burrows several hours to calm down until one hour after sunset when burrow entry points were unblocked, and birds were allowed to move freely. A similar approach had been previously adopted for the release of ten unpaired subadult ONE birds during 2011–13. The average age of these roroa at release was 1.1 years (range 0.9–1.3). They were released in January 2011 (2), February 2011 (3), February 2012 (3), and January 2013 (2). In these instances, the 2–3 subadult birds were placed together in one large release burrow.

After the 2015 translocation, we monitored all eight translocated wild-caught birds and four kiwi previously released as ONE subadults using ground-based radio telemetry. The remaining ONE birds were not monitored because they had either dropped their transmitters before 2015 (4), died due to misadventure soon after the release (1), or occupied remote areas of the Nina Valley (1), which prevented regular monitoring. However, we included location data for one unmonitored ONE bird that was incidentally captured and paired with a monitored ONE bird. All the monitored birds were fitted with leg-mount diagnostic transmitters designed for roroa (Sirtrack V2.0 GSK, <2% of the body weight, 142–174 MHz) before the 2015 translocation transfer, and then for up to two years following the release. The transmitters allowed us to locate each bird for health checks and transmitter changes, or to remotely triangulate birds' locations (Neill & Jansen 2014). To triangulate the birds, we recorded the bearing of the signal multiple times from several (>3) points to achieve ideally at least a 90° overall angle between the bearings (Kenward 2001). Subsequently, we estimated the locations of monitored kiwi from a series of intercepting bearings using triangulation software Locate 3.34 (Pacer Computing).

Monitoring intensity differed throughout the monitoring period. In the first week after the release of the wild-caught adults, we aimed to triangulate all the birds every day. In the following month, we attempted to triangulate the birds at least once a week, and subsequently, the frequency of checks decreased to once every two weeks. After five months post-release, we attempted to triangulate the birds at least once every 2–3 weeks and after 18 months every 4–6 weeks. Locations of the roroa in the Hawdon Valley were triangulated fortnightly during the three months before the translocation. Locations of the four ONE birds in the Nina Valley were also triangulated for three months before the introduction of additional birds. As site visits were generally multi-day trips, we attempted to triangulate the birds on each day, when practicable.

Both triangulation and close approach (homing) took place during the day to locate nocturnal kiwi at their daytime shelters. Daytime triangulation provided ample time for a single surveyor to obtain multiple bearings while a kiwi is stationary at its daytime shelter. This approach generally reduced large location error when attempting to triangulate a moving animal, compared to more accurate GPS tracking (Guthrie et al. 2011). To measure triangulation accuracy, we estimated the location error from a beacon test carried out by placing a transmitter underground at a known location in the birds' habitat and then triangulating it multiple times (Millspaugh & Marzluff 2001). We estimated the location error of triangulated location fixes at 42.0 m (±7.1 SE, n = 8) with the mean distance between the observer's location and the beacon 201 m (26.4 *SD*).

3.2.3 Data analysis

The home range and dispersal path estimations were based on the analysis of daytime location fixes, similarly to other roroa studies (Jahn et al. 2013; Toy & Toy 2020). Most of the location fixes used in the analysis (76%) were obtained through triangulation. Additionally, we supplemented the triangulation data with locations from kiwi recaptures and transmitter retrievals, done by DOC staff or contractors.

To estimate the dispersal path of the translocated birds, we constructed a smoothed line between the release site and the last known location for each bird by calculating a rolling average of up to nine consecutive location fixes. We chose to use nine fixes because this was the overall number of location fixes for the bird with the shortest duration of post-release monitoring. Additionally, we calculated the straight-line distance between the release site and the last known location for each bird to supplement the information on the dispersal path length. To identify the area most likely crossed by each bird during post-release dispersal, we analysed their utilisation distribution (UD) based on the movement path using R 4.0.3 (R Core Team 2020) and the package ‘move’ 4.0.6 (Kranstauber et al. 2020). To construct the UD, we used the dynamic Brownian bridge movement model suited for irregular sampling because it incorporates the Brownian motion variance, location fixes’ timestamps, and the location error (Kranstauber et al. 2012). We used the data collected after the 2015 translocation to estimate the dispersal path and the UD for all the translocated wild-caught adults and four ONE birds that were released during 2011–12 (none of the birds released in 2013 were actively monitored). Lastly, we tested whether the UD size of the translocated adults was larger than that of the resident ONE birds. We used a Mann–Whitney *U* test, and we repeated this method in the following tests.

To identify possible changes in the home range size resulting from the translocation, we compared the home ranges of the adults in the Hawdon Valley before the translocation and after the translocation in the Nina Valley. Given that several birds moved substantially in the first six months post-release, we excluded this period from the home range estimation. We used location (homing) data obtained from DOC from up to five years before the translocation (3.1 years on average) to supplement the triangulation data collected during the three months immediately before the transfer. The longer monitoring period before the translocation compensated for infrequent location fixes and was not expected to substantially increase home range estimates due to a high population density and stable territorial structure of roroa in the Hawdon Valley. Because the data had substantial time gaps, we did not use the dynamic Brownian bridge movement model due to a large uncertainty of the movement paths between the consecutive location fixes. Therefore, we constructed minimum convex polygons (MCP) to estimate home range sizes, similar to other roroa studies (Keye et al. 2011; Jahn et al. 2013; Toy & Toy 2020). We used the R package ‘splancs’ 2.1.42 (Rowlingson & Diggle 2021) to calculate the size of MCP based on all location fixes and ‘ggmap’ 3.0.0 (Kahle & Wickham 2013) to map both MCP and UD. To inspect if the home range of translocated birds had become stable or kept shifting, we carried out an incremental area analysis with the R package ‘adehabitatHR’ 0.4.19 (Calenge 2006). Subsequently, we tested whether the MCP home range size of the wild adults increased due to the translocation.

To assess possible impacts of the wild birds' translocation on the ranging behaviour of the previously released ONE birds, we examined their home ranges in the two years before-and-after the release of the wild adults. Three out of the four ONE birds were released to the Nina Valley in 2011, the fourth individual in 2012, so the two year pre-release period started after the birds had been in the Nina for 26 and 14 months respectively. We assumed that this was sufficient time for the ONE birds to settle and establish stable home ranges, despite their transitioning from subadult to adult life stages during the monitoring period (Colbourne et al. 2020). To investigate if the home ranges of the ONE birds shifted following the release of the wild adults, we carried out an overlap analysis of their MCPs using the R package 'splancs' 2.1.42 (Rowlingson & Diggle 2021). We included ONE birds' location fixes from 2 years pre- and 0.5–2 years post-translocation of the wild adults, including location data (homing) obtained from DOC. We excluded the six months period after the wild bird's translocation from the MCP comparison to focus on the long-term effects of the wild bird's introduction as the immediate effects were captured in the previous UD analysis. Additionally, we tested whether there was a difference in the MCP home range sizes between the two periods.

3.3 Results

3.3.1 Dispersal path and utilisation distribution

All but one of the eight released wild adults stayed in the Nina Valley during the post-translocation monitoring (Fig. 3.1). The only bird known to have left the valley (male 'wild 3') was still within approximately one kilometre of the release site two weeks post-release but could not be detected afterwards. Eight weeks later, it was found dead, hit by a car, more than 10 km from the previous last known location in the Nina Valley. Another bird (male 'wild 2') was not detected from 11 months post-release after being reliably found in a defined area for eight months. We could not detect the transmitter's signal despite repeated searches over several months within and outside the Nina Valley, including an aircraft telemetry search of the nearby valleys. However, we assumed that the bird likely survived and stayed, but its transmitter probably failed. This was based on repeated male calls recorded in its presumed territory 17–20 months post-release and nightly activity pattern indicating possible incubation by its mate, female 'wild 2' (unpubl. data). Attempts were made to recapture male ‘wild 2’ during transmitter checks of female ‘wild 2’, but no male roroa was found.

We aimed to monitor all the wild-caught adults for at least two years post-release but we achieved this with only three birds translocated in 2015 (Table 3.1). Two birds dropped their transmitters at approximately one year post-release and could not be found for transmitter re-attachment. The DOC staff and contractors could not recapture another bird for a transmitter change despite several attempts one year after the translocation, so we monitored it until the transmitter battery died 1.5 years post-release. The remaining two birds either dispersed and died or were not able to be detected due to likely transmitter failure, as mentioned above. In contrast, we managed to monitor all four ONE birds for the two years following the 2015 translocation.

Although the seven surviving wild birds appeared to settle within the project area, only two of them (pair ‘wild 1’) settled in the proximity of their release site and stayed there during the monitoring period. The length of their dispersal path was similar to the path length of three previously released ONE birds, that had been in the valley for more than four years at the time of the wild adults’ release (Table 3.1). The remaining five birds moved widely around the valley without a clear pattern. In most cases, the dispersal path changed direction several times before home ranges started to stabilise after approximately six months. Three out of the four translocated pairs separated during the first four months. However, two of them reunited within the six months post-release in new areas, after being in different parts of the valley (>2 km apart) between approximately 1–3.5 months and crossing the Nina River repeatedly. The last pair (‘wild 3’) parted within two weeks post-release, headed in nearly opposite directions (Fig. 3.1), and the male later died outside the Nina Valley.

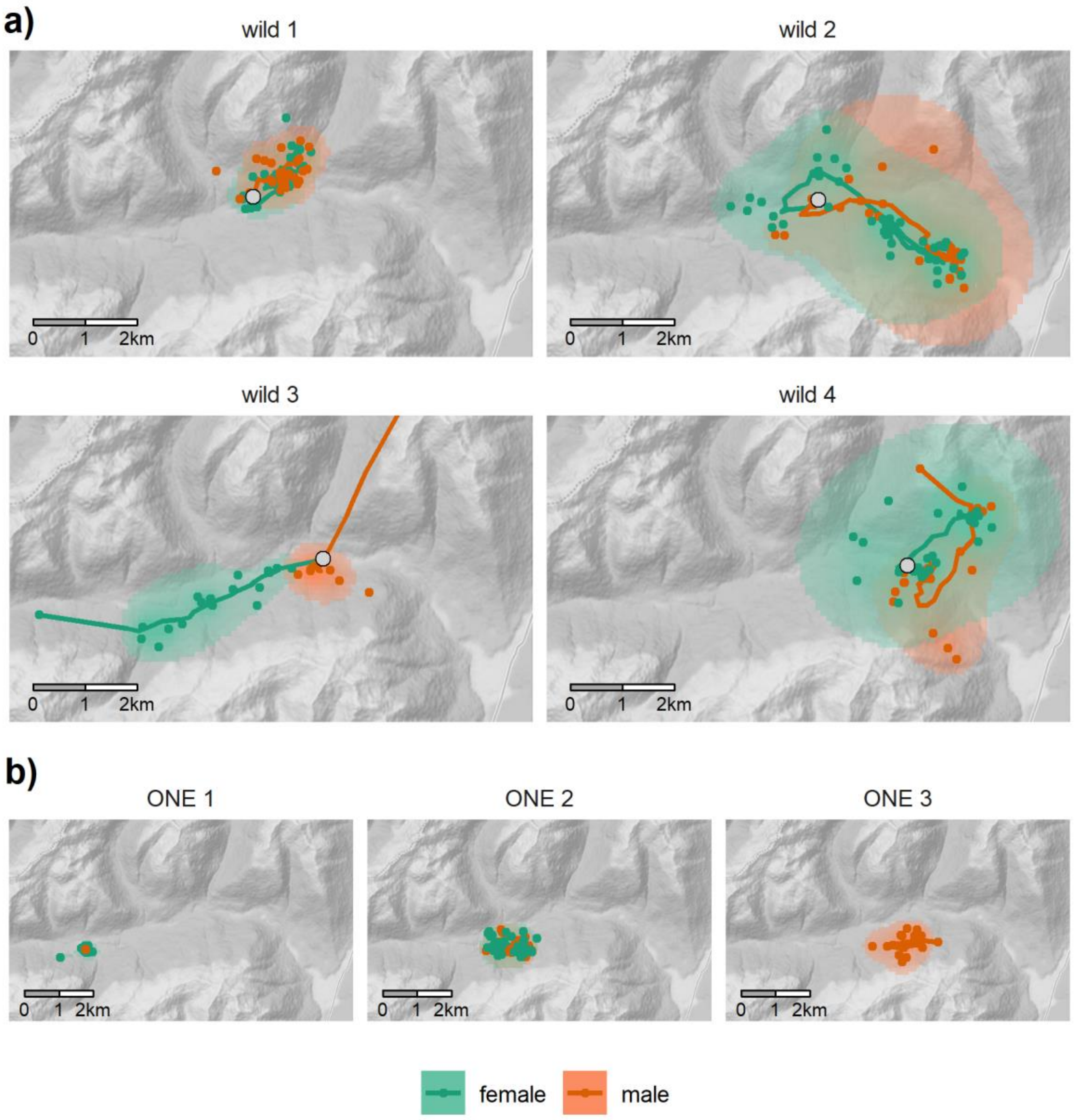

Figure 3.1 Utilisation distributions (UD) and dispersal paths of (a) wild-caught birds translocated in 2015 and (b) birds from the Operation Nest Egg (ONE) released during 2011–13 in the Nina Valley. The maps display release sites (grey circles) of the birds translocated on 16 April 2015 (pairs 'wild 2–4') and 23 April 2015 (pair 'wild 1'). The coloured dots show the location fixes of the birds following the 2015 translocation until May 2017. Solid lines connect each bird's first and last point during this monitoring period and represent the rolling average of up to nine consecutive location fixes. The coloured polygons display 95% UD for each bird except for male 'ONE 1', which was not actively monitored, so we did not have a sufficient number of location fixes for

the UD calculation. The last location of male 'wild 3' is not shown as it left the Nina Valley after at least two weeks and dispersed within ten weeks post-release 10.3 km north-west from the release site.

Table 3.1 Estimated dispersal path, distance, and size of utilisation distribution (UD) of the translocated wild-caught birds and previously released Operation Nest Egg (ONE) birds. The number of location fixes and monitoring length include only the period following the 2015 translocation of the wild-caught birds. Mean dispersal speed is based on the estimated dispersal path. The 75% and 95% UD represent an area where the individual would be located with the specified probability during the monitoring period.

pair	sex	dispersal path [m]	straight distance [m]	# location fixes	monitoring length [days]	dispersal speed [m/day]	75% UD [ha]	95% UD [ha]
wild 1	M	2,844	463	35	728	4	71	213
	F	2,985	1,079	39	746	4	91	204
wild 2	M	4,999	2,968	31	325	15	519	1,692
	F	7,046	1,660	44	736	10	433	1,459
wild 3	M	10,929	10,304	9	71	154	55	125
	F	5,824	5,552	20	346	17	141	420
wild 4	M	4,857	1,888	28	362	13	279	645
	F	3,614	1,514	33	554	7	555	1,653
ONE 1	F	629	62	15	718	1	9	36
ONE 2	M	2,951	475	40	749	4	56	172
	F	2,550	274	35	749	3	62	183
ONE 3	M	2,533	514	21	749	3	101	240

During the post-translocation monitoring period, the mean dispersal speed and the size of the utilisation distribution (UD) were highly variable among the released wild-caught birds (Table 3.1, Fig. 3.1). Both the core 75% UD and broader 95% UD were significantly larger among the newly released wild-caught birds compared to the resident ONE birds ($p = 0.036$, Mann–Whitney U test). The larger UD of the wild birds was consistent with their longer dispersal paths and straight-line distance between the first and last known locations, despite a 35% shorter average monitoring period compared to the ONE birds.

3.3.2 Home range size before and after the translocation

After six months post-release, the translocated wild birds appeared to be restricted to more defined areas, indicating stabilisation of their home ranges. The home range area (100% MCP, Fig. 3.2) kept incrementally increasing and appeared to reach an asymptote only in the three translocated wild birds that were monitored for the entire two years post-release. They had 22+ location fixes per bird in the period 0.5–2 years post-release. The home range of the other four surviving wild birds was still increasing at the end of their monitoring periods, which lasted between 0.9–1.5 years, resulting in a lower number of location fixes (<15). Similar to the three wild birds, home ranges of three out of four resident ONE birds reached an asymptote within the two year monitoring period. In contrast, the home range of the last bird (male 'ONE 3') continued to gradually increase even after two years.

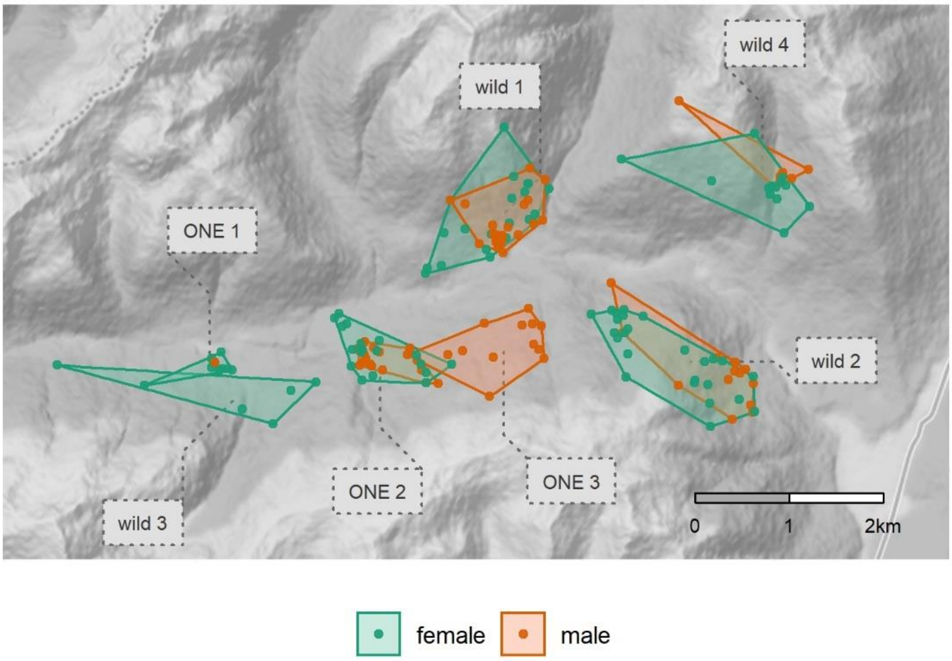

Figure 3.2 Minimum convex polygons (100% MCP) and location fixes of the monitored ONE and wild-caught birds in the Nina Valley after six months from the 2015 translocation (mid-October 2015 until May 2017). This monitoring period ranged among individual birds between 5–19 months. An MCP was not possible to construct for an unmonitored ONE male with a single recorded location.

The MCP home range size varied substantially among the monitored individuals (Fig. 3.3). The mean home range size of the translocated wild adults was 76.34 ha (±11.16 SE), which was significantly larger (p = 0.036, Mann–Whitney *U* test) than the home range size of the resident ONE birds at 37.31 ha (±13.93 SE). The mean home range size of the wild adults increased from that in the Hawdon Valley (54.39 ha ±5.13 SE), but this increase was not significant (p = 0.055, paired Mann–Whitney *U* test).

3.3.3 Home range stability of the previously released ONE birds

The ONE birds that were released 3–4 years before the 2015 translocation did not show any clear signs of changing their ranging behaviour following the release of wild adults. This was despite several of these translocated individuals moving through the ONE birds' territories (Fig. 3.1). Particularly, the ONE birds in known pairs ('ONE 1' and 'ONE 2') showed generally lower UD and MCP home ranges (Table 3.1, Fig. 3.3), indicating higher site fidelity. There was no major shift in the MCP home ranges of the resident ONE birds following the wild birds' translocation. Between the two monitoring periods, their MCPs had a mean overlap of 39.7% (±6.8 SE). Also, there was no significant difference in the MCP home range size of the ONE birds before-and-after the release of the wild adults, excluding the 6-months post-release period (p = 0.625, paired Mann–Whitney *U* test).

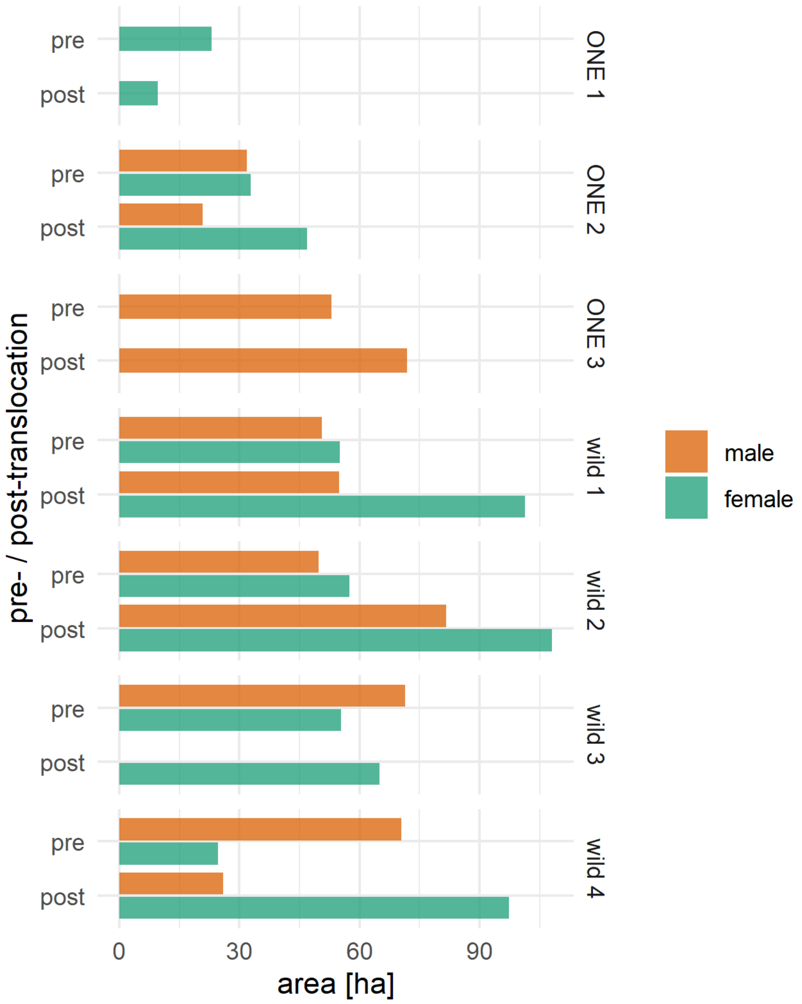

Figure 3.3 Comparison of 100% minimum convex polygon sizes representing home ranges of the monitored roroa before-and-after the 2015 translocation from the Hawdon Valley to the Nina Valley. The post-translocation period excludes the first six months post-release, in which the birds showed increased movements. The birds are grouped into pairs based on their origin – the Operation Nest Egg (ONE) birds released in the Nina Valley between 2011–13 and wild-caught birds translocated in 2015.

3.4 Discussion

3.4.1 Post-translocation dispersal

Dispersal of released animals plays a critical role in translocation outcomes (Richardson et al. 2015) and is often reported as one of the main issues encountered by various translocation projects (Brichieri-Colombi & Moehrenschlager 2016; Berger-Tal et al. 2019). Kiwi translocations to unfenced mainland sites also contend with dispersal outside the project area, particularly from small reserves under 3000 ha, although this issue occurs in reserves of any area size (Jahn et al. 2022a). Indeed, post-release dispersal appeared to be one of the main factors contributing to the failure of several previously reintroduced kiwi populations (MacMillan 1990; Colbourne & Robertson 2000).

In the Nina Valley, only one released bird was observed to disperse outside the project area, and travelled more than 10 km from its release site within ten week of translocation. It is unknown if the bird was settling in this remote area or was continuing to disperse because no information was available on its dispersal path between the Nina Valley and the location where it was eventually struck by a car. The remaining seven translocated wild-caught adults stayed within the valley. However, three of them were monitored for just under one year due to either dropped transmitters or probable transmitter failure, so their longer-term movements remain uncertain. Post-translocation monitoring of the ten previously released ONE subadults did not indicate dispersal outside the Nina Valley either (unpublished data). Our monitoring and bird recaptures in the Nina Valley confirmed the survival of two ONE birds for 4.2 years, one for 5.2 years, and three birds for at least 6.2 years post-release.

The absence of a clear dispersal pattern among translocated birds post-release is similar to other roroa reintroduction projects. Translocated roroa both at Lake Rotoiti (Gasson 2005) and in the Flora Valley project areas (Toy & Toy 2020) displayed high variability in overall dispersal distance and dispersal period before settling to stable home ranges. Interestingly, both projects observed shorter dispersal periods and distance in established translocated pairs that stayed together than those individuals who either re-paired or were translocated without a mate. Such behaviour is consistent with the observed dispersal in the Nina Valley, where the only pair that did not separate (‘wild 1’) showed the shortest dispersal path, distance, and lowest dispersal speed and UD, a pattern of behaviour similar to the resident ONE birds with established territories (Table 3.1). In contrast, pairs that separated, temporarily or permanently, moved around substantially more before settling down. Pair 'wild 1' was the only pair in the Nina placed in the release burrow together, while individuals from the other pairs were placed approximately 20 m apart, but this factor did not seem to play a role for pair bond survival at Lake Rotoiti or the Flora Valley.

The straight-line dispersal distance was the highest in the pair that separated soon after release ('wild 3') and the individuals headed in near-opposite directions (Fig. 3.1, Table 3.1). Only one of them, the female dispersing over 5.5 km upstream, likely remained in the valley, at least during the monitoring period. Pairs 'wild 2' and 'wild 4' also had long dispersal paths, but repeatedly changed direction resulting in larger UDs, although within the Nina Valley. Large dispersal distances up to 10 km from the release site were also observed in some roroa translocated to the Flora Valley, resulting in at least 14% of the birds (6 out of 44) settling outside the project area (Toy & Toy 2020). Similarly, one individual had a dispersal path over 11 km within a year post-release at Lake Rotoiti. However, the project area at Lake Rotoiti is delineated by natural barriers, the lakeshore on one side a high mountain range on another, which likely limited the dispersal to within the project area boundaries (Gasson 2005).

3.4.2 Home range establishment

The post-translocation monitoring of two years for the released wild-caught adults only produced observable stable home ranges for three birds. Due to the noted transmitter difficulties, the remaining four birds were only monitored for 0.9–1.5 years post-release, resulting in a home range estimation based on 0.4–1 year of data points. During this shorter monitoring period, these birds still had increasing home range areas, so it was likely the home ranges were not fully realised yet. In the Flora Valley project area, roroa have been observed to disperse for up to 2.5 years before establishing stable home ranges, based on monitoring data of up to eight years post-release (Toy & Toy 2020). Therefore, it is possible that the home ranges of the four birds with shorter monitoring duration could have kept expanding or shifting before eventually stabilising.

The estimated home range size (MCP) of the translocated wild-caught birds in the Nina, at 76.34 ha (±11.16 SE), was similar to the mean annual home range size

(July–June each year) in the Flora Valley, 73.26 ha (±4.82 SE), based on an average 3.8 years post-release monitoring duration for each bird (Toy & Toy 2020). In contrast, the mean home range size of translocated roroa at Lake Rotoiti 6–8 years post-release was 34.42 ha (±9.40 SE); however, the monitoring took place during only the winter season (Jahn et al. 2013) and therefore is not directly comparable. The mean home range size of translocated birds in the Nina Valley was larger than their pre-translocation mean home range in the Hawdon Valley, which was 54.39 ha (±5.13 SE). Although the difference was not statistically significant, given the truncated monitoring of four of the birds post-release, it is likely that the difference would be significant if monitoring for all birds could have been achieved for the full two-year period. The significantly larger home range estimates of translocated wild adults compared to the resident ONE birds in the Nina Valley was likely caused by an ongoing range shift/expansion. In other naturally established populations, the home ranges of adult roroa appear substantially smaller, such as in the North Branch Hurunui, Lake Sumner Forest Park (32.64 ha ±2.15 SE, summer–mid-autumn only), or in Gouland Downs, Kahurangi National Park (pair territory size 23 ha, range 9.9–42 ha) (McLennan & McCann 1991; Keye et al. 2011).

3.4.3 Translocation impacts on resident birds

The release of the wild-caught adults into the Nina Valley did not appear to substantially impact the ranging behaviour of the previously released ONE birds, likely due to a very low population density and little need to compete for resources. Apart from a minimal temporary home range shift of unpaired male 'ONE 3' and an insignificant increase in the ONE birds' nightly activity immediately after the release of wild adults, there were no other obvious behavioural changes among the ONE birds (Mander 2016). The ONE birds' home ranges (MCP) before-and-after the 2015 translocation were not identical but had a substantial overlap, which is consistent with a naturally occurring range shift over time (Toy & Toy 2020). Additionally, we found no significant change in the home range size of the resident ONE birds following the release of the wild adults suggesting that the ONE birds were successful in maintaining/defending their territories after the release of the wild-caught birds. The monitoring periods were not the same in length, as we compared home ranges 24 months before and 6–24 months after the release of wild adults. The pre-translocation period was longer due to data points being collected less frequently than after the translocation, but we did not expect it to affect the results.

The comparison of ONE birds' home ranges should, however, consider the transitioning between age class of the monitored birds. While the ONE birds were already adults by the time of the 2015 translocation, they were only recruited to the adult population during the 2-year pre-translocation monitoring period. The ONE birds were 2.4–3.4 years old at the start of the monitoring period, and they would be considered adults at four years or whenever they start breeding (Colbourne et al. 2020). Subadult roroa have been shown to frequently share the territory and even the nesting burrow with their parents (Jahn et al. 2013; Toy & Toy 2021a), unlike subadult North Island brown kiwi that usually disperse and establish their own territories (Basse & McLennan 2003). Given that all of the monitored ONE birds appeared settled within 2 km from their original release sites and there was no need to disperse from natal territories, we assumed their ranging behaviour was similar to those of adults throughout the pre- and post-translocation monitoring periods.

3.4.4 Future of the Nina population

Since 2011, 18 roroa have been released in the Nina Valley, ten ONE and eight wild-caught birds. Of these, two birds (one ONE and one wild) are known to have died. The 2015 translocation proposal planned for subsequent releases to establish a self-sustaining and genetically viable population founded by at least 40 unrelated individuals by 2020 (Morrison & Yong 2014), but this target has not yet been met. All of the released birds were sourced from the lower Hawdon Valley. The ten ONE birds were produced by seven different pairs and an offspring of one of these pairs died. Assuming that all birds last recorded alive in the Nina Valley survive and breed, the current founder group is 13 unrelated individuals: seven wild-caught adults and ONE offspring of six different pairs in the Hawdon Valley. However, most of these birds come from adjacent territories in the Hawdon Valley, and despite not knowing their pedigree, a degree of some relatedness is likely (Taylor et al. 2021).

The possibility of supplementing the reintroduced Nina population by natural immigration is very low. Prior to the reintroduction project, roroa in the Nina and surrounding valleys had likely been functionally extinct, with only occasional calls reported (Hulsman et al. 2010). None of the translocated birds is known to have paired up with any original birds that may have survived in the Nina Valley. During a 2012 acoustic survey, only a few roroa calls were recorded, which were likely to have been from then recently released ONE birds (Morrison & Yong 2014; Jahn et al. 2022b). Based on an acoustic survey in 2017–18 (Jahn et al. 2022b), it appears that the roroa population in the Nina Valley is growing due to successful breeding by translocated birds. Therefore, roroa releases to the Nina should resume as soon as possible to avoid genetic overrepresentation among the progeny of the current founder group and potential inbreeding. Failure to establish the population with a sufficiently genetically diverse founder group may lead to inbreeding depression or genetic drift, which may compromise the long-term population sustainability and ultimately lead to local extinction (Groombridge et al. 2012; Jamieson & Lacy 2012; Weeks et al. 2015).

The current species management plan marks completion of the Nina reintroduction project as a high priority and identifies an issue of insufficient pest control in the project area (Roroa Practitioner Group 2021). Only approximately 1600 ha of the valley is trapped for stoats, mainly alongside the Nina River. Such an area could theoretically cover approximately 25 roroa territories, based on the observed average home range size. However, existing home ranges appear to be spread on the valley slopes, and therefore only a limited portion of each territory is managed for predators along the valley floor. Currently proposed translocation guidelines recommend that translocation project areas should provide habitat for at least 100 pairs to allow sufficient retention of genetic diversity (Department of Conservation 2018). That will require the entire Nina River catchment to be under a sustained pest control regime ideally with a buffer zone covering surrounding valleys to provide safe space for post-release or natal dispersal from the Nina Valley.

3.4.5 Implications for kiwi translocations

The post-translocation behaviour of roroa in the Nina Valley underscores large habitat size requirements for kiwi reintroduction projects in unfenced mainland areas. Large UDs and long dispersal paths show the need for intensive post-release monitoring that will inform management interventions such as retrieval of dispersed birds, as demonstrated in the Flora Valley project area (Toy & Toy 2020), or in other large flightless birds, e.g. takahē (*Porphyrio hochstetteri*, Department of Conservation 2020). Radio telemetry is a commonly used method for monitoring translocated kiwi populations, but the monitoring period and effort are highly variable, and usually, only a sample of released birds is monitored for only a part of the dispersal period (Jahn et al. 2022a). Extended monitoring duration and increased numbers of monitored birds enable better adaptive management, detection of likely population founders based on territory establishment and breeding, and selection of future release sites based on gaps between territories. However, such approach can be more expensive, labour-intensive, and intrusive to radio-tagged birds (Toy & Toy 2021b). Subsequent periodic acoustic surveys, coupled with occupancy analysis (Jahn et al. 2022b) and potential identification of individuals by their calls (Digby et al. 2014; Dent & Molles 2016), can facilitate non-intrusive and cost-effective population monitoring. Additionally, regular genomic assessments can provide a tool to identify and manage possible inbreeding depression or genetic drift (Ramstad & Dunning 2021), and therefore maximise the probability of a long-term positive translocation outcome.

3.5 Acknowledgements

We want to acknowledge the contribution of Lincoln University staff, students, and volunteers assisting with the fieldwork and data collection, namely Michael Hargraves, James Ranstead, Myles Mackintosh, Marc Filla, Cameron Mackenzie, Jenny Dent, Bryn Williams, Robin Pieper, Andrea Pavlasová, Roseanna Gamlen-Greene, and many others. We are also thankful to the staff of the DOC Rangiora office, particularly Sandy Yong, for providing data from the local kiwi population database and hut passes to allow the use of DOC huts during the fieldwork. Additionally, we thank Fernando Cagua for help with the data analysis and the Hurunui College Nina Valley Restoration Group for assistance and co-operation with the post-translocation monitoring in the Nina Valley. We are thankful to Hugh Robertson, Robin Toy, and Craig Symes for their insightful comments to improve the manuscript. PJ was supported by a Lincoln University Doctoral Scholarship and a Gordon Williams Postgraduate Scholarship in Ecological Sciences.

Chapter 4

Acoustic monitoring and occupancy analysis: cost-effective tools in reintroduction programmes for roroa–great spotted kiwi

4.1 Introduction

Conservation translocations, together with *in situ* management interventions, are important tools to safeguard threatened species and avert the risk of their extinction. For these efforts to be successful, it is crucial to understand how individuals of the target species are spatially distributed and how their populations respond to applied management measures (Nichols & Armstrong 2012; Robinson et al. 2018; Metcalf et al. 2019). Monitoring programmes generally attempt to estimate population trends and identify factors that lead to changes in abundance and distribution (Marsh & Trenham 2008). However, monitoring rare and cryptic species can be technically challenging and labour-intensive (MacKenzie et al. 2005; MacKenzie et al. 2018). So, there is a need to develop sensitive cost-efficient methods for effective monitoring.

One of the emerging tools to monitor animals is passive acoustic monitoring (PAM) (Teixeira et al. 2019), which is increasingly used for monitoring cryptic species, such as kiwi (*Apteryx* spp.), and has been widely used in avian studies (Furnas & McGrann 2018; Metcalf et al. 2019; Franklin et al. 2020). PAM enables more extended survey periods by leaving automated recorders at study sites and is less demanding on personnel than traditional call counts by human surveyors, given that a single person can deploy multiple recorders within a day (Digby et al. 2013b; Shonfield & Bayne 2017). Acoustic monitoring in kiwi management tends to focus either on call counts or presence/absence solely based on detection/non-detection (Robertson & Colbourne 2017), which implies an assumption of complete or near-complete detection of target individuals. However, in most survey regimes, the detection probability (detectability) is <1 (Gu & Swihart 2004). The consequences of imperfect detectability tend to result in underestimation of occupancy in the studied area and thus biases in estimates of rates in population change (Seddon et al. 2011).

Therefore, it is essential to account for imperfect detection probability when using PAM to eliminate occupancy estimate bias.

Occupancy modelling addresses the differences in species detectability at a given study site by incorporating the probability of detection to estimate the site's true occupancy (MacKenzie et al. 2002). Occupancy is defined as the proportion of a study area where the species occurs or a fraction of landscape units occupied by the species (MacKenzie et al. 2005; MacKenzie et al. 2018). By assigning a probability that the species occurs in the sample unit, despite the non-detection, we estimate true occupancy. This process is usually more straightforward than estimating abundance for cryptic species, and it can be considered as a surrogate for abundance (MacKenzie et al. 2018) following appropriate calibration (Royle & Nichols 2003). The utility of occupancy modelling for site-faithful territorial species has been demonstrated in estimating abundance (Tingley et al. 2016) and thus is promising for kiwi studies.

Kiwi species are iconic New Zealand birds. All kiwi taxa are classified either at risk or threatened (Robertson et al. 2017) and are the target of conservation management across the country. Populations of all taxa are managed to various degrees, either *in situ* or by translocations (Innes et al. 2015b). However, the population response to management is challenging to measure because of the kiwi's elusive behaviour and the often remote and rugged areas that they inhabit. Monitoring of kiwi translocations mainly focuses on translocated populations, but is also useful for understanding the impacts of the birds' removal on source populations, understanding whether and how quickly the populations recover and informing further translocation planning.

Outcomes of kiwi translocations are often monitored by radio-tracking of released birds (Robertson & Colbourne 2017), which typically takes place over several months to a few years post-release (Jahn et al. 2022a). Due to the cost of radio-transmitters and labour intensiveness of this method, only a subsample of the released birds is usually monitored. Long-term outcomes of some kiwi translocations are evaluated by the mark-recapture method (Robertson et al. 2019a, 2019c), which is also labour intensive, intrusive to the monitored birds, and with limited practicality over large areas with low-density populations (Robertson & Colbourne 2017). In contrast to these methods, occupancy analysis based on acoustic data allows inferences on population status and change through time without the need for intrusive capture and handling, is easily scaled up over large areas, and is better suited for long term population studies (Noon et al. 2012). Therefore, occupancy modelling displays great promise for evaluating management outcomes for kiwi species.

Great spotted kiwi or roroa (*Apteryx maxima*, previously known as *A. haastii*, Shepherd et al. 2021) were reintroduced to the Nina Valley in Lake Sumner Forest Park between 2011–2015 in the course of a local ecosystem restoration project. Eighteen birds were released to date: eight wild-caught adults and ten subadults hatched in captivity from wild-sourced eggs as part of head-starting efforts under the Operation Nest Egg programme (Colbourne et al. 2005). This translocation was the first, and to date only, reintroduction of the Arthur's Pass–Hurunui population of roroa. It is vital to assess the outcome of this reintroduction and the impacts of the removal on the source population. In the Nina, we monitored translocated birds by radio telemetry for two years after their release to track post-release dispersal and establishment of territories. However, even though this period likely captures the population establishment phase post-translocation, the radio telemetry provided only a limited insight into the development of the distribution and abundance of kiwi in the Nina Valley. The source population in the Hawdon Valley, Arthur's Pass National Park, was not monitored using radio telemetry following the birds' removal, so it did not allow us to determine whether the vacated territories were re-occupied or any other changes occurred in the population that would bar future population harvest. Hence, we used data from acoustic surveys in 2012–13 from both areas and carried out follow-up surveys in 2017–18 to examine occupancy. The occupancy analysis based on acoustic data allowed us to answer the following questions:

a) How did the distribution of roroa develop in the Nina Valley following their reintroduction?
b) What is the site occupancy of roroa in the Nina Valley post-translocation?
c) How was the site occupancy of roroa in the Hawdon Valley affected by removing the adult territorial individuals for translocation?

4.2 Methods

4.2.1 Study areas

Kiwi acoustic surveys were undertaken in the Nina Valley (42°28’ S, 172°19’ E) in the Lake Sumner Forest Park, and the Hawdon Valley (42°57’ S, 171°45’ E), Arthur’s Pass National Park. Both areas are located on the eastern side of the main divide near the Lewis Pass and the Arthur’s Pass, respectively, suggesting similar climatic conditions. The Nina Valley floor is at an altitude between 610–860 m a.s.l. and is surrounded by mountain peaks between 1500–1780 m high. The Hawdon Valley floor is at an altitude between 570–780 m and is surrounded by mountains 1400–1930 m high. Native montane beech forest covers steep slopes and lower terraces in both valleys until reaching the bush line at approximately 1300 m. The dominant tree species are mountain beech (*Fuscospora cliffortioides*), red beech (*F. fusca*), and silver beech (*Lophozonia menziesii*) (Read & O’Donnell 1987; Blakely et al. 2008). Both study areas are managed for invasive predators such as stoats (*Mustela erminea*) and possums (*Trichosurus vulpecula*), which are the main predators of juvenile kiwi and eggs (McLennan et al. 1996). However, while limited trapping for these predators occurs in the Nina Valley, the Hawdon Valley receives much more intensive pest management in terms of trapping effort and aerial applications of 1080 (sodium fluoroacetate) poison.

4.2.2 Species description

Great spotted kiwi or roroa is an endemic species to the South Island of New Zealand, inhabiting mainly montane beech and podocarp forests. They are ecologically similar to other kiwi species – cryptic, nocturnal, ground feeders and habitat generalists (McLennan & McCann 2002). The present range of roroa is restricted to the northwest of the South Island, separated into four main populations: northwest Nelson, Westport, Paparoa Range, and the Arthur’s Pass–Hurunui region (Germano et al. 2018). Both males and females produce sexually dimorphic whistle calls (Dent & Molles 2015), which are often recorded and used for monitoring populations and distribution surveys (Robertson & Colbourne 2017). The calling behaviour of roroa likely serves to maintain territories and the pair bond, and aid communication between males and females that co-operatively incubate eggs and take turns between feeding and tending the egg, similarly to other kiwi species (Digby et al. 2013a; Colbourne & Digby 2016).

4.2.3 Acoustic surveys

Our main objective was to compare occupancy before-and-after the 2015 translocation event and between the study areas. The acoustic surveys in the Nina and Hawdon valleys took place between August 2017 and June 2018, which allowed time for the translocated birds in April 2015 (eight wild-caught adults/four pairs) to settle in and establish new territories. Additionally, it provided sufficient time for ten subadults released between 2011–13 (two subadults were released after the 2012–13 surveys) to mature and establish territories. Because we wanted surveys to be directly comparable, the survey followed the same design as the acoustic surveys in 2012–13 in the Nina and Hawdon valleys carried out by the Department of Conservation (DOC). All sites were within the same habitat type – montane beech forest usually at the mid- or lower half of the forested valley slopes. DOC Electronics acoustic recorders (Department of Conservation 2019) were placed on trees at the same sites in the same month of the year to replicate similar detection probabilities between surveys at each site. The acoustic sampling rate was set at ‘Low’ – 8 kHz, creating 16-bit resolution WAV files. The same type of recorders (AR3) was used in both surveys to minimise systematic variability in the ability to capture kiwi calls.

In the Hawdon Valley, recorders’ positions were based on the previous radio telemetry monitoring of kiwi done by the DOC staff; individual recorder units were placed approximately in the centre of presumed kiwi territories to maximise the detection probability. Topography features were also considered and steep gullies with streams were avoided due to high volumes of noise and the assumption that kiwi use these features as territorial boundaries. The recorder locations generally followed a transect sampling scheme, however, there were gaps caused by logistical issues during the 2012–13 survey year. In the Nina Valley, the recorders were spatially distributed similarly to the Hawdon Valley, based on the assumption of equivalent territory size and ranging behaviour to roroa in other studies (Keye et al. 2011; Jahn et al. 2013). The mean distance between neighbouring recorders in the Nina and Hawdon valleys were 909 and 1174 m, respectively. Such distances are similar to the currently recommended guidance to position recorders at least 1 km apart to minimise detection of the same individuals by multiple recorders (Robertson & Colbourne 2017). Only the area adjacent to the recorder was effectively surveyed, which was likely only a part of the kiwi home range, even though the exact detection radius was unknown. However, we made inferences about the occupancy within the whole sample unit (MacKenzie et al. 2002), which aims to encompass the target animals' home range boundaries (Noon et al. 2012).

In both survey years, 2012–13 and 2017–18, 23 and 21 recorder units were placed in the Nina and Hawdon valleys, respectively (Figs 4.1 and 4.2). The recorders were activated approximately 30 min after sunset and stayed on until 30 min before sunrise (8.5–14 hours), according to best practice guidance (Robertson & Colbourne 2017). In both survey years, the recorders ran for up to three weeks to maximise the number of survey nights. If it was evident upon retrieval that a recorder had failed to record data, we immediately replaced it and attempted another recording period. Given the limited number of recorders available, we placed them at survey sites in stages, which meant that nearly a year was needed to survey all sites. Specifically, in the Nina, recorders were deployed between August–December and May–June. In the Hawdon, the recording periods were August–October and February–April. The first recording period at both study areas fell into the breeding season of roroa, whereas the second period fell into the non-breeding season (Heather & Robertson 2015). Despite this relatively long period to complete the surveys, it was likely that it would not affect the results of occupancy analysis due to the high stability of territories and longevity of roroa exceeding 25 years, even at sites without predator control (Robertson et al. 2005).

4.2.4 Kiwi call count

To model occupancy, we first searched for kiwi calls captured by the acoustic recorders. We reviewed raw acoustic data using Kaleidoscope Pro v.4.5 analysis software (Wildlife Acoustics 2019). We first trained Kaleidoscope to scan for roroa vocalisations using a sample of 7250 15-minute recordings collected from automated recorders placed within the territories (and in many cases near known nests) of kiwi pairs in the Hawdon Valley between 2013–2015. Our initial scan searched for target sounds with the following characteristics: frequency range of 650–3000 Hz, with a total duration of 6–40 s and a maximum between-syllable gap of 1.5 s. The FFT (Fast Fourier Transform) window size was set to 10.67 ms. Kaleidoscope uses Discrete Cosine Transform coefficients of spectra from sequential FFT frames to build feature vectors representing each target sound. K-Means clustering and Hidden Markov Models are then used to determine initial clusters among these feature vectors. Following the initial scan, we manually labelled vocalisations identified as male or female roroa and re-scanned the same recordings. In this step, Kaleidoscope uses human-supervised labelling to refine clusters and better separate target sounds from other kinds of recorded animal vocalisation (such as calls of ruru – *Ninox novaeseelandiae*).

We used the resulting classification algorithm from the training process to auto-detect roroa vocalisations in all recordings collected during the 2012–13 and 2017–2018 survey years. Following Kaleidoscope's auto-detection, we manually confirmed the auto-detection results by visually inspecting the spectrograms and, where necessary because of unclear spectrograms, listening to the identified calls, including those marked as other species. We removed any false positives and captured false negatives (calls initially labelled as other species) through this approach and minimised the need for more complex models adjusting for misclassification of the calls (Miller et al. 2011). We classified all calls as either male or female, and we considered male and female calls as duets if they likely occurred in the same territory (i.e. were captured by the same recorder) within a minute of one another (Robertson & Colbourne 2017). The visual inspection and confirmation process took approximately one hour of manual work per 1000 hours of raw acoustic data. Each detected kiwi call was accompanied by information on the date, time, and recorder site. For each survey night, we recorded the survey's duration and environmental variables such as the local daily amount of rain and daily average wind speed obtained from the Niwa Virtual Climate Station Network (https://data.niwa.co.nz).

4.2.5 Occupancy analysis

To analyse the occupancy at listening sites, we used the detection of at least one call versus zero calls as presence/absence data for every night that a recorder was active at each site. This approach allowed us to construct a detection history of kiwi calls for up to 21 consecutive survey nights, provided that the batteries lasted this whole period. Visualisation of naïve site occupancy was done with the package 'ggmap' in R (Kahle & Wickham 2013; R Core Team 2020). Subsequently, we estimated site occupancy ψ (the probability of species presence) and the detection probability p (MacKenzie et al. 2002) using the R package 'unmarked' 1.0.1 (Fiske & Chandler 2011). We analysed occupancy using single-season models (MacKenzie et al. 2002) based on kiwi presence/absence regardless of sex. The low number of detected female calls did not allow for more advanced co-occurrence analyses of sexes. However, we reported the detected presence of sexes and possible pair status of birds separately to understand better the breeding potential and population structure in each study area.

The single-season models allowed a comparison of the differences between the site occupancy when the study area (Nina and Hawdon) and survey year (2012–13 and 2017–18) were treated as covariates for occupancy ψ. We did not use a dynamic multi-season occupancy model (MacKenzie et al. 2003), because we had data only from two survey years and the focus of the analysis was not to model long-term population dynamics, but the differences between the two survey years, for which the single-season model provided sufficient tools.

Our analysis assumed that site occupancy was constant within each survey year, which implies static home ranges and no colonisation or local extinction during the survey year. The intensity of the pest management regime, potentially affecting occupancy, differed between study areas, but within study areas occupancy was assumed to be constant with respect to habitat type and pest control intensity. In addition, we considered the habitat type identical at all listening sites across both study areas. To account for the variable detectability of kiwi calls, we included four covariates possibly affecting detectability: length of survey night, breeding/non-breeding season, precipitation (mm/day), and wind speed (m/s, daily average). We included the breeding/non-breeding season because of an observed call rate variability between the seasons, and the environmental factors because it was shown that kiwi calls, notably lower-frequency female calls, are often masked by strong wind and heavy rain (Colbourne & Digby 2016). To account for different survey durations at each site caused by the different length of night throughout the year and varying battery capacity, we included this metric as another covariate affecting detection probability. It has been proposed that moonlight illumination might affect kiwi call rates, affecting detectability (Colbourne & Kleinpaste 1984). However, we did not include moon illumination as it was shown not to have a significant effect on roroa male calls and it correlates with a reduction in female vocalisations only at the brightest moon period (Colbourne & Digby 2016).

Finally, we analysed occupancy in the Nina and the Hawdon and the difference in occupancy between 2012–13 and 2017–18 by fitting a set of 16 candidate models to the data. We assumed that detection probability would be affected by the survey night length and interaction between the study area and the survey year in each model. The models varied based on the inclusion or absence of breeding/non-breeding season, wind speed, and rainfall covariates affecting the detection probability. All models assumed that occupancy is specific to the study area and survey year (by including the interaction between these two factors). We also tested models assuming that the survey year had the same effect in both study areas. Subsequently, we evaluated the covariates' impact by selecting the model with the most parsimonious Akaike's Information Criterion (AIC) (Burnham & Anderson 2002).

4.3 Results

4.3.1 Detection of kiwi calls

Acoustic data was successfully recorded from most recording sites during both survey years (Table 4.1). The number of detected kiwi calls at each study area substantially varied as there were nearly 12x more calls detected in the Hawdon than in the Nina Valley, in total. The numbers of male kiwi calls were substantially higher than female calls across both survey years and study areas.

Table 4.1 Numbers of recorders and kiwi calls detected during 2012–13 and 2017–18 roroa acoustic surveys. Duet calls consist of both male and female calls and are counted separately in the total call count.

study area	survey year	recorder sites	active recorders	kiwi calls total	male	female	duet
Nina	2012–13	23	21	14	13	1	0
	2017–18	23	22	91	84	7	0
Hawdon	2012–13	21	20	578	403	95	40
	2017–18	21	20	636	510	66	30

In the Nina, only three sites recorded kiwi calls in 2012–13, in the vicinity of the release site where five subadult birds were translocated to in early 2011 (Fig. 4.1). Three more subadults were released in the valley in early 2012, but no recorder site detected kiwi calls near their release site. The number of recorder sites detecting kiwi calls increased to 15 in 2017–18, following the release of two more subadults in

2013 and eight adults in 2015. Most of the sites detecting kiwi calls were near the release areas from 2011–15 translocations and no recorders detected kiwi in the top of the valley. In 2012–13, only one site in the Nina detected male and female calls indicating the presence of a potential pair, the remaining two sites with detections recorded males only. In 2017–18, the only pair detected was at the same site as in 2012–13. In contrast, eleven sites recorded males only and three sites recorded female calls only, despite both males and females being detected by radio telemetry and recapture in 2017 near several of the recorder sites (unpublished data).

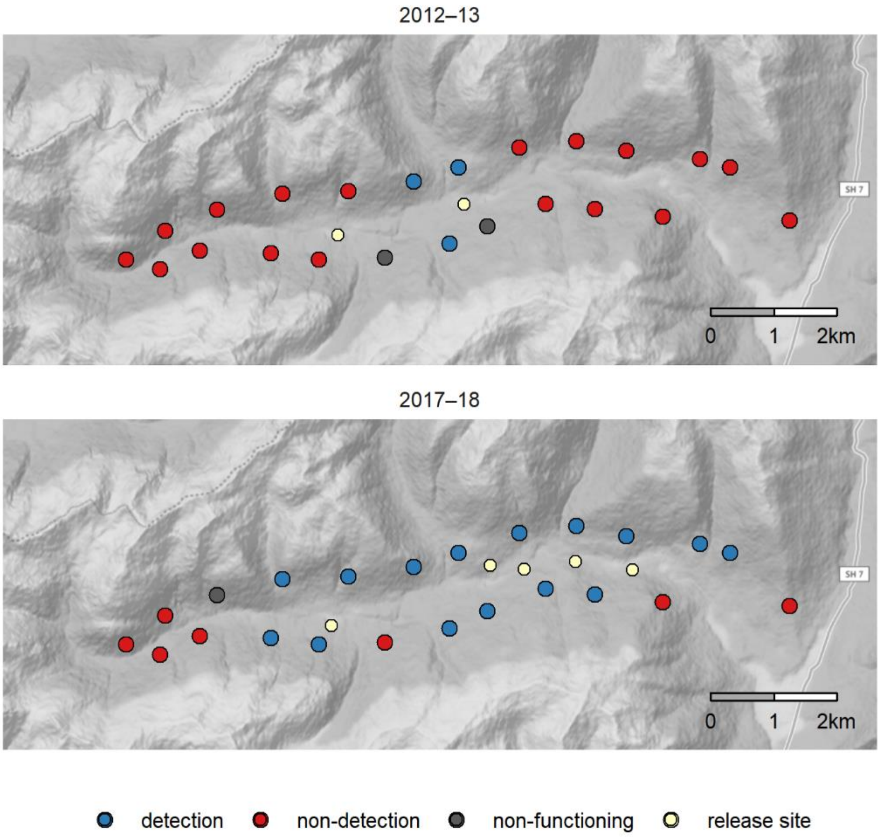

Figure 4.1 Site occupancy of roroa in the Nina Valley increased between 2012–13 and 2017–18. Sites detecting kiwi calls are in blue, non-detecting sites in red, and non-functioning recorders in grey. Release sites of reintroduced kiwi are displayed in yellow. Birds were released prior to the 2012–13 survey year at two sites shown on the top map. Five more release sites from between 2012–13 and 2017–18 survey years are displayed on the bottom map.

In the Hawdon Valley, 13 sites recorded kiwi calls in 2012–13 (Fig. 4.2). All but one of these recorded both male and female calls indicating the presence of at least 12 potential pairs. The remaining site detected only male calls. In 2017–18, the number of sites detecting kiwi increased to 18, and all of these recorded both male and female calls, which indicates the presence of at least 18 potential pairs. These included three recording sites within known territories of adult birds that were translocated to the Nina in 2015. Detection of both male and female calls during both survey years suggests that these territories were re-occupied by another pair within three years following the original pairs' removal. The recording site nearest to (but not in the centre of) the fourth pair's territory did not detect any kiwi calls at either of the survey years.

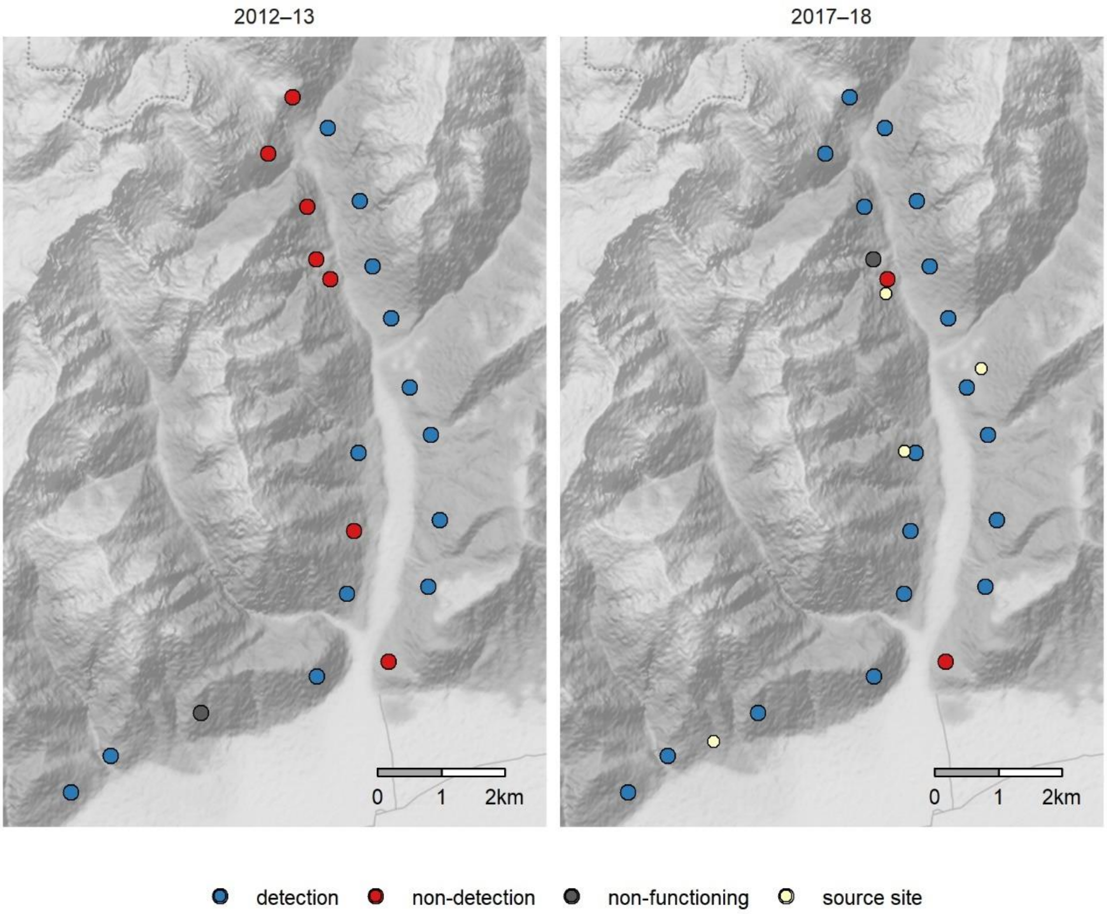

Figure 4.2 Site occupancy of roroa in the Hawdon Valley. The occupancy increased between 2012–13 and 2017–18 despite removing eight adults from four sites (displayed in yellow) and translocating them to the Nina Valley in 2015. Sites detecting kiwi calls are in blue, non-detecting sites in red, and non-functioning recorders in grey.

4.3.2 Occupancy analysis

We fitted a set of 16 candidate models to the data (Table 4.2). The most parsimonious model suggests the influence of survey night length, breeding/non-breeding season, wind speed, and the interaction between the study area and survey year on the detection probability (summary of the data in Table 4.3). The evidence to include the rain accumulation in the model is weaker, however not negligible. We fitted models both with and without the interaction between the study area and survey year as a covariate affecting occupancy. Without this interaction, the models showed a smaller AIC as both study areas, Nina and Hawdon, displayed a similar pattern in occupancy change between the survey years.

Table 4.2 Models of roroa site occupancy in the Nina and Hawdon valleys. K denotes the number of parameters. The model structure includes covariates affecting occupancy ψ and detection probability p: study area, survey year, length of survey night, breeding/non-breeding season, wind speed, and amount of rain.

model	**ΔAIC**	***K***	**–2LogLike**	**weight**
ψ(area + year), p(area * year + length + season + wind)	0.00	10	985.55	0.35
ψ(area + year), p(area * year + length + season + wind + rain)	1.62	11	985.17	0.16
ψ(area * year), p(area * year + length + season + wind)	1.77	11	985.33	0.15
ψ(area + year), p(area * year + length + wind)	2.57	9	990.13	0.10
ψ(area * year), p(area * year + length + season + wind + rain)	3.39	12	984.94	0.06
ψ(area + year), p(area * year + length + wind + rain)	3.84	10	989.40	0.05
ψ(area * year), p(area * year + length + wind)	4.21	10	989.77	0.04
ψ(area + year), p(area * year + length + season)	5.17	9	992.73	0.03
ψ(area * year), p(area * year + length + wind + rain)	5.48	11	989.03	0.02
ψ(area * year), p(area * year + length + season)	6.99	10	992.54	0.01
ψ(area + year), p(area * year + length + season + rain)	7.17	10	992.73	0.01
ψ(area + year), p(area * year + length)	7.57	8	997.12	0.01
ψ(area * year), p(area * year + length + season + rain)	8.99	11	992.54	0.00
ψ(area * year), p(area * year + length)	9.29	9	996.84	0.00
ψ(area + year), p(area * year + length + rain)	9.50	9	997.06	0.00
ψ(area * year), p(area * year + length + rain)	11.23	10	996.78	0.00

Table 4.3 Summary data for the occupancy model covariates. Values for survey night length, wind speed, and rain accumulation display their mean and value ranges.

study area	**survey year**	**breeding / non-breeding nights**	**survey night length [hrs]**	**wind speed average [m/s]**	**rain [mm/day]**
Nina	2012–13	168 / 87	11.1 (0.5–14.0)	2.7 (1.2–4.8)	9.5 (0–61.8)
	2017–18	259 / 111	10.0 (0.3–13.8)	2.9 (1.3–5.3)	5.2 (0–49.4)
Hawdon	2012–13	142 / 122	9.0 (0.4–12.0)	3.3 (1.7–6.4)	2.9 (0–28.0)
	2017–18	157 / 181	9.7 (0.2–12.5)	3.4 (1.8–6.7)	5.2 (0–78.8)

Detection probability *p* was higher during the roroa breeding season (August–December), as opposed to the non-breeding season. Also, as expected, detectability increased with longer survey nights at both study areas and survey years given that longer recording times raise the chance of recording a calling roroa moving through its territory within the acoustic range of the recorder (Fig. 4.3). The length of the survey night varied throughout the year due to changing daylight duration (programmed between 8.5–14 hours) and decreasing battery charge after several consecutive survey nights. The differences in detection probability between the study areas vary with the overall numbers of kiwi calls and the time until the first call was detected (Table 4.4). While in the Nina it took on average 3.3 and 4.5 nights to detect a kiwi call for the first time in 2012–13 and 2017–18, respectively, in the Hawdon it was in less than two nights on average during both survey years.

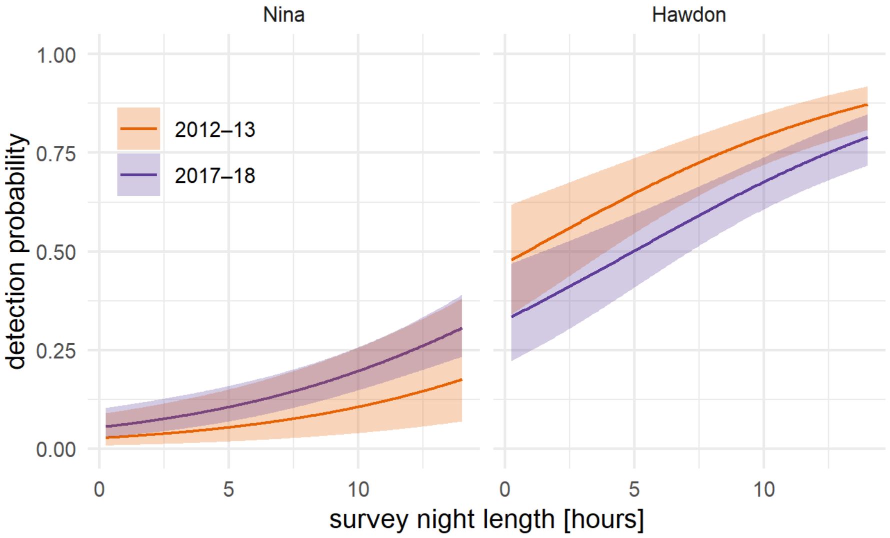

Figure 4.3 Detection probability in the Nina and Hawdon valleys increases with the length of the survey night. Shaded areas represent one standard error around detection probability estimates projected for the breeding season with wind speed at its mean observed values. The detection probability is projected for the survey night length ranging between the observed 15 min–14 hours. The recorders were set to record between 8.5–14 hours, however low battery charge at the end of the recording cycle sometimes resulted in substantially shorter survey nights.

Occupancy estimates ψ based on the top-ranked model were higher or similar to the naïve occupancy – the proportion of sites where kiwi calls were detected (Table 4.4). Despite substantial uncertainty around the estimates due to statistically small sample sizes in each survey year at each study area, there was a significant difference in overall site occupancy in the Nina between the survey years (Fig. 4.4, Table 4.5). As expected, the increase in occupancy took place after releasing additional birds into the valley following the first survey year and after sufficient time had passed for all released subadults to reach maturity and develop the calling behaviour typical of adults.

Table 4.4 Occupancy estimates increase between the survey years and the mean time to the first detection of roroa calls differ substantially between the study areas. Only sites detecting calls were included in the calculation of the mean.

study area	survey year	naïve occupancy	occupancy estimate ψ	standard error	survey nights	first detection night
Nina	2012–13	0.14	0.20	0.097	255	3.3 (2–4)
	2017–18	0.68	0.72	0.098	370	4.5 (1–10)
Hawdon	2012–13	0.65	0.63	0.103	264	1.3 (1–3)
	2017–18	0.90	0.95	0.037	338	1.9 (1–6)

A similar pattern in the occupancy change appeared in the Hawdon Valley. Both naïve and estimated occupancy increased in the Hawdon between the survey years, contrary to the expectation that site occupancy may be negatively affected by removing the eight adult birds from four territories. The significant increase in site occupancy (Table 4.5) suggests no adverse impact on the extant roroa population in the Hawdon by the removal of territorial adults for reintroduction elsewhere.

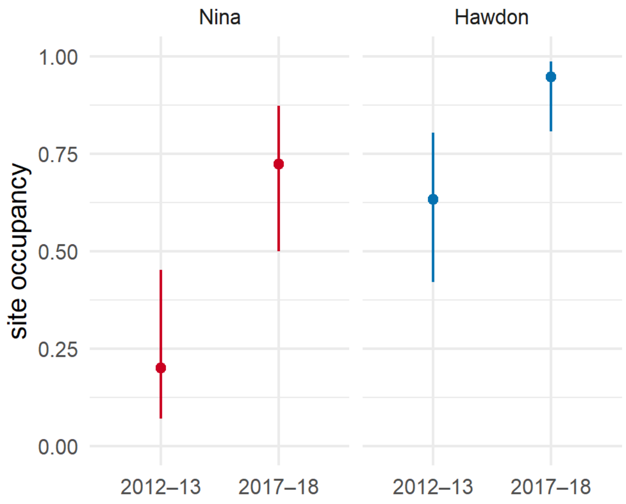

Figure 4.4 Estimated site occupancy in the Nina and Hawdon valleys show a significant increase between 2012–13 and 2017–18 survey years. Error bars display one standard error of the occupancy estimates.

Table 4.5 Occupancy and detection probability estimates relative to their covariates based on the most parsimonious model.

occupancy ψ (logit-scale):	**estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	0.55	0.44	1.24	0.21
area Nina	-1.93	0.69	-2.80	<0.01
year 2017–18	2.35	0.71	3.32	<0.01
detection probability p (logit-scale):	**estimate**	**SE**	**z**	**P(>|z|)**
(Intercept)	1.32	0.20	6.57	<0.01
area Nina	-3.47	0.59	-5.87	<0.01
year 2017–18	-0.60	0.21	-2.84	<0.01
survey length	0.50	0.09	5.40	<0.01
wind speed	-0.22	0.08	-2.65	0.01
season breeding/non-breeding	-0.36	0.17	-2.13	0.03
area Nina : year 2017–18	1.33	0.60	2.20	0.03

4.4 Discussion

The results from the acoustic surveys suggest that site occupancy by roroa increased in both the Nina and Hawdon valleys between the survey years 2012–13 and 2017–18 (Fig. 4.4). The naïve occupancy was generally lower than the estimated site occupancy, which highlights the need to address incomplete detectability during analysis to avoid occupancy underestimation (MacKenzie et al. 2002; Seddon et al. 2011). The significant increase in site occupancy in the Hawdon was surprising given that the 2017–18 survey year took place only a few years after approximately 20% of the known territorial adults in the surveyed area were removed to establish the population in the Nina as part of the roroa reintroduction project. Even though the exact roroa population size in the entire Hawdon Valley was unknown, it was estimated to be at least 20 territorial pairs based on the previous radio telemetry and acoustic monitoring by the DOC staff (unpublished data).

The increase in site occupancy in the Nina was largely expected because of the translocations of adult and subadult birds from the Hawdon. Additionally, the observed distribution of roroa in the Nina was consistent with the post-translocation radio telemetry monitoring (unpublished data). Detecting kiwi at recording sites near the translocation release sites indicates the establishment of a territorial structure of adjacent territories in the central part of the valley, the retention of most of the released birds, and possible recruitment of subadults.

The detectability of roroa calls differed between the study areas (Fig. 4.3). In the Nina, the detection probability was markedly lower than in the Hawdon during both survey years. The lower detectability corresponds to the overall lower number of calls detected in the Nina despite a similar survey effort, which may indicate a lower population density (Colbourne & Digby 2016). Consistently lower numbers of detected female calls suggest substantially lower acoustic detectability of females, even though we did not have enough data to formally analyse the difference between sexes across study areas and survey years. The higher detectability during the breeding season was consistent with expected higher call rates during mating and the beginning of the incubation period (Robertson & Colbourne 2017). In contrast, another study found lower call rates for roroa during the breeding season, which could potentially decrease detectability; however, this study also identified large interannual variability in roroa calling patterns (Colbourne & Digby 2016).

We also found evidence that detectability was influenced by wind speed and, to a smaller degree, by rain accumulation, which is consistent with other studies (Buxton & Jones 2012; Willacy et al. 2015; Colbourne & Digby 2016). Presumably, the evidence for the influence of rain would likely be stronger in our models if more detailed data on rain accumulation throughout the day was available. Finally, the detection probability could have also been affected by recorder sensitivity. Although all the recorders were of the same model, microphone sensitivity can be variable and possibly degrade over time (Turgeon et al. 2017). Unfortunately, the information on the age and previous use of all recorders was not available, and therefore we could not include it as a covariate in the occupancy modelling. We did not consider within-season microphone degradation substantial enough to have a dramatic effect on the results based on the testing of a sample of recorder units used in this study in a subsequent project (unpublished data).

Following our modelling, we assumed that most of the recorded birds were territorial individuals that would be detected only at one recording site. Nonetheless, wide-ranging non-territorial subadults could have been potentially detected at multiple sites. Post-translocation radio telemetry monitoring from the Nina (unpublished data) suggests that several adult birds were roaming widely in the first year post-release (until mid-2016) and likely settled into stable home ranges before the 2017–18 surveys took place. However, the likely lower density in the Nina led to the establishment of larger territories than in the Hawdon, as indicated by radio telemetry data from translocated adult birds from two years post-release monitoring. If some birds were detected at more than one recording site, model assumptions would be violated, and it would potentially lead to an overestimation of occupancy (Berigan et al. 2019). Therefore occupancy estimates in the Nina should rather be interpreted as location 'use' rather than 'site occupancy' (MacKenzie et al. 2004).

4.4.1 Conservation implications

PAM has proven to be a useful tool for monitoring and informing conservation management for cryptic and rare species (Teixeira et al. 2019), such as roroa in this study. Acoustic recorders have been shown to be able to detect comparable numbers of kiwi calls as human observers during listening surveys, and therefore can be highly efficient in monitoring kiwi populations (Digby et al. 2013b; Stewart & Hasenbank 2018). PAM techniques can generate extremely large volumes of raw recordings to process, but there is a rapidly expanding set of tools for automated processing and analysis, providing training data is available. For example, training Kaleidoscope software to identify roroa calls required dozens of examples of target calls as well as calls of non-target species that could be potentially confused with roroa. Occupancy models based on bioacoustics can be particularly useful in monitoring kiwi species and evaluating population response to conservation management, as shown in this study. Changes in site occupancy estimates displayed a positive response to the continuation of the roroa reintroduction programme in the Nina. In the Hawdon, the occupancy increase likely displayed a positive response to ongoing intensive pest mammal control by trapping and regular applications of 1080 (sodium fluoroacetate) poison, which presumably outweighed the negative impact of the birds' removal for the Nina translocation. Moreover, reliability of occupancy analysis has been demonstrated in other territorial bird species, such as California spotted owl (*Strix occidentalis occidentalis*), where changes in occupancy were shown to match population changes based on mark-recapture data (Tempel & Gutiérrez 2013). Additionally, the utility of bioacoustics-based occupancy analysis has been shown in several common and rare bird species, together with the potential to inform their conservation management (Campos-Cerqueira & Aide 2016; Furnas & McGrann 2018; Stiffler et al. 2018; Metcalf et al. 2019; Abrahams & Geary 2020). If expanded, large scale studies have the capability to detect even small changes in territorial occupancy with high confidence (Furnas & Callas 2015; Wood et al. 2019), which may be useful in monitoring populous and sparsely distributed kiwi species, such as roroa or tokoeka (*A. australis*).

Occupancy modelling is particularly promising in monitoring translocation outcomes alongside other types of data such as vital rates (Armstrong & Reynolds 2012). A specified site occupancy can be expressed as a target objective for a particular reintroduction project as an alternative to a population growth rate (Nichols & Armstrong 2012). Moreover, the benefits of occupancy modelling using PAM data include the ability to estimate differences in detectability caused by the use of various recorder models, animal behaviour, weather conditions, or differing survey effort over multiple survey years (Shonfield & Bayne 2017). As shown, this method is likewise useful for assessing impacts of the birds' removal on the source population and informing when further harvests are suitable. Since it also tracks distribution changes over time, it is an effective tool for monitoring reintroduced populations and their spread within and beyond designated project areas (Nichols & Armstrong 2012; Noon et al. 2012).

Advances in identifying individual birds by their calls, as demonstrated in roroa (Dent & Molles 2016), little spotted kiwi (*A. owenii*) (Digby et al. 2014), and other territorial birds species (Odom et al. 2013; Wood et al. 2021), have the potential to further improve inferences from PAM data (Juodakis et al. 2021). By identifying individual birds during territorial occupancy analysis, we should be able to account for wide-ranging individuals and assign them to a single site (Berigan et al. 2019). This approach avoids double counting of individuals at multiple recording sites and therefore ensures key model assumptions are not violated when estimating occupancy. In our case, it would allow us to determine if any bird was detected at more than one recording site in the Nina and whether these sites were truly occupied or transitionally used. Additionally, it will allow non-intrusive mark-recapture studies of population dynamics and estimating abundance based on territorial occupancy (Tingley et al. 2016). Therefore, PAM and associated occupancy analysis with individual identification could provide tools for cost-effective and long-term monitoring in kiwi reintroduction programmes – both at the translocated and source populations. Occupancy analysis – potentially coupled with individual identification – can also be used to demonstrate and measure the effectiveness of *in situ* conservation programmes and so to improve kiwi recovery management.

4.5 Acknowledgements

We are thankful to the staff of the DOC Rangiora office, particularly Sandy Yong, for lending a portion of acoustic recorders for the 2017–18 survey year and for providing data from 2012–13 acoustic surveys, which was collected by DOC staff and Jono Dobbs. The 2017–18 surveys would not have been possible without a research grant awarded by the Kiwis for Kiwi. We also thank Lincoln University for financial support in forensic data recovery from damaged memory cards and Fernando Cagua for advice on data analysis. PJ was supported by a Lincoln University Doctoral Scholarship and a Marj Davis Scholarship awarded by the Ornithological Society of New Zealand. We are grateful to two anonymous reviewers for their insightful comments to improve the manuscript.

Chapter 5
General discussion

This thesis aimed to review kiwi translocations, explore their role in kiwi conservation management, and investigate monitoring methods to evaluate translocation outcomes. First, in Chapter 2, the translocation review summarises information from over 100 kiwi translocations spanning from the second half of the 19th century until the present and identifies factors influencing translocation outcomes. Second, in the roroa reintroduction case study, Chapters 3 and 4 address the shortcomings of many kiwi translocations identified in the translocation review concerning inadequate post-release monitoring. These chapters demonstrate the utility of post-translocation monitoring methods to detect successful population establishment and subsequent growth.

5.1 Kiwi translocation review

Translocations are increasingly used in kiwi conservation management (Cromarty & Alderson 2013; Miskelly & Powlesland 2013). However, they lack a strategic direction and general understanding of what elements determine a successful translocation (Germano et al. 2018). In Chapter 2, the kiwi translocation review examines all known kiwi translocations from 1863 until 2018 to identify trends in the use of translocations in kiwi management. The dataset includes 102 translocations of all kiwi taxa and therefore is more comprehensive than any other published translocation review or summary (Atkinson 1990; McHalick 1998; Colbourne & Robertson 2000; Colbourne 2005; Miskelly & Powlesland 2013). While most of the historical translocations are mentioned in those reviews and summaries, the most significant contribution of Chapter 2 is the inclusion of translocations from the last two decades. More than half of all the projects occurred in this period where more than three-quarters of all recorded translocated birds were released. This finding highlights the rapid increase in translocation use and the need for a robust review to identify lessons learnt across all projects.

Kiwi translocations have been mostly used to create secure populations to address the threat of introduced predators (Colbourne & Robertson 2000). However, in the last two decades, there has been a rapid increase of community-led translocation projects driven by ecosystems restoration and advocacy efforts, a trend found across translocations of various species (Cromarty & Alderson 2013; Nally & Adams 2015). Most translocations were for North Island brown kiwi (*Apteryx mantelli*) that recently improved its conservation status (2016 assessment) from a threatened species to a species 'at risk' (Robertson et al. 2017). However, this improvement was attributed mainly to *in situ* management through predator control rather than translocations (Innes et al. 2015b). In contrast, the rarest taxa, such as rowi (*A. rowi*), Haast tokoeka (*A. australis* 'Haast'), and little spotted kiwi (*A. owenii*), had the most translocated birds relative to their overall populations. Translocations have played a vital role in the survival and recovery of these taxa (Seddon et al. 2015; Robertson et al. 2017). For instance, little spotted kiwi became extinct on the mainland by the 1980s and currently only survive in translocated populations sourced from an introduced population to Kapiti Island in 1912 (Ramstad et al. 2021).

Chapter 2 classifies kiwi introductions and reintroductions to medium–long term outcome categories, similar to other translocation reviews of New Zealand birds (Miskelly & Powlesland 2013) or North American fauna (Brichieri-Colombi & Moehrenschlager 2016). However, the kiwi translocation review accounts for the longevity of kiwi species (Heather & Robertson 2015) by setting a longer timeframe for evaluating the translocation outcome (Seddon 2015). The 15-year 'in progress' period is longer than the five years used in the review by Miskelly and Powlesland (2013). Therefore, Chapter 2 categorises more translocations as in progress to avoid early misclassification as successful, which may be interpreted that no further releases or monitoring are needed (Seddon 1999). Additionally, this review addresses the requirement to maintain genetic diversity in kiwi populations (Germano et al. 2018) and thus incorporates the need to establish new populations with the correct minimum number of founders (Weeks et al. 2015). Currently, the official guidance is to release at least 40 unrelated individuals (Sporle 2013; Colbourne et al. 2020). Consequently, several translocations marked earlier as successful by Miskelly and Powlesland (2013) and Colbourne and Robertson (1997) are now classified as translocations requiring further management due to insufficient founder numbers. However, because the 15-year benchmark excludes most kiwi translocations from the outcome assessment and leaves them classified as in progress, the proportion of successful and unsuccessful reflects mostly on historical translocations rather than current practice.

Although it was impossible to evaluate the outcome for most recent translocations, the analysis of post-translocation survival provided insight into possible factors influencing translocation outcome. Findings from this analysis apply to all translocations and can be used in translocation planning. The survival model revealed no substantial differences between survival times of various kiwi taxa. However, it confirmed the impact of predators and reserve size on the survival and retention of released birds. Predation by mustelids, mainly stoats, was reported at most unfenced mainland sites despite ongoing predator control, indicating that the likely cause of previous population decline or extinction was not fully eliminated. Moreover, most unfenced mainland sites were less than 10 000 ha, a minimum area recommended for a self-sustaining kiwi population that would provide sufficient space for dispersal and territory establishment (Basse & McLennan 2003; Westbrooke 2007; Brown et al. 2015). Additionally, the substantially shorter survival times of the birds from the Operation Nest Egg (ONE) programme indicate lower prospects of a successful introduction/reintroduction that would only comprise currently recommended 40 released individuals of the ONE origin. These findings suggest a need for a case-specific number of the released birds based on the type of project area and released birds to address the differences in expected mean survival times.

5.2 Reintroduction of roroa–great spotted kiwi in the Nina Valley

Uncertainty around survival model estimates in Chapter 2 highlights the necessity for robust post-translocation monitoring of released birds to inform future kiwi conservation management. Chapters 3 and 4 demonstrate monitoring methods in a translocation case study: a reintroduction of roroa–great spotted kiwi (*A. maxima*) in the Nina Valley, Lake Sumner Forest Park. Additionally, these chapters explore the response of roroa to management by translocation, both at the source population and the release area. This translocation was the first reintroduction project of the genetically distinct inland population of roroa (Taylor et al. 2021). Therefore, it was essential to monitor the population response to the translocation that would allow comparison to reintroductions of the western roroa population to Lake Rotoiti, Nelson Lakes National Park (Gasson 2005), and the Flora Valley, Kahurangi National Park (Toy & Toy 2020).

Chapter 3 focused on population establishment through post-translocation monitoring in the Nina Valley two years after the release of wild-caught adults. Through ground-based radio telemetry, a common monitoring tool in kiwi management (Colbourne et al. 2020), it was possible to triangulate birds' locations and construct their dispersal paths post-release. It showed highly variable dispersal behaviour among the released birds with a maximum dispersal distance exceeding 10 km. These findings were similar to the dispersal behaviour observed at other roroa reintroductions (Gasson 2005; Toy & Toy 2020). Additionally, the triangulation points allowed an estimation of utilisation distributions that represent the area likely crossed by dispersing individuals during the monitoring duration (Kranstauber et al. 2012). The utilisation distributions reached nearly 1700 ha, and most were found entirely within the Nina Valley. Still, they demonstrate large habitat requirements for kiwi introduction and reintroduction projects. Such large utilisation distributions are consistent with a recommendation of at least 10 000 ha areas for kiwi persistence (Basse & McLennan 2003; Westbrooke 2007; Brown et al. 2015).

The release of the wild-caught adults did not have any measurable impact on the ranging behaviour of the previously released ONE birds. It suggests that the ONE birds, already adults by the time of the wild-caught birds' release, were able to defend their territories. Thus, the developing territorial structure in the Nina Valley remained intact. Chapter 3 also demonstrated the need for sufficiently long monitoring to inform further management of translocated kiwi populations. Only birds that were monitored for two years in the Nina Valley demonstrated stable home ranges, similar to the findings in the Flora Valley, where roroa kept dispersing up to 2.5 years (Toy & Toy 2020).

While Chapter 3 addresses monitoring of a translocated population during the population establishment phase, Chapter 4 explores how acoustic monitoring can be used to demonstrate population growth and the long-term persistence of translocated populations. Additionally, it provides a tool to assess the impacts of a translocation on the source population, which may get negatively impacted by the removal of translocated individuals (Converse & Armstrong 2016). Chapter 4 demonstrates the utility of occupancy analysis (MacKenzie et al. 2018) based on data collected through acoustic monitoring in measuring the population response to kiwi conservation management. By incorporating imperfect detection (Gu & Swihart 2004) based on environmental and biological covariates, it was possible to estimate valid site occupancy as well as occupancy change in time. Unlike radio telemetry studies, which require intrusive capture and handling often leading to behavioural changes in monitored kiwi (Toy & Toy 2021b), acoustics-based occupancy analysis is non-intrusive, cost-effective, and easily scaled up over large areas (Noon et al. 2012). Therefore, this method is suitable for long-term monitoring of translocated populations, as demonstrated in this study.

Occupancy analysis in Chapter 4 showed a significant increase in site occupancy in the Nina Valley following the releases of wild-caught and ONE birds. The acoustic monitoring took place 2–3 years following the release of the wild-caught birds and after the radio telemetry monitoring ended. As expected, the results displayed a positive population response to the releases of roroa in the Nina Valley. Unexpectedly, the occupancy also significantly increased at the source population, the Hawdon Valley in Arthur's Pass National Park, despite removing eight territorial adults for the release in the Nina Valley. At least three out of the four territories, from where the birds were removed, were re-occupied by a new pair within 2.5 years. The results from the Hawdon Valley indicate that a source population can quickly recover if the main invasive predators, such as stoats, are suppressed to a very low level, as in the Hawdon Valley (unpublished data).

5.3 Recommendations

Chapter 2, the kiwi translocation review, outlines a set of recommendations for the management and planning of kiwi translocations based on the pooled information from over 100 translocation projects. It also incorporates advances in genetic management of translocated populations, e.g. Weeks et al. (2015), Biebach et al. (2016), and Frankham et al. (2017), together with recent findings in kiwi genetics (Ramstad et al. 2013; Weiser et al. 2013; Weir et al. 2016; Taylor et al. 2017). Differences in survival times based on habitat characteristics and the type of released birds highlight the need for a specific number of translocated individuals in each project to achieve the desired number of population founders for a genetically viable population (Weeks et al. 2015). Moreover, it advocates for a critical assessment of the project area’s habitat size and quality during translocation planning and subsequent adaptive management (Brown et al. 2015; Stadtmann & Seddon 2020).

Multi-year post-translocation monitoring and management are crucial to achieving translocation objectives, as identified in Chapter 2. Chapters 3 and 4 suggest how to monitor translocated kiwi populations and analyse collected data. Monitoring individuals from consecutive releases provide important information on survival, dispersal, and possible interactions among individual birds and release cohorts. Additionally, the use of diagnostic transmitters, such as in Chapter 3, assists in monitoring the released birds’ breeding behaviour. It also demonstrates the need to follow the birds for at least two years post-release while establishing their territories, a conclusion similar to Toy and Toy (2020). However, Chapter 3 revealed limitations of standard radio telemetry, particularly in identifying dispersal paths of released birds. A better understanding of post-translocation movements would be significantly improved by using GPS tags, currently trialled in South Fiordland tokoeka (*A. australis australis*) (unpublished data).

Long-term monitoring of translocated populations using passive acoustic monitoring should adopt occupancy analysis as shown in Chapter 4. Regular acoustic surveys and subsequent occupancy analysis have the potential to reliably detect even small changes in occupancy of territorial species (Furnas & Callas 2015; Wood et al. 2019). Therefore, this technique is suitable for monitoring response to translocations and other conservation management tools for kiwi species. Chapter 4 provides the baseline data for future repeated monitoring of roroa in the Nina Valley. However, this approach should also be followed at other translocation projects, to identify population trends and distribution changes. Improvements in the identification of individual kiwi by their calls, as demonstrated for roroa (Dent & Molles 2016) and little spotted kiwi (Digby et al. 2014), will further enhance inferences from acoustic monitoring. Additionally, advances in genetic monitoring and genomic assessments (Ramstad & Dunning 2021; Undin et al. 2021) will further inform the adaptive management of translocated populations and improve their prospects for long-term persistence and overall positive translocation outcomes.

References

Abrahams C, Geary M 2020. Combining bioacoustics and occupancy modelling for improved monitoring of rare breeding bird populations. Ecological Indicators 112: 106131.

Adams WM, Sandbrook C 2013. Conservation, evidence and policy. Oryx 47: 329-335.

Armstrong DP, McLean IG 1995. New Zealand translocations: theory and practice. Pacific Conservation Biology 2: 39-54.

Armstrong DP, Seddon PJ 2008. Directions in reintroduction biology. Trends in Ecology & Evolution 23: 20-25.

Armstrong DP, Reynolds MH 2012. Modelling Reintroduced Populations: The State of the Art and Future Directions. In: Ewen JG, Armstrong DP, Parker KA, Seddon PJ eds. Reintroduction Biology: Integrating Science and Management. Oxford, Blackwell Publishing Ltd. Pp. 165-222.

Armstrong DP, Moro D, Hayward MW, Seddon PJ 2015. The development of reintroduction biology in New Zealand and Australia. In: Armstrong DP, Moro D, Hayward MW, Seddon PJ eds. Advances in Reintroduction Biology of Australian and New Zealand Fauna. Clayton South, CSIRO Publishing. Pp. 1-6.

Armstrong DP, Le Coeur C, Thorne JM, Panfylova J, Lovegrove TG, Frost PGH, Ewen JG 2017. Using Bayesian mark-recapture modelling to quantify the strength and duration of post-release effects in reintroduced populations. Biological Conservation 215: 39-45.

Armstrong DP, Seddon PJ, Moehrenschlager A 2019. Reintroduction. In: Fath B ed. Encyclopedia of Ecology. Oxford, Elsevier. Pp. 458-466.

Atkinson IAE 1990. Ecological Restoration on Islands: Prerequisites for Success. In: Towns DR, Daugherty C, Atkinson IAE eds. Ecological restoration of New Zealand islands. Wellington, Department of Conservation. Pp. 73-90.

Bajomi B, Pullin AS, Stewart GB, Takács-Sánta A 2010. Bias and dispersal in the animal reintroduction literature. Oryx 44: 358-365.

Basse B, McLennan JA, Wake GC 1999. Analysis of the impact of stoats, *Mustela erminea*, on northern brown kiwi, *Apteryx mantelli*, in New Zealand. Wildlife Research 26: 227-237.

Basse B, McLennan JA 2003. Protected areas for kiwi in mainland forests of New Zealand: how large should they be? New Zealand Journal of Ecology 27: 95-105.

Batson W, Abbott R, Richardson KM 2015. Release strategies for fauna reintroductions: theory and tests. In: Armstrong DP, Moro D, Hayward MW, Seddon PJ eds. Advances in Reintroduction Biology of Australian and New Zealand Fauna. Clayton South, CSIRO Publishing. Pp. 7-16.

Bellingham PJ, Towns DR, Cameron EK, Davis JJ, Wardle DA, Wilmshurst JM, Mulder CP 2010. New Zealand island restoration: seabirds, predators, and the importance of history. New Zealand Journal of Ecology 34: 115-136.

Berger-Tal O, Blumstein DT, Swaisgood RR 2019. Conservation translocations: a review of common difficulties and promising directions. Animal Conservation 23: 121-131.

Berigan WJ, Jones GM, Whitmore SA, Gutiérrez RJ, Peery MZ 2019. Cryptic wide-ranging movements lead to upwardly biased occupancy in a territorial species. Journal of Applied Ecology 56: 470-480.

Biebach I, Leigh DM, Sluzek K, Keller LF 2016. Genetic Issues in Reintroduction. In: Jachowski DS, Millspaugh JJ, Angermeier PL, Slotow R eds. Reintroduction of Fish and Wildlife Populations. Oakland, University of California Press. Pp. 149-184.

BirdLife International 2016. *Apteryx owenii*. The IUCN Red List of Threatened Species 2016. P. 11.

Blakely TJ, Jellyman PG, Holdaway RJ, Young L, Ben B, Duncan P, Thirkettle D, Simpson J, Ewers RM, Didham RK 2008. The abundance, distribution and structural characteristics of tree-holes in *Nothofagus* forest, New Zealand. Austral Ecology 33: 963-974.

Brichieri-Colombi TA, Moehrenschlager A 2016. Alignment of threat, effort, and perceived success in North American conservation translocations. Conservation Biology 30: 1159-1172.

Briskie JV, Mackintosh M 2004. Hatching failure increases with severity of population bottlenecks in birds. Proceedings of the National Academy of Sciences of the United States of America 101: 558-561.

Brown K, Elliott G, Innes J, Kemp J 2015. Ship rat, stoat and possum control on mainland New Zealand: An overview of techniques, successes and challenges. Department of Conservation, Wellington, P. 40.

Bürkner P-C 2017. brms: An R Package for Bayesian Multilevel Models Using Stan. Journal of Statistical Software 80: 1-28.

Burnham KP, Anderson DR 2002. Model selection and multimodel inference: a practical information-theoretic approach. New York, Springer-Verlag. xxvi + 488 p.

Burns B, Innes J, Day T 2012. The Use and Potential of Pest-Proof Fencing for Ecosystem Restoration and Fauna Conservation in New Zealand. In: Somers MJ, Hayward M eds. Fencing for Conservation: Restriction of Evolutionary Potential or a Riposte to Threatening Processes? New York, NY, Springer New York. Pp. 65-90.

Buxton RT, Jones IL 2012. Measuring nocturnal seabird activity and status using acoustic recording devices: applications for island restoration. Journal of Field Ornithology 83: 47-60.

Calenge C 2006. The package “adehabitat” for the R software: A tool for the analysis of space and habitat use by animals. Ecological Modelling 197: 516-519.

Campos-Cerqueira M, Aide TM 2016. Improving distribution data of threatened species by combining acoustic monitoring and occupancy modelling. Methods in Ecology and Evolution 7: 1340-1348.

Chauvenet ALM, Ewen JG, Armstrong DP, Blackburn TM, Pettorelli N 2013. Maximizing the success of assisted colonizations. Animal Conservation 16: 161-169.

Chauvenet ALM, Parlato EH, Gedir JV, Armstrong DP 2015. Advances in modelling projections for reintroduced populations. In: Armstrong DP, Moro D, Hayward MW, Seddon PJ eds. Advances in Reintroduction Biology of Australian and New Zealand Fauna. Clayton South, CSIRO Publishing. Pp. 91-103.

Colbourne R, Kleinpaste R 1984. North Island Brown Kiwi vocalisations and their use in censusing populations. Notornis 31: 191-201.

Colbourne R, Robertson H 2000. The history of translocations and re-introductions of kiwi in New Zealand. Re-introduction News 19: 47-49.

Colbourne R, Robertson H 2004. Little Spotted Kiwi. In: Brown K ed. Restoring Kapiti: Nature's Second Chance. Dunedin, University of Otago Press. Pp. 53-60.

Colbourne R 2005. Kiwi (*Apteryx* spp.) on offshore New Zealand islands: Populations, translocations and identification of potential release sites. DOC Research & Development Series 208. Department of Conservation, Wellington, P. 24.

Colbourne R, Bassett S, Billing T, McCormick H, McLennan J, Nelson A, Robertson H 2005. The development of Operation Nest Egg as a tool in the conservation management of kiwi. Science for Conservation No. 259. Department of Conservation, Wellington, P. 24.

Colbourne R, Digby A 2016. Call rate behaviour of brown kiwi (*Apteryx mantelli*) and great spotted kiwi (*A. haastii*) in relation to temporal and environmental parameters. DOC Research & Development Series 348. Department of Conservation, Wellington, P. 18.

Colbourne R, Bean E, Coad N, Fuchs R, Graham I, Robertson H, Scrimgeour J 2020. Kiwi Best Practice Manual. Department of Conservation, Wellington, P. 116.

Colbourne RM, Robertson HA 1997. Successful translocations of Little Spotted Kiwi (*Apteryx owenii*) between offshore islands of New Zealand. Notornis 44: 253-258.

Converse SJ, Armstrong DP 2016. Demographic Modeling for Reintroduction Decision-Making. In: Jachowski DS, Millspaugh JJ, Angermeier PL, Slotow R eds. Reintroduction of Fish and Wildlife Populations. Oakland, University of California Press. Pp. 123-146.

Cromarty PL, Alderson SL 2013. Translocation statistics (2002-2010), and the revised Department of Conservation translocation process. Notornis 60: 55-62.

Dent JM, Molles LE 2015. Sexually dimorphic vocalisations of the great spotted kiwi (*Apteryx haastii*). Notornis 62: 1-7.

Dent JM, Molles LE 2016. Call-based identification as a potential tool for monitoring Great Spotted Kiwi. Emu: 315-322.

Department of Conservation 2018. Understanding translocation success: North Island brown kiwi translocation. https://www.doc.govt.nz/get-involved/run-a-project/translocation/translocation-success/ (Accessed 31 August 2021).

Department of Conservation 2019. AR3 Acoustic Recorder. https://ftp.doc.govt.nz/public/folder/CpR1cRv_cE_rqb9ua5WRTg/electronics/Acoustic%20Recorders/Old%20Recorders/AR3%20Recorder%20Instructions.pdf (Accessed 13 August 2021).

Department of Conservation 2020a. Biodiversity in Aotearoa: an overview of state, trends and pressures. Wellington, P. 166.

Department of Conservation 2020b. Returning takahē to the wild: lessons learned. https://blog.doc.govt.nz/2020/07/11/returning-takahe-to-the-wild-lessons-learned/ (Accessed 28 August 2021).

Digby A, Bell BD, Teal PD 2013a. Vocal cooperation between the sexes in Little Spotted Kiwi *Apteryx owenii*. Ibis 155: 229-245.

Digby A, Towsey M, Bell BD, Teal PD 2013b. A practical comparison of manual and autonomous methods for acoustic monitoring. Methods in Ecology and Evolution 4: 675-683.

Digby A, Bell BD, Teal PD 2014. Vocal individuality of Little Spotted Kiwi (*Apteryx owenii*). Emu 114: 326-336.

Dimond WJ, Armstrong DP 2007. Adaptive Harvesting of Source Populations for Translocation: a Case Study with New Zealand Robins. Conservation Biology 21: 114-124.

Fischer J, Lindenmayer DB 2000. An assessment of the published results of animal relocations. Biological Conservation 96: 1-11.

Fiske I, Chandler R 2011. unmarked: An R Package for Fitting Hierarchical Models of Wildlife Occurrence and Abundance. Journal of Statistical Software 43: 1-23.

Frankham R, Ballou JD, Ralls K, Eldridge MDB, Dudash MR, Fenster CB, Lacy RC, Sunnucks P 2017. Genetic Management of Fragmented Animal and Plant Populations. Oxford, Oxford University Press. 426 p.

Frankham R, Ballou JD, Ralls K, Eldridge M, Dudash MR, Fenster CB, Lacy RC, Sunnucks P 2019. A Practical Guide for Genetic Management of Fragmented Animal and Plant Populations. Oxford, Oxford University Press. 208 p.

Franklin MJM, Major RE, Bedward M, Bradstock RA 2020. Establishing the adequacy of recorded acoustic surveys of forest bird assemblages. Avian Conservation and Ecology 15: 8.

Furnas BJ, Callas RL 2015. Using automated recorders and occupancy models to monitor common forest birds across a large geographic region. The Journal of Wildlife Management 79: 325-337.

Furnas BJ, McGrann MC 2018. Using occupancy modeling to monitor dates of peak vocal activity for passerines in California. The Condor 120: 188-200.

Gasson PA 2005. Translocation of great spotted kiwi/roa (*Apteryx haastii*) to Rotoiti Nature Recovery Project. Occasional Publication No. 67. Department of Conservation, Nelson, P. 69.

Germano J, Barlow S, Castro I, Colbourne R, Cox M, Gillies C, Hackwell K, Harawira J, Impey M, Reuben A, Robertson H, Scrimgeour J, Sporle W, Yong S 2018. Kiwi Recovery Plan 2018–2028 Mahere Whakaora Kiwi 2018–2028. Threatened Species Recovery Plan 64. Department of Conservation, Wellington, P. 64.

Germano JM, Field KJ, Griffiths RA, Clulow S, Foster J, Harding G, Swaisgood RR 2015. Mitigation-driven translocations: are we moving wildlife in the right direction? Frontiers in Ecology and the Environment 13: 100-105.

Gillies R, McClellan R 2013. Operation Nest Egg Situation Analysis. Wildlands, Rotorua, P. 43.

Gitzen RA, Keller BJ, Miller MA, Goetz SM, Steen DA, Jachowski DS, Godwin JC, Millspaugh JJ 2016. Effective and Purposeful Monitoring of Species Reintroductions. In: Jachowski DS, Millspaugh JJ, Angermeier PL, Slotow R eds. Reintroduction of Fish and Wildlife Populations. Oakland, University of California Press. Pp. 283-318.

Grant L, Johnson C, Thiessen C 2019. Evaluating the efficacy of translocation: maintaining habitat key to long-term success for an imperiled population of an at-risk species. Biodiversity and Conservation 28: 2727-2743.

Griffith B, Scott JM, Carpenter JW, Reed C 1989. Translocation as a Species Conservation Tool: Status and Strategy. Science 245: 477-480.

Groombridge JJ, Raisin C, Bristol R, Richardson DS 2012. Genetic Consequences of Reintroductions and Insights from Population History. In: Ewen JG, Armstrong DP, Parker KA, Seddon PJ eds. Reintroduction Biology: Integrating Science and Management. Oxford, Blackwell Publishing Ltd. Pp. 395-440.

Gu W, Swihart RK 2004. Absent or undetected? Effects of non-detection of species occurrence on wildlife–habitat models. Biological Conservation 116: 195-203.

Guthrie JD, Byrne ME, Hardin JB, Kochanny CO, Skow KL, Snelgrove RT, Butler MJ, Peterson MJ, Chamberlain MJ, Collier BA 2011. Evaluation of a Global Positioning System backpack transmitter for wild turkey research. The Journal of Wildlife Management 75: 539-547.

Hayward MW, Slotow R 2016. Management of Reintroduced Wildlife Populations. In: Slotow R, Jachowski DS, Millspaugh JJ, Angermeier PL eds. Reintroduction of Fish and Wildlife Populations. Oakland, University of California Press. Pp. 319-340.

Heather BD, Robertson HA 2015. The field guide to the birds of New Zealand. Auckland, Penguin Random House New Zealand. 464 p.

Heber S, Briskie JV 2010. Population Bottlenecks and Increased Hatching Failure in Endangered Birds. Conservation Biology 24: 1674-1678.

Henry R 1895. On Dusky Sound. Transactions of the New Zealand Institute 28: 50-54.

Hill S, Hill J 1987. Richard Henry of Resolution Island. Dunedin, John McIndoe. 364 p.

Hitchmough R, Bull L, Cromarty P 2007. New Zealand Threat Classification System lists 2005. Department of Conservation, Wellington, P. 194.

Hitchmough R 2013. Summary of changes to the conservation status of taxa in the 2008-11 New Zealand Threat Classification System listing cycle. New Zealand Threat Classification Series 1. Department of Conservation, Wellington, P. 20.

Holzapfel SA, Robertson HA, McLennan JA, Sporle W, Hackwell K, Impey M 2008. Kiwi (*Apteryx* spp.) recovery plan: 2008–2018. Threatened Species Recovery Plan 60. Department of Conservation, Wellington, P. 72.

Hulsman J, Wylie M, Yong S 2010. Kiwi - Waimak GSK Translocation Proposal 2010 - 2013: Juveniles to the Nina. Department of Conservation, Rangiora, P. 23.

Innes J, Kelly D, Overton JM, Gillies C 2010. Predation and other factors currently limiting New Zealand forest birds. New Zealand Journal of Ecology 34: 86-114.

Innes J, Burns B, Sanders A, Hayward MW 2015a. The impact of private sanctuary networks on reintroduction programs. In: Armstrong DP, Moro D, Hayward MW, Seddon PJ eds. Advances in Reintroduction Biology of Australian and New Zealand Fauna. Clayton South, CSIRO Publishing. Pp. 185-199.

Innes J, Eppink FV, Robertson H 2015b. Saving a national icon: Preliminary estimation of the additional cost of achieving kiwi population stability or 2% growth. Landcare Research, Hamilton, P. 57.

Innes J, Eppink F, Anderson D, Robertson H 2016. Roles for 'kōhanga' in kiwi conservation – a review. Landcare Research, Hamilton, P. 69.

Innes J, Fitzgerald N, Binny R, Byrom A, Pech R, Watts C, Gillies C, Maitland M, Campbell-Hunt C, Burns B 2019. New Zealand ecosanctuaries: types, attributes and outcomes. Journal of the Royal Society of New Zealand 49: 370-393.

IUCN/SSC 2013. Guidelines for Reintroductions and Other Conservation Translocations: version 1.0. Gland, Switzerland, IUCN Species Survival Commission. viii + 57 p.

Jachowski DS, Slotow R, Angermeier PL, Millspaugh JJ 2016. Animal Reintroduction in the Anthropocene. In: Jachowski DS, Slotow R, Angermeier PL, Millspaugh JJ eds. Reintroduction of Fish and Wildlife Populations. Oakland, University of California Press. Pp. 1-4.

Jahn P, Harper GA, Gilchrist J 2013. Home range sharing in family units of great spotted kiwi (*Apteryx haastii*) at Nelson Lakes National Park. Notornis 60: 201-209.

Jahn P, Cagua EF, Molles LE, Ross JG, Germano JM 2022a. Kiwi translocation review: are we releasing enough birds and to the right places? New Zealand Journal of Ecology 46: 3454.

Jahn P, Ross JG, MacKenzie DI, Molles LE 2022b. Acoustic monitoring and occupancy analysis: cost-effective tools in reintroduction programmes for roroa–great spotted kiwi. New Zealand Journal of Ecology 46: 3466.

Jamieson IG, Lacy RC 2012. Managing Genetic Issues in Reintroduction Biology. In: Ewen JG, Armstrong DP, Parker KA, Seddon PJ eds. Reintroduction Biology: Integrating Science and Management. Oxford, Blackwell Publishing Ltd. Pp. 441-475.

Jolly JN, Colbourne RM 1991. Translocations of the little spotted kiwi (*Apteryx owenii*) between offshore islands of New Zealand. Journal of the Royal Society of New Zealand 21: 143-149.

Jolly JN, Daugherty CH 2002. Comparison of little spotted kiwi (*Apteryx owenii*) from Kapiti and D'Urville Islands. In: Overmars F ed. Some early 1990s studies in kiwi (*Apteryx* spp.) genetics and management. Wellington, Department of Conservation. Pp. 57-65.

Jones CG, Merton DV 2012. A Tale of Two Islands: The Rescue and Recovery of Endemic Birds in New Zealand and Mauritius. In: Ewen JG, Armstrong DP, Parker KA, Seddon PJ eds. Reintroduction Biology: Integrating Science and Management. Oxford, Blackwell Publishing Ltd. Pp. 33-72.

Juodakis J, Castro I, Marsland S 2021. Precision as a metric for acoustic survey design using occupancy or spatial capture-recapture. Environmental and Ecological Statistics 28: 587-608.

Kahle D, Wickham H 2013. ggmap: Spatial Visualization with ggplot2. The R Journal 5: 144-161.

Kenward RE 2001. A Manual for Wildlife Radio Tagging. San Diego, Academic Press. 311 p.

Keye C, Roschak C, Ross J 2011. Summer home range size and population density of great spotted kiwi (*Apteryx haastii*) in the North Branch of the Hurunui River, New Zealand. Notornis 58: 22-30.

Kranstauber B, Kays R, LaPoint SD, Wikelski M, Safi K 2012. A dynamic Brownian bridge movement model to estimate utilization distributions for heterogeneous animal movement. Journal of Animal Ecology 81: 738-746.

Kranstauber B, Smolla M, Scharf AK 2020. move: Visualizing and Analyzing Animal Track Data.

MacKenzie DI, Nichols JD, Lachman GB, Droege S, Andrew Royle J, Langtimm CA 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83: 2248-2255.

MacKenzie DI, Nichols JD, Hines JE, Knutson MG, Franklin AB 2003. Estimating site occupancy, colonization, and local extinction when a species is detected imperfectly. Ecology 84: 2200-2207.

MacKenzie DI, Bailey LL, Nichols JD 2004. Investigating species co-occurrence patterns when species are detected imperfectly. Journal of Animal Ecology 73: 546-555.

MacKenzie DI, Nichols JD, Sutton N, Kawanishi K, Bailey LL 2005. Improving inferences in population studies of rare species that are detected imperfectly. Ecology 86: 1101-1113.

MacKenzie DI, Nichols JD, Royle JA, Pollock KH, Bailey LL, Hines JE 2018. Occupancy Estimation and Modeling: Inferring Patterns and Dynamics of Species Occurrence. Boston, Academic Press. 648 p.

MacMillan BWH 1990. Attempts to re-establish Wekas, Brown Kiwis and Red-Crowned Parakeets in the Waitakere Ranges. Notornis 37: 45-51.

Mander V 2016. How translocation affects the activity, home range and pair-bonds of great spotted kiwi (*Apteryx haastii*). Lincoln University, Lincoln, P. 101.

Marsh DM, Trenham PC 2008. Current Trends in Plant and Animal Population Monitoring. Conservation Biology 22: 647-655.

McCarthy MA, Armstrong DP, Runge MC 2012. Adaptive Management of Reintroduction. In: Ewen JG, Armstrong DP, Parker KA, Seddon PJ eds. Reintroduction Biology: Integrating Science and Management. Oxford, Blackwell Publishing Ltd. Pp. 256-289.

McHalick O 1998. Translocation database summary. Threatened Species Occasional Publication No. 14. Department of Conservation, Wellington, P. 70.

McLennan J, McCann T 2002. Genetic variability, distribution and abundance of great spotted kiwi (*Apteryx haastii*). In: Overmars F ed. Some early 1990s studies in kiwi (*Apteryx* spp.) genetics and management. Wellington, Department of Conservation. Pp. 35-56.

McLennan JA, McCann AJ 1991. Ecology of great spotted kiwi, *Apteryx haastii*. Department of Scientific and Industrial Research, Havelock North, P. 38.

McLennan JA, Potter MA 1992. Distribution, Population-Changes and Management of Brown Kiwi in Hawkes Bay. New Zealand Journal of Ecology 16: 91-102.

McLennan JA, McCann TJ 1993. Claw colour in Great Spotted Kiwi. Notornis 40: 76-78.

McLennan JA, Potter MA, Robertson HA, Wake GC, Colbourne R, Dew L, Joyce L, McCann AJ, Miles J, Miller PJ, Reid J 1996. Role of predation in the decline of kiwi, *Apteryx* spp., in New Zealand. New Zealand Journal of Ecology 20: 27-35.

Metcalf OC, Ewen JG, McCready M, Williams EM, Rowcliffe JM 2019. A novel method for using ecoacoustics to monitor post-translocation behaviour in an endangered passerine. Methods in Ecology and Evolution 10: 626-636.

Miller DA, Nichols JD, McClintock BT, Grant EHC, Bailey LL, Weir LA 2011. Improving occupancy estimation when two types of observational error occur: non-detection and species misidentification. Ecology 92: 1422-1428.

Miller KA, Bell TP, Germano JM 2014. Understanding Publication Bias in Reintroduction Biology by Assessing Translocations of New Zealand's Herpetofauna. Conservation Biology 28: 1045-1056.

Millspaugh JJ, Marzluff JM 2001. Radio Tracking and Animal Populations. San Diego, Academic Press. 474 p.

Miskelly CM, Dowding JE, Elliott GP, Hitchmough RA, Powlesland RG, Robertson HA, Sagar PM, Scofield RP, Taylor GA 2008. Conservation status of New Zealand birds, 2008. Notornis 55: 117-135.

Miskelly CM 2009. Ecological restoration and threatened species management in New Zealand. Ecological Management & Restoration 10: 160-161.

Miskelly CM, Powlesland RG 2013. Conservation translocations of New Zealand birds, 1863-2012. Notornis 60: 3-28.

Moro D, Hayward MW, Seddon PJ, Armstrong DP 2015. Reintroduction biology of Australian and New Zealand fauna: progress, emerging themes and future directions. In: Armstrong DP, Moro D, Hayward MW, Seddon PJ eds. Advances in Reintroduction Biology of Australian and New Zealand Fauna. Clayton South, CSIRO Publishing. Pp. 285-290.

Morrison K, Yong S 2014. Kiwi - Waimak GSK Translocation Proposal 2014. Department of Conservation, Rangiora, P. 52.

Nally S, Adams L 2015. Evolution of the translocation approval process in Australia and New Zealand. In: Armstrong DP, Moro D, Hayward MW, Seddon PJ eds. Advances in Reintroduction Biology of Australian and New Zealand Fauna. Clayton South, CSIRO Publishing. Pp. 273-284.

Neill E, Jansen P 2014. Ground-based radio tracking: a best practice protocol. Department of Conservation, Wellington, P. 23.

Nelson NJ, Briskie JV, Constantine R, Monks J, Wallis GP, Watts C, Wotton DM 2019. The winners: species that have benefited from 30 years of conservation action. Journal of the Royal Society of New Zealand 49: 281-300.

Nichols JD, Armstrong DP 2012. Monitoring for Reintroductions. In: Ewen JG, Armstrong DP, Parker KA, Seddon PJ eds. Reintroduction Biology:

Integrating Science and Management. Oxford, Blackwell Publishing Ltd. Pp. 223-255.

Noon BR, Bailey LL, Sisk TD, McKelvey KS 2012. Efficient Species-Level Monitoring at the Landscape Scale. Conservation Biology 26: 432-441.

Odom KJ, Slaght JC, Gutiérrez RJ 2013. Distinctiveness in the Territorial Calls of Great Horned Owls within and among Years. Journal of Raptor Research 47: 21-30.

Oliver WRB 1955. New Zealand birds. Wellington, N.Z., A. H. & A. W. Reed.

Pacioni C, Trocini S, Wayne AF, Rafferty C, Page M 2020. Integrating population genetics in an adaptive management framework to inform management strategies. Biodiversity and Conservation 29: 947-966.

Palma RL 1991. A new species of *Rallicola* (Insecta: Phthiraptera : Philopteridae) from the North Island brown kiwi. Journal of the Royal Society of New Zealand 21: 313-322.

Parker KA 2008. Translocations: Providing Outcomes for Wildlife, Resource Managers, Scientists, and the Human Community. Restoration Ecology 16: 204-209.

Parker KA, Ewen JG, Seddon PJ, Armstrong DP 2013. Post-release monitoring of bird translocations: why is it important and how do we do it? Notornis 60: 85-92.

R Core Team 2020. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Ramstad KM, Colbourne RM, Robertson HA, Allendorf FW, Daugherty CH 2013. Genetic consequences of a century of protection: serial founder events and survival of the little spotted kiwi (*Apteryx owenii*). Proceedings of the Royal Society B: Biological Sciences 280: 20130576.

Ramstad KM, Dunning LT 2021. Population Genomics Advances and Opportunities in Conservation of Kiwi (*Apteryx* spp.). In: Hohenlohe PA, Rajora OP eds. Population Genomics: Wildlife. Cham, Springer International Publishing. Pp. 493-521.

Ramstad KM, Gibb GC, Robertson HA, Colbourne RM, Doran EE, Shepherd LD 2021. Recent extinctions among Little Spotted Kiwi (*Apteryx owenii*) and the origin of extant populations. Emu - Austral Ornithology: 23-32.

Read AF, O'Donnell CFJ 1987. Abundance of Yellowheads in the Hawdon River Valley, Arthur's Pass National Park, in 1983 and 1984. Notornis 34: 307-315.

Richardson KM, Doerr V, Ebrahimi M, Lovegrove TG, Parker KA 2015. Considering dispersal in reintroduction and restoration planning. In: Armstrong DP, Moro D, Hayward MW, Seddon PJ eds. Advances in Reintroduction Biology of Australian and New Zealand Fauna. Clayton South, CSIRO Publishing. Pp. 59-72.

Robert A, Colas B, Guigon I, Kerbiriou C, Mihoub JB, Saint-Jalme M, Sarrazin F 2015. Defining reintroduction success using IUCN criteria for threatened species: a demographic assessment. Animal Conservation 18: 397-406.

Robertson H, Colbourne R 2017. Kiwi Best Practice Manual. Department of Conservation, Wellington, P. 113.

Robertson HA, McLennan JA, Colbourne RM, McCann TJ 2005. Population status of great spotted kiwi (*Apteryx haastii*) near Saxon Hut, Heaphy Track, New Zealand. Notornis 52: 27-33.

Robertson HA, Colbourne RM, Graham PJ, Miller PJ, Pierce RJ 2011. Experimental management of Brown Kiwi *Apteryx mantelli* in central Northland, New Zealand. Bird Conservation International 21: 207-220.

Robertson HA, de Monchy PJM 2012. Varied success from the landscape-scale management of kiwi *Apteryx* spp. in five sanctuaries in New Zealand. Bird Conservation International 22: 429-444.

Robertson HA, Dowding JE, Elliott GP, Hitchmough RA, Miskelly CM, O'Donnell CFJ, Powlesland RG, Sagar PM, Scofield RP, Taylor GA 2013. Conservation status of New Zealand birds, 2012. New Zealand Threat Classification Series. Department of Conservation, Wellington, P. 26.

Robertson HA, Baird K, Dowding JE, Elliott GP, Hitchmough RA, Miskelly CM, McArthur N, O'Donnell CFJ, Sagar PM, Scofield RP, Taylor GA 2017. Conservation status of New Zealand birds, 2016. New Zealand Threat Classification Series. Department of Conservation, Wellington, P. 27.

Robertson HA, Coad NB, Colbourne RM, Fraser JR 2019a. Status of little spotted kiwi (*Apteryx owenii*) on Red Mercury Island (Whakau) in March 2016. DOC Research and Development Series 361. Department of Conservation, Wellington, P. 9.

Robertson HA, Coad NB, Colbourne RM, Fraser JR 2019b. Translocations of little spotted kiwi (*Apteryx owenii*) for genetic management, 2016–17. DOC Research and Development Series 358. Department of Conservation, Wellington, P. 13.

Robertson HA, Coad NB, Colbourne RM, Fraser JR 2019c. Status of little spotted kiwi (*Apteryx owenii*) on Tiritiri Matangi Island in April 2017. DOC Research and Development Series 360. Department of Conservation, Wellington, P. 8.

Robertson HA, Baird KA, Elliott GP, Hitchmough RA, McArthur NJ, Makan T, Miskelly CM, O'Donnell CJ, Sagar PM, Scofield RP, Taylor GA, Michel P 2021. Conservation status of birds in Aotearoa New Zealand, 2021. New Zealand Threat Classification Series. Department of Conservation, Wellington, P. 43.

Robinson NM, Scheele BC, Legge S, Southwell DM, Carter O, Lintermans M, Radford JQ, Skroblin A, Dickman CR, Koleck J, Wayne AF, Kanowski J, Gillespie GR, Lindenmayer DB 2018. How to ensure threatened species monitoring leads to threatened species conservation. Ecological Management & Restoration 19: 222-229.

Roroa Practitioner Group 2021. Roroa / great spotted kiwi (*Apteryx maxima*) species plan 2019–2029. Department of Conservation, Nelson, P. 46.

Rowlingson B, Diggle P 2021. splancs: Spatial and Space-Time Point Pattern Analysis.

Royle JA, Nichols JD 2003. Estimating abundance from repeated presence-absence data or point counts. Ecology 84: 777-790.

Sarrazin F 2007. Introductory remarks: A demographic frame for reintroductions. Écoscience 14: iv-v.

Saunders A 1995. Translocations in New Zealand: an overview. In: Serena M ed. Reintroduction biology of Australian and New Zealand fauna. Chipping Norton, NSW, Surrey Beatty and Sons. Pp. 43-46.

Saunders A, Norton DA 2001. Ecological restoration at Mainland Islands in New Zealand. Biological Conservation 99: 109-119.

Seddon PJ 1999. Persistence without intervention: assessing success in wildlife reintroductions. Trends in Ecology & Evolution 14: 503.

Seddon PJ 2010. From Reintroduction to Assisted Colonization: Moving along the Conservation Translocation Spectrum. Restoration Ecology 18: 796-802.

Seddon PJ, Roughton CM, Reardon J, MacKenzie DI 2011. Dynamics of an endangered New Zealand skink: accounting for incomplete detectability in estimating patch occupancy. New Zealand Journal of Ecology 35: 247-253.

Seddon PJ, Strauss WM, Innes J 2012. Animal Translocations: What are they and why do we do them? In: Ewen JG, Armstrong DP, Parker KA, Seddon PJ eds. Reintroduction Biology: Integrating Science and Management. Oxford, Blackwell Publishing Ltd. Pp. 1-32.

Seddon PJ, Griffiths CJ, Soorae PS, Armstrong DP 2014. Reversing defaunation: Restoring species in a changing world. Science 345: 406-412.

Seddon PJ 2015. Using the IUCN Red List criteria to assess reintroduction success. Animal Conservation 18: 407-408.

Seddon PJ, Moro D, Mitchell NJ, Chauvenet ALM, Mawson PR 2015. Proactive conservation or planned invasions? Past, current and future use of assisted colonisation in Australasia. In: Armstrong DP, Moro D, Hayward MW, Seddon PJ eds. Advances in Reintroduction Biology of Australian and New Zealand Fauna. Clayton South, CSIRO Publishing. Pp. 105-125.

Sheean VA, Manning AD, Lindenmayer DB 2012. An assessment of scientific approaches towards species relocations in Australia. Austral Ecology 37: 204-215.

Shepherd LD, Worthy TH, Tennyson AJD, Scofield RP, Ramstad KM, Lambert DM 2012. Ancient DNA Analyses Reveal Contrasting Phylogeographic Patterns amongst Kiwi (*Apteryx* spp.) and a Recently Extinct Lineage of Spotted Kiwi. PLoS ONE 7: e42384.

Shepherd LD, Tennyson AJD, Robertson HA, Colbourne RM, Ramstad KM 2021. Hybridisation in kiwi (*Apteryx*; Apterygidae) requires taxonomic revision for the Great Spotted Kiwi. Avian Research 12: 24.

Shonfield J, Bayne EM 2017. Autonomous recording units in avian ecological research: current use and future applications. Avian Conservation and Ecology 12: 14.

Smuts-Kennedy C, Parker KA 2013. Reconstructing avian biodiversity on Maungatautari. Notornis 60: 93-106.

Sporle W 2013. A guide to establishing a new kiwi population. Kiwis for kiwi, Auckland, P. 7.

Stadtmann S, Seddon PJ 2020. Release site selection: reintroductions and the habitat concept. Oryx 54: 687-695.

Stewart P, Hasenbank M 2018. North Island brown kiwi (*Apteryx mantelli*) monitoring at Whenuakite: Trend comparison of observer and acoustic recorder collected call counts. Notornis 65: 179-186.

Stiffler LL, Anderson JT, Katzner TE 2018. Occupancy Modeling of Autonomously Recorded Vocalizations to Predict Distribution of Rallids in Tidal Wetlands. Wetlands 38: 605-612.

Sutherland WJ, Pullin AS, Dolman PM, Knight TM 2004. The need for evidence-based conservation. Trends in Ecology & Evolution 19: 305-308.

Sutherland WJ, Armstrong D, Butchart SHM, Earnhardt JM, Ewen J, Jamieson I, Jones CG, Lee R, Newbery P, Nichols JD, Parker KA, Sarrazin F, Seddon PJ, Shah N, Tatayah V 2010. Standards for documenting and monitoring bird reintroduction projects. Conservation Letters 3: 229-235.

Sutherland WJ, Wordley CFR 2017. Evidence complacency hampers conservation. Nature Ecology & Evolution 1: 1215-1216.

Tavecchia G, Viedma C, Martínez-Abraín A, Bartolomé M-A, Gómez JA, Oro D 2009. Maximizing re-introduction success: Assessing the immediate cost of release in a threatened waterfowl. Biological Conservation 142: 3005-3012.

Taylor HR, Colbourne RM, Robertson HA, Nelson NJ, Allendorf FW, Ramstad KM 2017. Cryptic inbreeding depression in a growing population of a long-lived species. Molecular Ecology 26: 799-813.

Taylor HR, Robertson H, Carter AL, Ramstad KM 2021. The conservation management implications of isolation by distance and high genetic diversity in Great Spotted Kiwi (*Apteryx haastii*). Emu - Austral Ornithology 121: 10-22.

Teixeira D, Maron M, van Rensburg BJ 2019. Bioacoustic monitoring of animal vocal behavior for conservation. Conservation Science and Practice 1: e72.

Tempel DJ, Gutiérrez RJ 2013. Relation between Occupancy and Abundance for a Territorial Species, the California Spotted Owl. Conservation Biology 27: 1087-1095.

Tingley MW, Wilkerson RL, Howell CA, Siegel RB 2016. An integrated occupancy and space-use model to predict abundance of imperfectly detected, territorial vertebrates. Methods in Ecology and Evolution 7: 508-517.

Toy R, Toy S 2020. Post-translocation dispersal and home range establishment of roroa (great spotted kiwi, *Apteryx haastii*): need for long-term monitoring and a flexible management strategy. Notornis 67: 511-525.

Toy R, Toy S 2021a. Breeding ecology of a translocated population of great spotted kiwi (*Apteryx haastii*). Notornis 68: 131-146.

Toy R, Toy S 2021b. Changes in behaviour of great spotted kiwi (*Apteryx haastii*) following handling. Notornis 68: 173-176.

Turgeon PJ, Van Wilgenburg SL, Drake KL 2017. Microphone variability and degradation: implications for monitoring programs employing autonomous recording units. Avian Conservation and Ecology 12: 9.

Undin M, Hills SFK, Lockhart PJ, Castro I 2021. Gaps in genetic knowledge affect conservation management of kiwi (*Apteryx*) species. Ibis 163: 1155-1174.

Weeks AR, Sgro CM, Young AG, Frankham R, Mitchell NJ, Miller KA, Byrne M, Coates DJ, Eldridge MDB, Sunnucks P, Breed MF, James EA, Hoffmann AA

2011. Assessing the benefits and risks of translocations in changing environments: a genetic perspective. Evolutionary Applications 4: 709-725.

Weeks AR, Moro D, Thavornkanlapachai R, Taylor HR, White NE, Weiser EL, Heinze D 2015. Conserving and enhancing genetic diversity in translocation programs. In: Armstrong DP, Moro D, Hayward MW, Seddon PJ eds. Advances in Reintroduction Biology of Australian and New Zealand Fauna. Clayton South, CSIRO Publishing. Pp. 127-140.

Weir JT, Haddrath O, Robertson HA, Colbourne RM, Baker AJ 2016. Explosive ice age diversification of kiwi. Proceedings of the National Academy of Sciences 113: E5580-E5587.

Weiser EL, Grueber CE, Jamieson IG 2013. Simulating Retention of Rare Alleles in Small Populations to Assess Management Options for Species with Different Life Histories. Conservation Biology 27: 335-344.

Weiser EL 2014. Informing genetic management of small populations of threatened species. University of Otago, Dunedin, P. 177.

Westbrooke I 2007. How large a managed area is needed to protect a threatened animal species? Combining simple dispersal and population models. New Zealand Journal of Ecology 31: 154-159.

Wildlife Acoustics 2019. Kaleidoscope Pro Analysis Software.

Willacy RJ, Mahony M, Newell DA 2015. If a frog calls in the forest: Bioacoustic monitoring reveals the breeding phenology of the endangered Richmond Range mountain frog (*Philoria richmondensis*). Austral Ecology 40: 625-633.

Wilson K-J 2004. Flight of the Huia: Ecology and conservation of New Zealand's frogs, reptiles, birds and mammals. Christchurch, Canterbury University Press. 411 p.

Wolf CM, Griffith B, Reed C, Temple SA 1996. Avian and Mammalian Translocations: Update and Reanalysis of 1987 Survey Data. Conservation Biology 10: 1142-1154.

Wolf MC, Garland T, Griffith B 1998. Predictors of avian and mammalian translocation success: reanalysis with phylogenetically independent contrasts. Biological Conservation 86: 243-255.

Wood CM, Popescu VD, Klinck H, Keane JJ, Gutiérrez RJ, Sawyer SC, Peery MZ 2019. Detecting small changes in populations at landscape scales: a bioacoustic site-occupancy framework. Ecological Indicators 98: 492-507.

Wood CM, Klinck H, Gustafson M, Keane JJ, Sawyer SC, Gutiérrez RJ, Peery MZ 2021. Using the ecological significance of animal vocalizations to improve inference in acoustic monitoring programs. Conservation Biology 35: 336–345.

Appendix 1

List of kiwi translocations projects

List of recorded kiwi translocations and their outcomes. Even though reinforcements and translocations resulting in hybrid populations were not assessed for their outcome, they appear in outcome column as either reinforcement or hybrid. Information about numbers of released birds from several translocations before 1990 may be incomplete or missing (denoted as —).

taxon	release site	year	outcome	released
Northland brown kiwi	Kawau I, Hauraki Gulf	1863–1864	mgmt required	—
	Rangitoto Range, Waikato	1973–1977	unsuccessful	50
	Motuarohia I (Roberton I), Bay of Islands	1975–1976	mgmt required	5
	Matuku Reserve, N Waitākere Ranges	1979–1985	unsuccessful	41
	Huia Bay, S Waitākere Ranges	1981–1982	unsuccessful	3
	Moturoa I, Bay of Islands	1981–1985	mgmt required	7
	Burma Rd Reserve, Tapora	1982–1986	unsuccessful	32
	Moturua I, Bay of Islands	1983–1985	mgmt required	15
	Motukiekie I, Bay of Islands	1984–1984	unsuccessful	—
	Logue’s bush, Wellsford, Auckland	1988–1988	unsuccessful	1
	Motukawanui I, Cavalli Is	1995–1997	in progress	12
	Motuora I, Hauraki Gulf	1999–2015	in progress	92
	Bream Head	2000–2016	reinforcement	74
	Whangārei Heads (Manaia, Kaiamba)	2004–2016	reinforcement	114
	Tutukaka/Ngunguru	2005–2018	reinforcement	50
	Tāwharanui Open Sanctuary	2006–2008	in progress	44
	Tanekaha, Hikurangi	2011–2018	in progress	19

	Motukawanui I, Cavalli Is	2013–2013	reinforcement	5
	Marunui, Brynderwyn Range	2013–2015	in progress	45
	Mataia, Glorit	2013–2015	in progress	41
	Pataua North, Ngunguru Bay	2017–2018	in progress	23
	Pukenui Forest, Whangārei	2018–2018	in progress	12
Coromandel brown kiwi	Motutapu I, Hauraki Gulf	2012–2018	in progress	79
	Whangapoua Forest, Coromandel	2014–2018	reinforcement	57
	Kohukohunui, Hunua Ranges	2017–2017	unsuccessful	7
Western brown kiwi	Te Hauturu-o-Toi / Little Barrier I	1903–1919	reinforcement	19
	Kapiti I	1915–1940	hybrid	10
	Karioi Rahui, Rangataua Forest	2000–2015	reinforcement	59
	Maungatautari	2005–2018	in progress	89
	Mount Taranaki	2005–2018	reinforcement	106
	Purangi	2008–2008	reinforcement	2
	Tongariro Forest	2010–2014	reinforcement	23
	Lake Rotokare, Taranaki	2010–2018	in progress	33
	Waimarino Forest	2012–2014	reinforcement	20
	Parininihi	2015–2015	reinforcement	1
Eastern brown kiwi	Mokoia I, Lake Rotorua	1974–1974	unsuccessful	2
	Boundary Stream Mainland Island	2000–2007	in progress	40
	Moutohorā I (Whale I), Bay of Plenty	2001–2013	in progress	29
	Mokoia I, Lake Rotorua	2003–2008	in progress	9
	Kāweka Forest	2004–2018	reinforcement	29
	Tuhua I (Mayor I), Bay of Plenty	2006–2011	in progress	13
	Otanewainuku, Bay of Plenty	2007–2018	in progress	50
	Cape (Kidnappers) Sanctuary, Hawke's Bay	2008–2011	in progress	74
	Maungataniwha, Hawke's Bay	2011–2017	reinforcement	37

	Whareama Peninsula, Lake Waikaremoana	2014–2014	reinforcement	3
Mixed origin brown kiwi	Ponui I, Hauraki Gulf	1964–1964	mgmt required	14
	Gwavas Forest, Hawke's Bay	1980–1981	unsuccessful	16
	Pūkaha Mount Bruce	2003–2014	in progress	60
	Remutaka Forest	2006–2012	in progress	32
Little spotted kiwi	Long I, Dusky Sound	1896–1897	unsuccessful	16
	Resolution I, Dusky Sound	1896–1898	unsuccessful	50
	Anchor I, Dusky Sound	1898–1898	unsuccessful	3
	Cooper I, Dusky Sound	1903–1903	reinforcement	7
	Kapiti I	1912–1912	mgmt required	5
	Maud I / Te Hoiere, Pelorus Sound	1980–1980	unsuccessful	2
	Long I, Queen Charlotte Sound	1982–1989	mgmt required	5
	Red Mercury I (Whakau), Coromandel	1983–1983	in progress	12
	Taranga I (Hen I), Hauraki Gulf	1988–1989	mgmt required	38
	Tiritiri Matangi I	1993–1995	in progress	16
	Zealandia Sanctuary	2000–2008	likely successful	41
	Chalky I, Chalky Inlet	2008–2010	in progress	39
	Motuihe I, Hauraki Gulf	2009–2010	in progress	40
	Cape (Kidnappers) Sanctuary, Hawke's Bay	2015–2016	in progress	31
	Anchor I, Dusky Sound	2015–2017	in progress	40
	Red Mercury I (Whakau), Coromandel	2016–2016	reinforcement	10
	Tiritiri Matangi I	2017–2017	reinforcement	10
	Shakespear Open Sanctuary	2017–2018	in progress	40
LSK/Rowi hybrid	Mana I, Wellington	1992–1994	unsuccessful	2
	Allports I, Queen Charlotte Sound	2006–2006	hybrid	2
Rowi	Blumine I, Queen Charlotte Sound	2010–2016	in progress	31
	North Ōkārito Forest	2010–2017	in progress	169

	Mana I, Wellington	2012–2013	in progress	24
	Ōmoeroa Ranges	2018–2018	in progress	27
Roroa–Great spotted kiwi	Te Hauturu-o-Toi / Little Barrier I	1915–1915	unsuccessful	19
	Lake Rotoiti Mainland Island	2004–2016	in progress	31
	Hawdon/Andrews Valley, Arthur's Pass	2009–2010	reinforcement	4
	Flora, Kahurangi	2010–2016	in progress	44
	Nina Valley, Lewis Pass	2011–2015	in progress	18
Haast tokoeka	Rarotoka I (Centre I), Foveaux Strait	2008–2015	in progress	20
	Coal I, Preservation Inlet	2009–2015	in progress	33
	Orokonui Sanctuary, Dunedin	2010–2016	in progress	26
	Pomona I, Lake Manapouri	2011–2015	in progress	22
North Fiordland tokoeka	Te Hauturu-o-Toi / Little Barrier I	1903–1903	unsuccessful	1
	Doubtful I 1, Lake Te Anau	2002–2002	unsuccessful	4
	Doubtful I 2, Lake Te Anau	2002–2002	unsuccessful	7
	Te Ana-au Caves, Murchison Mtns	2004–2004	reinforcement	4
	Sinbad Gully, Milford Sound / Piopiotahi	2011–2011	reinforcement	3
South Fiordland tokoeka	Indian I, Dusky Sound	1895–1897	mgmt required	5
	Resolution I, Dusky Sound	1895–1898	reinforcement	15
	Long I, Dusky Sound	1896–1897	mgmt required	13
	Parrot I, Dusky Sound	1896–1897	mgmt required	7
	Harbour I, Breaksea Sound	1897–1897	unsuccessful	3
	Anchor I, Dusky Sound	1897–1900	unsuccessful	2
	Breaksea I, Breaksea Sound	1900–1900	unsuccessful	3
	Maori I, Dusky Sound	1900–1900	unsuccessful	—
	Nomans I, Dusky Sound	1900–1900	unsuccessful	—
	Prove I, Dusky Sound	1900–1900	unsuccessful	—
	Kapiti I	1908–1908	hybrid	2

Rakiura tokoeka	Horomamae / Owen I, Rakiura	1955–1955	mgmt required	—
	Ulva I, Rakiura	1980–1980	in progress	7
	Ulva I, Rakiura	2013–2013	reinforcement	10
	Ackers Point, Halfmoon Bay, Rakiura I	2013–2013	reinforcement	11