MAKING BETTER DECISIONS FOR THE CONSERVATION OF THREATENED SPECIES: SCIENCE, VALUES, AND NEW ZEALAND'S RAREST BIRD

A thesis submitted in fulfilment of
the requirements for the degree of
Doctor of Philosophy (PhD)

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May 2021

Declaration

I, Thalassa McMurdo Hamilton, confirm that the work presented in this thesis is my own. Where information has been derived from other sources, I confirm that this has been identified in the thesis.

Thesis abstract

Decision making for threatened species recovery can be complex: there is often a diverse range of stakeholders holding values that may be conflicting, data are typically deficient and imperfect, and there is uncertainty about the outcomes of proposed actions. Yet in this pressured and challenging context, decisions must still be made. Conservationists therefore need the right tools to address these complexities, and structured decision making (SDM) is effective in this space. Here, I demonstrate the utility of SDM and its component tools to assist recovery planning for Aotearoa-New Zealand's rarest indigenous breeding bird, tara iti (New Zealand fairy tern, *Sternula nereis davisae*). My PhD aims to advance (i) the way we approach decisions via inclusivity and expression of values, (ii) the way we make decisions by recognising objectives, creating alternatives and making explicit trade-offs, and (iii) the way we use data to support these decisions by analysing and interpreting biased or imperfect datasets.

Values drive decisions, and I first demonstrate how SDM, a values-focused approach, can be used to meaningfully integrate stakeholder values such as mātauranga Māori (Māori [indigenous New Zealander] knowledge/perspective) into conservation decisions and provide a basis for co-management between different peoples. Second, I analyse a seabird translocation trial, showing how creative thinking about alternatives can help better achieve conservation objectives. Third, I show how the application of SDM resulted in a new management recommendation that balanced across multiple objectives and was evidence-based. This was the first action after a decade of inaction and communication breakdown between stakeholders. Finally, I use a decision tree and counterfactuals to analyse the efficacy of tara iti egg management, showing how these tools can help navigate complex and biased monitoring data sets to improve future decisions. This thesis provides a detailed real-world example of how SDM can be applied effectively to a complex conservation problem, and highlights the importance of clear, values-focused thinking and inclusive approaches in species recovery.

Impact statement

My PhD has focused on the recovery of Aotearoa-New Zealand's rarest indigenous breeding bird, tara iti (New Zealand fairy tern, *Sternula nereis davisae*). Throughout the course of my PhD, I have worked and collaborated directly with Aotearoa-New Zealand's government Department of Conservation (DOC), and the surrounding tara iti stakeholder community, to assist in recovery planning. This work has led to tangible improvements in tara iti conservation, including resolving unaddressed conflicts in the community, finding a new management plan and a path forward for the first time in a decade, as well as measuring management efficacy and population viability for the first time in fifteen years. These outputs will impact positively on tara iti population recovery. I also met and engaged with the whole stakeholder group, listening to their ideas and values, and explaining my research findings to them. I provided a framework, datasets and code that DOC can use to continue to structure their programme.

Beyond its immediate impact on tara iti conservation, my research demonstrates the utility of decision analysis and structured decision making (SDM) in conservation planning for threatened species management and provides applicable examples for conservationists elsewhere. In particular, I provide a clear example of how SDM can improve inclusive co-management, widely sought but rarely achieved in conservation practice (Chapter 2; published in *Conservation Biology* (McMurdo Hamilton et al. 2020)).

Key outputs:

- Assisted the Tara iti Recovery Group in choosing a new management strategy.
- Created population modelling code and three unique datasets for DOC and took part in a tara iti data workshop to support future data management.
- Publication: McMurdo Hamilton, T., Canessa, S., Clarke, K., Gleeson, P., Mackenzie, F., Makan, T., Moses-Te Kani, G., Oliver, S., Parker, K.A. and Ewen, J.G., 2020. Applying a values-based decision process to facilitate comanagement of threatened species in Aotearoa New Zealand. *Conservation Biology.*
- Production of four documents for Department of Conservation (DOC):
    - Full structured decision making report, in press as: "Tara iti/New Zealand Fairy Tern: A structured approach to recovery planning". 2021. McMurdo Hamilton, T., Canessa, S., Makan, T., Ewen, J. G. Department of Conservation.

- Report on the initial stakeholder workshop: “Tara iti / New Zealand Fairy Tern: A structured approach to recovery planning workshop report”. 2018. McMurdo Hamilton, T., Makan, T., Ewen, J. G.
- Report for DOC and The Shorebirds Trust: “Trialling nest surveillance using camera traps for New Zealand fairy tern *Sternula nereis davisae*”. 2018. McMurdo Hamilton, T.
- Report for DOC and The Shorebirds Trust: “Tern species in captivity: a global review”. 2018. McMurdo Hamilton, T.

Acknowledgements

This PhD thesis would not have been possible without the help, support, generosity and kindness of so many people. My primary supervisors, John Ewen and Stefano Canessa, thank you so much for your guidance, for dedicating so much time and energy with good humour and patience. I have learned so much from both of you, and I hope to carry your wisdom with me in future. Robin Freeman, I will never forget how supportive and kind you have been over the past few years, from helping me with coding when I can barely see straight from stress to random chats over tea, to just believing in me, thank you so much. Tim Blackburn, thank you for stepping in as my UCL supervisor and giving me excellent advice along the way (whilst chatting about birds). Thank you, Trent Garner, you encouraged me to apply for the London NERC DTP, gave me confidence when I needed it, and did some cheerleading from the side!

My late supervisor, Ben Collen: over the one short year we worked together, he changed the way I thought about leadership and kindness in science forever. Thank you, Ben.

I owe endless thanks to my support networks at ZSL and UCL for supporting me through the highs and lows of my PhD journey. Guy Cowlishaw and Monika Böhm in particular helped me through a very difficult time after we tragically lost Ben Collen from our lives. To the 'lost children': Judith Ament, Anthony Dancer, Sergio Henriques, Gee Ferreira: having our friendship was (and is!) so important to me. Thanks to Xav Harrison for your help with mixed models. Fi Spooner aka Mrs Newt, I have finally caught you back up! Not only are you a great friend but you are my R queen and have really helped me through so many coding problems. Thanks especially for the times you had to listen to me explain problems only for me to then realise my mistake about halfway through the explanation. Lawrence Bellamy and Serein Sumner, you showed me what great teaching looked like and helped me with my associate fellowship journey. I am grateful for the support of Sarah Carter. Thank you to Amy Godfrey, Chris Langridge, Amrit Dehal and Cheryl Buffonge at UCL and ZSL for your administrative support.

I had the great privilege to work with a pretty cool bird and the excellent people that are involved with its recovery. Everyone in Aotearoa-New Zealand was very welcoming and helpful. First of all, thanks to Troy Makan at Department of Conservation for your guidance, commitment and energy for tara iti SDM, it was great to work with you, your positivity for this work made it happen. I want to extend that gratitude to Tony Beauchamp, Les Judd, Alex

Wilson, Ayla Wiles, Keven Drew, Gwenda Pulham, Tony Harbrakan, Robyn Davies, Eliane Lagnaz and John Neilson. Thank you for taking me out into the field and imparting all your tara iti wisdom. Shona Oliver, Pani Gleeson and the late Katie Clark were generous with their time and knowledge. Rest in peace, Katie. Thank you Jana Wold for tara iti chat, good craic, camping equipment, cool illustrations and for making me see tara iti as wee corgis forevermore (and you too now, reader). Amber, Anne, Bella and Pete Bellingham and Joyful, thanks for welcoming me into your family and teaching me how to bake lemonade scones. Thanks also to Christine Moginie. I was generously funded by Department of Conservation and The Shorebirds Trust while I carried out fieldwork.

As the world plunged into weirdness, chatting to the Ewen lab group every week provided me with much-needed support and community. Thanks Donal Smith, Amanda Trask, Katie Bickerton, Caitlin Andrews, Debbie Fogell, for chats, frog hunts, sneaky biscuits and snowy owl one-upmanship. Thank you to the CBER community, particularly Ali Fairbrass and Dan Bayley. The ‘informal lab group’ ZSL Social Science ladies, you continue to be a source of inspiration to me and have enhanced the way I think about conservation. I made so many fantastic friends on the London NERC DTP, with whom I can be the geek that I truly am, especially Lydia Franklinos, João Leite, Michael Stevens, Dan Nicholson, Joe Williamson and Mattie Brindle. Thanks to Ben Taylor being there for me through highs and lows and for listening to me say the same thing a thousand times and still responding with kindness, jokes and memes.

To all my friends: thanks for being there for me. Alice Ward, Rosemary Hahn, Philippa Richards, Sarashka King, Amelia Hawkes and Sally Hughes, thanks for the love and the laughs. Sabina Ahmadov you kept me sane and healthy over this last year. Barry, Gen Finerty and Gwen Maggs, your advice has helped so much. To my Edinburgh pals, you can call me Dr McBirdo now. My family have supported me unconditionally, and Mum, you taught me to love nature, thank you. To Julien Clin, you have given me strength, joy and love, and I couldn’t have done this without you, merci beaucoup.

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This PhD is dedicated to my late supervisor, Ben Collen.

“Remember the joy”

Chapter 1: Introduction

"I think of decision analysis as 'a formalisation of common sense for decision problems that are too complex for the informal use of common sense.'" (Keeney 1982).

In this thesis, I explore the complexities of decision making for conservation of threatened species through the focus on an imperilled subspecies of fairy tern in Aotearoa-New Zealand. I aim to advance the way we approach decisions (inclusivity, values), the way we make decisions (objectives, alternatives, rationality) and the way we use data to support these decisions (managing biased, imperfect datasets).

1.1 Conservation of biodiversity

Conservation of biodiversity has emerged in many forms throughout humanity's existence, from indigenous peoples' balance with and management of treasured flora and fauna, to the mainstream Western 'conservation movement' of the last 120 years (Pascual et al. 2021). For those who value the maintenance of diversity of life on Earth, for whatever reason, the need for conservation has perhaps never been more urgent. In response to declining wildlife populations and increased extinction rates in the 20th century, the scientific field of conservation has expanded and progressed through different framings and goals (Mace 2014), alongside a large increase in research, publications and funding (Griffiths & Dos Santos, 2012; Stroud et al., 2014). There have been success stories. For a few threatened taxa, recovery groups formed and implemented direct intervention to bring species back from the brink of extinction (Hoffmann et al. 2010; Young et al. 2014; Bolam et al. 2021). Programmes that have been particularly effective in researching and implementing workable conservation interventions include the recovery of poisoned vulture populations, reintroductions of whooping cranes and the reduction of bycatch of procellariform seabirds (Williams et al. 2020). Yet, these successes notwithstanding, biodiversity remains in grave peril, and conservation falls far short of global Aichi targets (IPBES 2019). The reasons cited for this range from difficulties in accessing or using information and evidence (Catalano et al. 2019; Walsh et al. 2019; Christie et al. 2019) to a lack of resources (Catalano et al. 2019; Wright et al. 2020) and poor resource allocation (Gerber et al. 2018; Buxton et al. 2020), capitalism (Soulé 2013; Büscher & Fletcher 2020), the human–nature dichotomy (Büscher & Fletcher 2020), socio-economic inequality (Matulis & Moyer 2017) and human population growth (Wilson 2016). This is a daunting list of large, difficult constraints to achieving conservation goals.

There are other barriers that limit conservationists' ability to progress that are perhaps less controversial and more tractable. One example is that, precisely because of the issues described above, conservation problems are more complex, more "wicked," than they are often treated as, and conservationists still over-rely on biological science to solve them (Gregory et al. 2006; Tulloch et al. 2015; Williams et al. 2020). Complexity pervades all conservation decisions, whether they pertain to selecting land for management, permitting development, deciding how to monitor a species or choosing management strategies for threatened species recovery (McCarthy 2014). There are valid reasons why this happens, such as the way conservationists are predominantly trained in biological sciences (Fox et al. 2006), the limitations of the institutes they operate in (Game et al. 2014), the slow integration of interdisciplinary tools into conservation science (Mascia et al. 2003; Bennett et al. 2017; Williams et al. 2020) and the particular pressures of this 'crisis' discipline (Meine et al. 2006). Reducing complex problems to more simple ones need not be negative, and it is something humans do instinctively to make decisions in our complex environment (Kahneman 2011). Nonetheless, conservationists increasingly realise that tackling complexity and learning to work with it provides an opportunity to improve thinking for biodiversity conservation and to generate solutions that effect change in the real world (Game et al. 2014; Bennett et al. 2017; Evans et al. 2017; Knight et al. 2019; Evans 2021).

Complexity of decision making in threatened species recovery

In species recovery planning, complexity pervades the people involved and their values, the data available and the decision making process. If not treated appropriately, complexity can prevent practitioners and managers from finding informed, reasoned and equitable solutions.

Data and science have key roles to play in informing managers of how things are now and predicting how different actions might change things. Yet, the data collected by conservation programmes are often complex and deficient, making it harder for recovery groups to use them effectively in decision making. The most common difficulty is uncertainty, primarily epistemic uncertainty (in estimates, the ecological system, and models (Regan et al. 2002)) but also aleatory uncertainty (stochasticity), which is a critical concern when dealing with small populations (McCarthy 2014). Furthermore, because robust, experimental setups are rare, systematic errors can occur (Regan et al. 2002; Christie et al. 2019). To be effective in decision making—that is, to reduce error, bias and subjective judgment—it is essential to understand and tackle these uncertainties (Tversky & Kahneman 1974; Regan et al. 2002; Tulloch et al. 2015; Milner-Gulland & Shea 2017; Dobson et al. 2020). This can be done by explicitly predicting how a tool that was effective for another species would impact the target species (Snyder et al. 1996; Martínez-Abraín & Oro 2013); by incorporating uncertainty in predictive modelling exercises (McGowan et al. 2011; Milner-Gulland & Shea 2017); by acknowledging and counteracting bias through appropriate tools, such as counterfactual analysis (Ferraro & Pattanayak 2006), and by avoiding making ad hoc decisions based on cursory examination of biased data (Walsh et al. 2019).

The second complexity relates to the social aspects of species management, namely the diverse stakeholder groups involved (Gregory et al. 2012b). Indeed, this is one of the most common reasons that projects fail (Yaffee 1994; Scheele et al. 2018; Catalano et al. 2019). The challenges of this social complexity have long been identified (Brown 2003) but are not fully resolved (Gregory 2016; Salomon et al. 2018; Wheeler et al. 2020). While the integration of other knowledge systems beyond scientific evidence is gathering pace, showing positive outcomes for biodiversity and people (Garnett et al. 2018; Dam Lam et al. 2019; Stoate et al. 2019; Rayne et al. 2020), the current conservation model is not always viewed as inclusive or receptive to value systems outside science (Brown 2003; Matulis & Moyer 2017; Pascual et al. 2021). Stakeholder participation is rightly becoming an important feature in conservation management (Sterling et al. 2017), but simply fielding concerns in a town-hall or listening to stakeholder problems and analysing them separately does not amount to meaningful engagement (Gregory et al. 2012a; Gregory 2016). A more equitable approach to decision making seeks to understand diverse stakeholder values and knowledge from the start, then to integrate these in a meaningful and decision-relevant way, all the while promoting participation and transparency (Gregory et al. 2008; Sterling et al. 2017; Bennett et al. 2019). This may need a step-change to recognise, first, that values drive our decisions (Keeney 1996), and second, that science itself is a value and not free from subjectivity (Giles et al. 2016; Salomon et al. 2018). Finally, although the importance of legitimising stakeholder values in species management has been long recognised (e.g. Yaffee 1994), the teaching and training in tools to do this effectively have been lacking in conservation (Mascia et al. 2003; Bennett et al. 2017; Evans 2021).

Decision making in conservation – an opportunity to improve

While not every decision requires systematic thinking (Keeney 2004), for the reasons described above, many in conservation do. More practitioners are realising that they need to adopt formal, transdisciplinary and inclusive approaches to decision making, and there is a growing number of papers that describe the different tools available (Bower et al. 2018; Schwartz et al. 2018). Recognising the complexity of conservation decisions and accepting that values or objectives (what we want) drive our decisions calls for a fundamental shift. Instead of focusing on available conservation actions first (alternatives-focused thinking), recovery groups should use values as their starting point (values-focused thinking, Keeney (1996)). From here, conservationists then need tools that can integrate multiple objectives in a meaningful way. Defining objectives first and explicitly modelling alternatives to achieve these can help to close the research–implementation gap. Decision analysis and, in particular, structured decision making (SDM) provide a framework and tools to address all these elements of complexity in decision making for conservation. Indeed, SDM was developed with these problems in mind (Gregory & Keeney 2002).

1.2 Structured decision making

Structured decision making is a facilitated, collaborative approach to making environmental decisions. It is not a single method or theory, but a way of thinking and a set of approaches. Based on Multi-Attribute Utility Theory (approaches to make decisions with multiple, sometimes competing objectives) and Decision Analysis (methods for logically proceeding through and analysing complex decision problems), SDM combines these elements with theory and practice from disciplines such as facilitation and psychology. In SDM, a decision problem is broken down into its component steps (Fig. 1.1): (1) setting the decision context, (2) clearly defining objectives, (3) developing potential management alternatives, (4) predicting the performance (with uncertainty) of each management alternative against the objectives, (5) finding the best decision across objectives (accounting for risk), and (6) monitoring to track the outcomes of a given choice (Gregory & Keeney 2002; Gregory et al. 2012a). These steps are supported by techniques that have been developed in the decision sciences for the past 50 years.

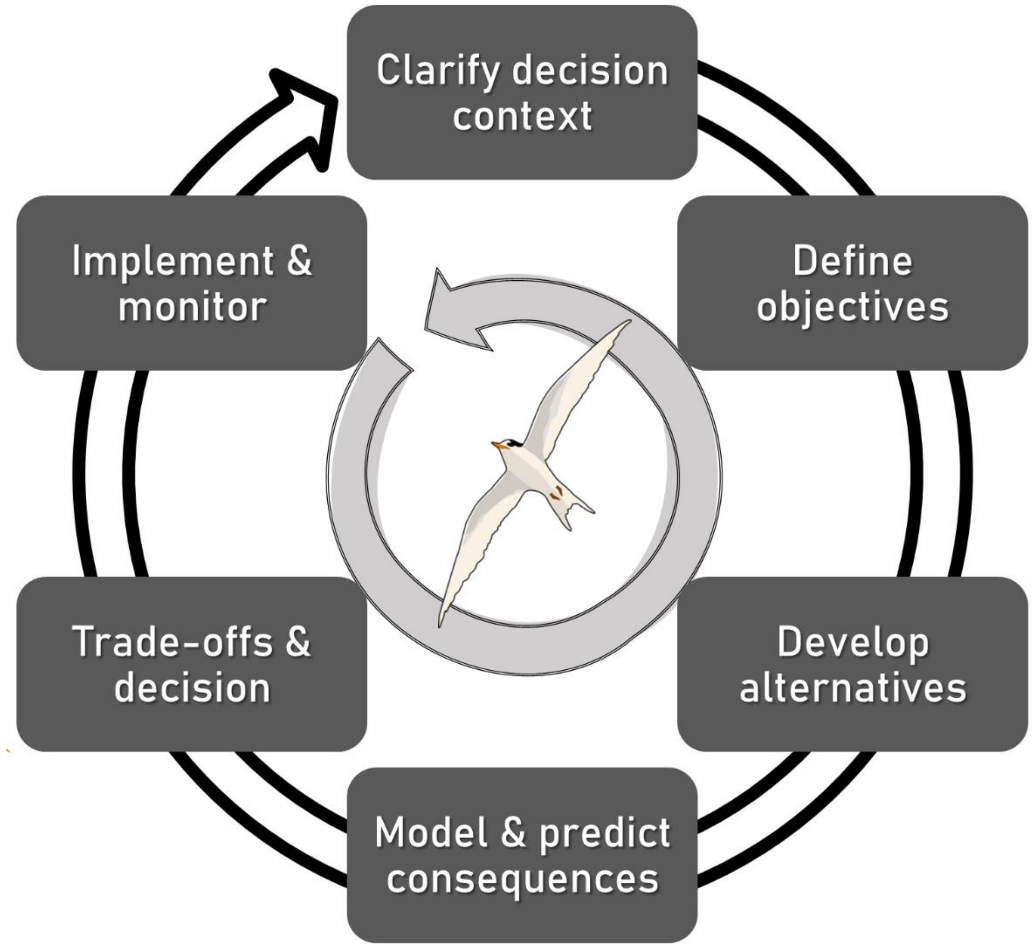

Figure 1.1. The decision making cycle (adapted from Gregory et al. (2012a)). Anticlockwise arrow indicates that steps may be reiterated if required. Illustration by J. Wold.

Because SDM acknowledges that values (objectives) are the main driver of decisions (Keeney 1996), it is ideal for the kind of diverse stakeholder groups involved in conservation. As such, it has been used in a wide range of environmental decisions (Moore & Runge 2012; Thorne et al. 2015; Hayek et al. 2016; Runge et al. 2020). Its emphasis on transparency, values and inclusion of multiple partners allows for shared, co-developed solutions and their implementation (Bennett et al. 2019). Aside from its focus on values, another feature of SDM is its emphasis on structured group deliberations and interaction supported by the best available information. People learn together in SDM. They build a common information base, agree on relevant information and knowledge, and learn about the values that form the basis of the decision. The primary goal is thus to improve thinking in order to make smart, robust decisions (Gregory 2000).

At its core, SDM represents a highly logical and practical set of steps that is relevant to all sizes of conservation problems. The decision analytic framework more broadly provides tools and techniques that are highly applicable to complex problems with great uncertainty. Furthermore, through its aforementioned focus on objectives, SDM can help close the research–implementation gap. The purpose of applied science is to help in the choice of management actions to achieve objectives. In SDM, scientists can work directly to predict the consequences of alternative actions and so produce decision-relevant information. There is a growing recognition of the importance of structured approaches such as SDM (it is widely adopted in the US by state and federal natural resource agencies, for example (Runge et al. 2020)), but they remain underused in conservation globally. Consequently, there are still only few examples of their application in biodiversity conservation. In this thesis, *tara iti* (New Zealand fairy tern, *Sternula nereis davisae*) provides a case study for how decision-analytic tools can be used to address complexity and aid better, more inclusive planning and decision making in threatened species management.

1.3 Study system: tara iti (New Zealand fairy tern, *Sternula nereis davisae*)

Tara iti (‘small one’ in te reo Māori) is a small, non-migratory seabird found in the North Island of Aotearoa-New Zealand (Fig. 1.2). It is New Zealand’s most endangered native breeding bird and is listed as ‘Nationally Critical’ (Robertson et al. 2017). Adults weigh about 70 g, have a wingspan of 44-53 cm and length of 25 cm (Heather & Robertson 2005). It is the rarest subspecies of the fairy tern, *S. nereis,* which is listed as ‘Vulnerable’ on the IUCN Red List (BirdLife International, 2016). Tara iti is morphologically, behaviourally and genetically distinct from the two other fairy tern subspecies populations that are found, respectively, in New Caledonia (*S. n. exsul*) and southern and western Australia (*S. n. nereis*) (Higgins & Davies 1996; Hansen 2006; Baling 2008).

Figure 1.2. Tara iti at Mangawhai. Reproduced with kind permission from K. Drew.

Tara iti is highly scarce. Up until the early 1900s, it was thought to be widely distributed across the coastline of the North Island and inland rivers of South Island (but probably confused with little tern, *S. albifrons*). By the 1950s, however, it had become restricted to approximately 18 pairs in the northern parts of the North Island (Parrish & Pulham 1995; Heather & Robertson 2005). The population continued to decline until 1983, when there were only three known pairs (Parrish & Pulham 1995; Ferreira et al. 2005; Brooks et al. 2011). Conservation efforts have increased the population size (Ferreira et al. 2005), but it remains low: in 2017, its population was estimated to be c. 40 individuals (DOC unpublished data). Over the last 20 years, its breeding population has plateaued between 7–12 pairs (Fig. 1.3; Brooks et al. 2011; Maloney et al. 2017). In 2005, scientists estimated tara iti’s extinction risk in 50 years to be 0.39 (Ferreira et al. 2005).

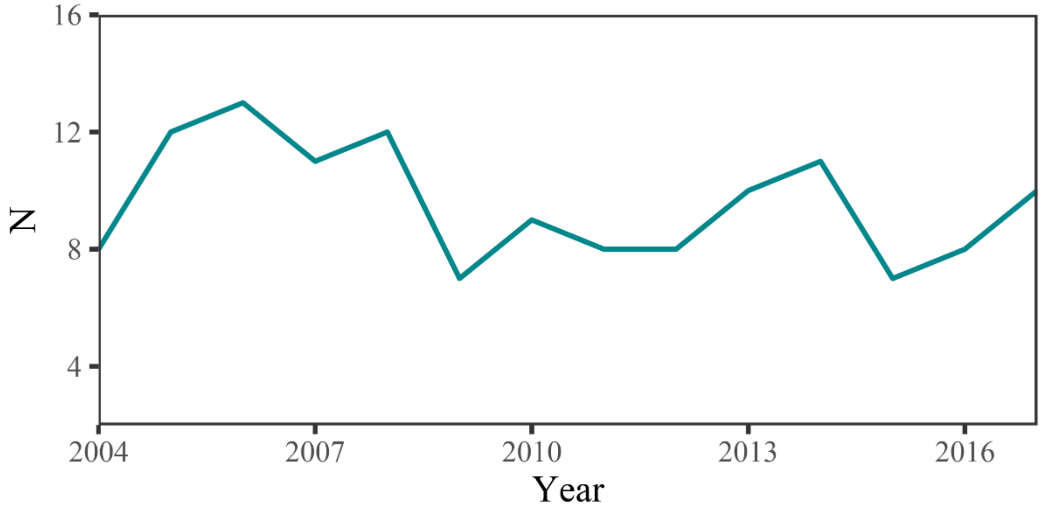

Figure 1.3. Number of breeding female tara iti between 2004 and 2017 (DOC unpublished data)

Tara iti inhabits estuarine and coastal habitats and breeds at four locations in Northland (Waipu Estuary, Mangawhai Spit) and Auckland (Pakiri Beach, Papakanui Spit) regions (Fig. 1.4). Each of these is marked by a proximity to tidal, estuarine waters and oceanic waters, and the presence of open, sandy areas often with little vegetation, and shell distributed through them ('shell patches'). Outside the breeding season, the population flocks at various harbours in the northern half of the North Island, but predominantly in the Kaipara Harbour (Fig. 1.4). Adult tara iti feed on juvenile fish and elvers. Goby species form most of their diet and are the most important prey for chick-rearing, but they also take flounder and shrimp (Ismar et al. 2014). Tara iti are splash-divers, only submerging in the very top portion of the water column when hovering and diving for their prey in mid-estuary, tidal pool and shallow oceanic waters (Heather & Robertson 2005; Ismar et al. 2014).

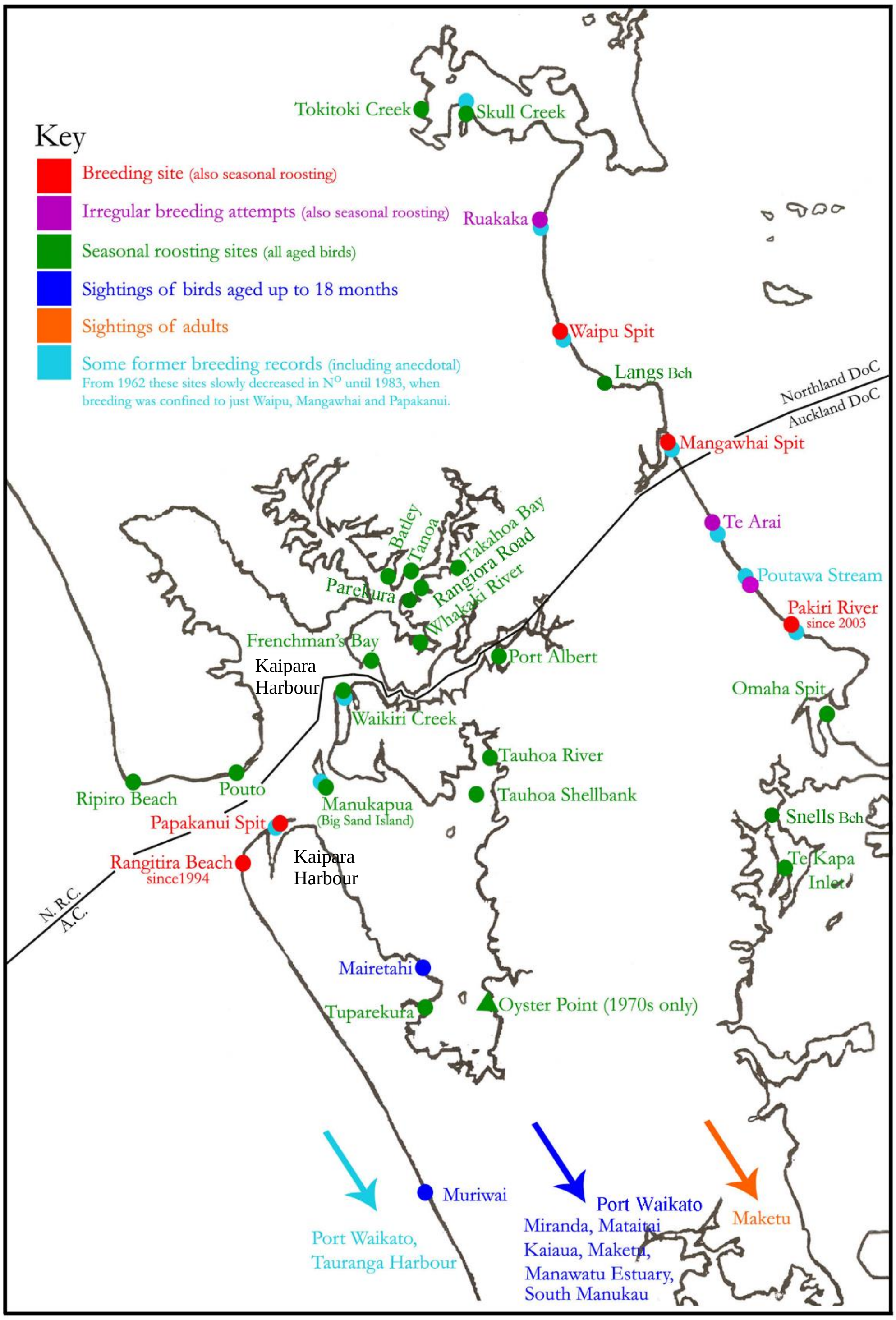

Figure 1.4. Recorded sightings of tara iti by Birds NZ and New Zealand Wader Study Group between 1991 and 2017. Area shows northern North Island regions of Northland and Auckland. Key shows different site uses. Map reproduced with permission by Pulham, G.A., Harbraken, A.M. & Vaughan, G.N., 2018

Tara iti nest between October and January and are predominantly monogamous with pair bonds held between seasons. In contrast with many tern species, they nest solitarily, although this could be an artefact of their small population size. Nests tend to be a minimum of 10 metres apart, and at some sites, adults exhibit highly territorial behaviour (Parrish & Pulham 1995;

Hansen 2006). They scrape a nest in the sand and lay 1–2 eggs (average 1.67, Ferreira et al., (2005)). Both adults incubate for 23–25 days, and chicks fledge at 23 days old (Heather & Robertson 2005). After fledglings depart the natal nest, they continue to be cared for by parents (typically the male) for possibly up to two months (Preddey 2008). They can re-nest up to two times after a clutch has been lost, up until early January, but they only raise one clutch per year to fledging (Hansen 2006). Not all adult females (age 2 and older) attempt to breed; the mean annual frequency between 1997–2017 was 67% (± 18%, S.D.). Fertility appears to be low and, over the same period, the annual mean proportion of fertile eggs laid was 68% (± 15.6%, S.D.). There are two known ‘infertile’ pairs, and some fertile pairs have been known to lay inviable eggs (Maloney et al., 2017). The population’s sex ratio is currently biased towards males, resulting in many unpaired males each season. Breeding productivity is low (Fig. 1.5); between 2006–2015, the fledging rate was 0.33 fledglings per fertile egg per annum and fledgling survival rates are highly varied (Ferreira et al., 2005; Maloney et al., 2017).

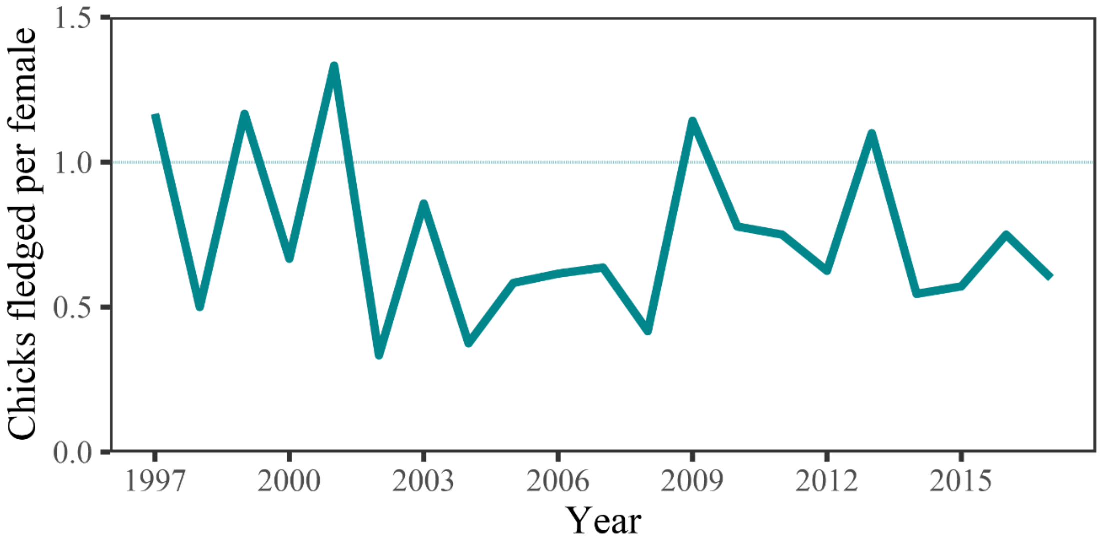

Figure 1.5. Number of chicks fledged per female between 1997 and 2017. Dotted line indicates one chick fledged per female (DOC unpublished data).

Tara iti conservation

Many extrinsic threats that tara iti face are common amongst shore-breeding tern species. These include: disturbance at their breeding sites by humans and dogs; habitat loss due to development of coastal areas; stormy and extreme weather causing tidal inundation or sand burial of low-lying nests; the predation of eggs, chicks and adults by introduced mammalian predators (namely rats *Rattus* spp., mustelids *Mustela* spp., domestic cats *Felix catus*) and native aerial predators (namely southern black-backed gulls (*Larus dominicanus*) and kahu (Australasian harrier, *Circus approximans*)) (Hansen, 2006). Tara iti also face intrinsic threats, such as a high rate of relatedness between some breeding pairs, egg infertility and early embryo death, as well as a skewed sex ratio in the population. There is a high possibility that the population suffers from inbreeding depression (Baling 2008).

Species conservation management started in 1983 by Aotearoa-New Zealand's Department of Conservation (DOC; then known as New Zealand Wildlife Service) but began in earnest in 1991, when a recovery group was established (Hansen 2006). Management has mostly focused on increasing productivity, and evolved in an ad hoc way, beginning with ranger presence, building up to full-time rangers and predator control at all sites, to regular captive management of eggs by Auckland Zoo staff from 1997 onwards (Ferreira et al. 2005). Now, rangers are based at each breeding site throughout the season and carry out the core elements of management with support from species experts: monitoring and protecting nests, managing eggs, carrying out predator control (checking live and kill traps), vegetation management and public outreach. Eggs are managed when a nest is perceived to be under threat. Foster pairs, artificial incubation and nest shifting are all used. To further improve the quality of nesting sites, DOC and volunteer groups work outside the breeding season to create shell patches, control vegetation and carry out additional predator control.

Many elements of tara iti conservation (e.g., monitoring, predator control, advocacy, research) are supported by the diverse stakeholder group, mostly in the local communities. Tara iti live across the *rohe* (land) of many different *iwi* (Māori tribes), and representatives from the iwi trusts (mainly Te Uri o Hau, Ngati Whatua and Ngati Manuhiri) advise DOC and provide support, such as additional predator control. Other active stakeholders include volunteer community groups such as About Tern, non-governmental organisations Birds NZ and Forest and Bird, and charitable trusts New Zealand Fairy Tern Charitable Trust and The Shorebirds Trust, as well as scientists within DOC and based at universities.

Tara iti management and research were coordinated by a DOC-led recovery group from 1997 until it was disbanded by senior managers in 2006 (Maloney et al. 2017). The absence of the recovery group led to various problems in both species management and relationships between stakeholders, which came to a head in 2017. The last recovery plan period was 2005–2015, and its target of reaching 100 pairs by 2021 was missed by a considerable margin. Most of its research goals were missed, including measuring the efficacy of the current management. Egg management protocols remained unchanged, although scientists suggested it might have reached its threshold (Ferreira et al. 2005), and the number of breeding females plateaued at a low level (Fig 1.2; Maloney et al. (2017)). Without the recovery group, information sharing between DOC and other stakeholders had stopped, which contributed to a growing mistrust and a breakdown in communications between parties (Maloney et al. 2017).

In 2017, a DOC internal review recommended the recovery group be reformed immediately (Maloney et al. 2017). While the review consulted with many in the community, its field management recommendations (such as captive rearing with releases and genetic rescue) met a mixed response from the stakeholder group. This was perhaps because only the review's authors (four scientists) had contributed to these, or because of the uncertainty around the success of the intensive actions that were proposed, such as captive rearing with releases. This technique had failed in the past (Gummer 2003), and there is little information available about its application for terns globally (Chapter 3). Divergent opinions and communication problems persisted, and recognising there was no clear way forward, the newly re-formed recovery group decided to use an SDM process to restart planning and implementation.

1.4 Thesis aims and chapter overview

With this work, I aim to demonstrate how decision-analytic tools can improve the way decisions are thought about and made, and benefit threatened species and the people involved. In the following chapters, I use various decision-analytic methods to assist the recovery planning of tara iti and to provide examples of how practitioners can best address complexity, increase inclusivity and make robust decisions.

In **Chapter 2**, I demonstrate the power of structured decision making as a tool to support the inclusive co-management of endangered species between Aotearoa-New Zealand's government, Māori (New Zealanders of indigenous descent) and the wider community. Working with tara iti and *pekapeka* (short-tailed bat, *Mystacina tuberculata*) recovery groups, I integrated *mātauranga Māori* (Māori knowledge and perspectives) into each decision step, from a transparent definition and assessment of management objectives to co-developed alternatives and trade-offs.

In **Chapter 3**, I provide novel information for potential captive rearing and releases of tara iti, analysing the outcomes of a unique seabird translocation trial. The trial is unique in that it used chick translocation techniques on seabirds with post-fledging parental care. This research shows the value of creative thinking about alternative management actions and informs uncertainty in captive management efficacy for tern species.

In **Chapter 4**, I describe the full SDM process I carried out with the tara iti stakeholder community to assist their recommendation of a new recovery plan after a decade of deadlock. The process navigated conflict, uncertain outcomes and diverse values to produce a new recommendation for tara iti.

In **Chapter 5**, I use a combination of data analysis and decision-support tools to evaluate tara iti egg management over the last 20 years. This chapter illustrates how biased monitoring data can be used to inform decisions, and the utility of using multiple counterfactuals with uncertainty to overcome biases.

1.5 Notes on contributions and publications

Apart from the Introduction and Discussion, each chapter in this thesis has been prepared as a manuscript and so it is written in first person plural perspective, with some methodology repeated. I am primary author and contributor in all chapters, and my supervisors John Ewen and Stefano Canessa (both at Institute of Zoology, Zoological Society of London) contributed to all aspects of the thesis. My supervisor Tim Blackburn (UCL) read and commented on drafts. Troy Makan (Department of Conservation, DOC) chaired the Tara iti Recovery Group throughout my PhD and steered the SDM work in-country. Other co-authors and notable contributions are as follows:

Chapter 2 is co-authored by Stefano Canessa, Fiona Mackenzie, Troy Makan, Gena Moses-Te Kani, Shona Oliver, Kevin A. Parker, Katie Clark, Pani Gleeson, and John Ewen. It was published in *Conservation Biology* (McMurdo Hamilton et al. 2020). FM (Ngāti Manuhiri Settlement Trust, Pou Kaitiaki), GMTK (Ngati Kuia, Hōkai Nuku (Ngāti Manuhiri & Ngāti Whātua), Pou Tātaki), SO and PG (Ngāti Whatua o Kaipara / Ngā Maunga Whakahii o Kaipara, Pouwhakahaere Te Tari Taiao) and KC (Te Uri O Hau) participated in the workshops and contributed towards the manuscript. JE, SC and KP facilitated and gathered the data from the pekapeka workshops, TM (DOC) co-facilitated tara iti workshops and contributed towards the manuscript.

Chapter 3 is co-authored by Stefano Canessa, Julie Cole, Nik Cole, Issabelle Desiré, Carl G. Jones, Vikash Tatayah, John Ewen. The seabird translocation programme is managed by NC, CJ and VT at Mauritian Wildlife Foundation and Durrell Wildlife Conservation Trust. The fieldwork and data collection were led by JC and ID, supported by Bhavnah Komul, Johann Bourgeois, Vasisht Seetapah, Marjorie Fassin and O'Brian Clarisse, all at Mauritian Wildlife Foundation.

Chapters 4 and 5 are co-authored by Stefano Canessa, Tony Beauchamp, Troy Makan and John Ewen. TB (DOC) collects and manages the tara iti breeding and sightings data and TM co-facilitated tara iti workshops, both contributed towards manuscripts. Many people contributed towards the tara iti datasets, notably Gwenda Pulham and Tony Harbrakan at Birds NZ and many rangers at DOC. The whole tara iti community participated in the workshops, but especially: Tony Harbraken, Gwenda Pulham and Ian Southey (Birds NZ); TB, TM, Richard Maloney, Ayla Wiles, Graeme Taylor, Alex Wilson and Les Judd (DOC); Susan Steedman and Sioux Plowman (About Tern); Richard Gibson (Auckland Zoo), Katie Clark (Te Uri O Hau); Pani Gleeson and Shona Oliver (Ngāti Whatua o Kaipara / Ngā Maunga Whakahii o Kaipara, Pouwhakahaere Te Tari Taiao).

Appendix A was published as Supplementary Material to Chapter 2 and contains additional method information. Appendix B contains additional methods and analysis pertaining to Chapter 4.

Chapter 2: Applying a values-based decision process to facilitate co-management of threatened species in Aotearoa-New Zealand

Abstract

Ko koe ki tēnā, ko ahau ki tēnai kīwai o te kete (you at that, and I at this handle of the basket). This *Māori* (New Zealanders of indigenous descent) saying conveys the principle of cooperation—we achieve more through working together, rather than separately. Despite decades of calls to rectify cultural imbalance in conservation, threatened species management still relies overwhelmingly on ideas from Western science and on top-down implementation. Values-based approaches to decision-making can be used to integrate indigenous peoples' values into species conservation in a more meaningful way. We used such a values-based method, structured decision making, to develop co-management of *pekapeka* (*Mystacina tuberculata*) (short-tailed bat) and *tara iti* (*Sternula nereis davisae*) (New Zealand fairy tern) between Māori and *Pākehā* (New Zealanders of European descent). We implemented this framework in a series of workshops, with facilitated discussions to articulate values, gather expert knowledge to predict outcomes and make management recommendations. For both species, stakeholders clearly stated their values as fundamental objectives from the start, which allowed alternative strategies to be devised that naturally addressed their diverse values, including *mātauranga Māori* (Māori knowledge and perspectives). On this shared basis, all partners willingly engaged in the process and decisions were largely agreed to by all. Most expectations of conflicts between values of Western science and Māori culture were unfounded. Where required, positive compromises were made by jointly developing alternative strategies. The values-based process successfully *taha wairua taha tangata* (brought both worlds together to achieve the objective) through co-developed recovery strategies. This approach challenges the traditional model of scientists first preparing management plans focused on biological objectives, then consulting indigenous groups for approval. We recommend values-based approaches, such as structured decision making, as powerful methods for the development of co-management conservation plans between different peoples.

2.1 Introduction

Historically, conservation actions have been overwhelmingly inspired by biological insights and implemented top-down. Despite over two decades of calls for more equal and inclusive conservation, where agencies work with communities and indigenous groups (United Nations 1992; Wright et al. 1995; Tallis & Lubchenco 2014; Lyver et al. 2018), inclusivity remains the exception (Mascia et al. 2003; Gregory 2016). Many indigenous groups feel marginalized by, or will not engage in, processes that do not recognize and account for their values significantly (Gregory 2016; Wheeler et al. 2020). Changing ingrained practices requires deep engagement between partners and interdisciplinary facilitation methods (Brown 2003; Bennett et al. 2017), but practical examples of how to achieve such a change remain scarce.

Most Western conservationists are trained as biologists and so focus on understanding and mitigating population declines (Fox et al. 2006). Therefore, biology dominates the drive for evidence-based conservation, that is the use of systematically collected data to choose management (Sutherland et al. 2004). Evidence-based conservation is a much-needed improvement of current practice, but it does not need to clash with inclusivity. Exclusive focus on biological evidence fails to acknowledge the complexity of decision making, particularly the diverse values involved (Evans et al. 2017; Toomey et al. 2017), making assumptions about what those values are or explicitly excluding them (Gregory et al. 2012a). Furthermore, evidence-based conservation itself is not objective because it is embedded in Western science beliefs about how to correctly interact with the environment (Giles et al. 2016; Salomon et al. 2018).

Because conservation is never value-neutral, evidence can only play a support role, albeit a crucial one (Brown 2003; Gregory et al. 2012b; Peterson et al. 2013; Evans et al. 2017). Conservationists must first understand that the objectives of recovery plans reflect values, including, but not limited to, ecological values of nature. Then they can gather the right information about all objectives, including scientific evidence, to generate long-term solutions that are widely accepted (Gregory et al. 2012a). The centrality of objectives is recognized by many decision making approaches (Schwartz et al. 2018). Among those, structured decision making (SDM) is a framework that originates from decision theory and risk analysis (Gregory et al. 2012a). Structured decision making is an iterative process with six steps: set the decision context; clearly define objectives; develop possible management alternatives; predict performances of alternatives against the objectives; find the best decision across objectives;

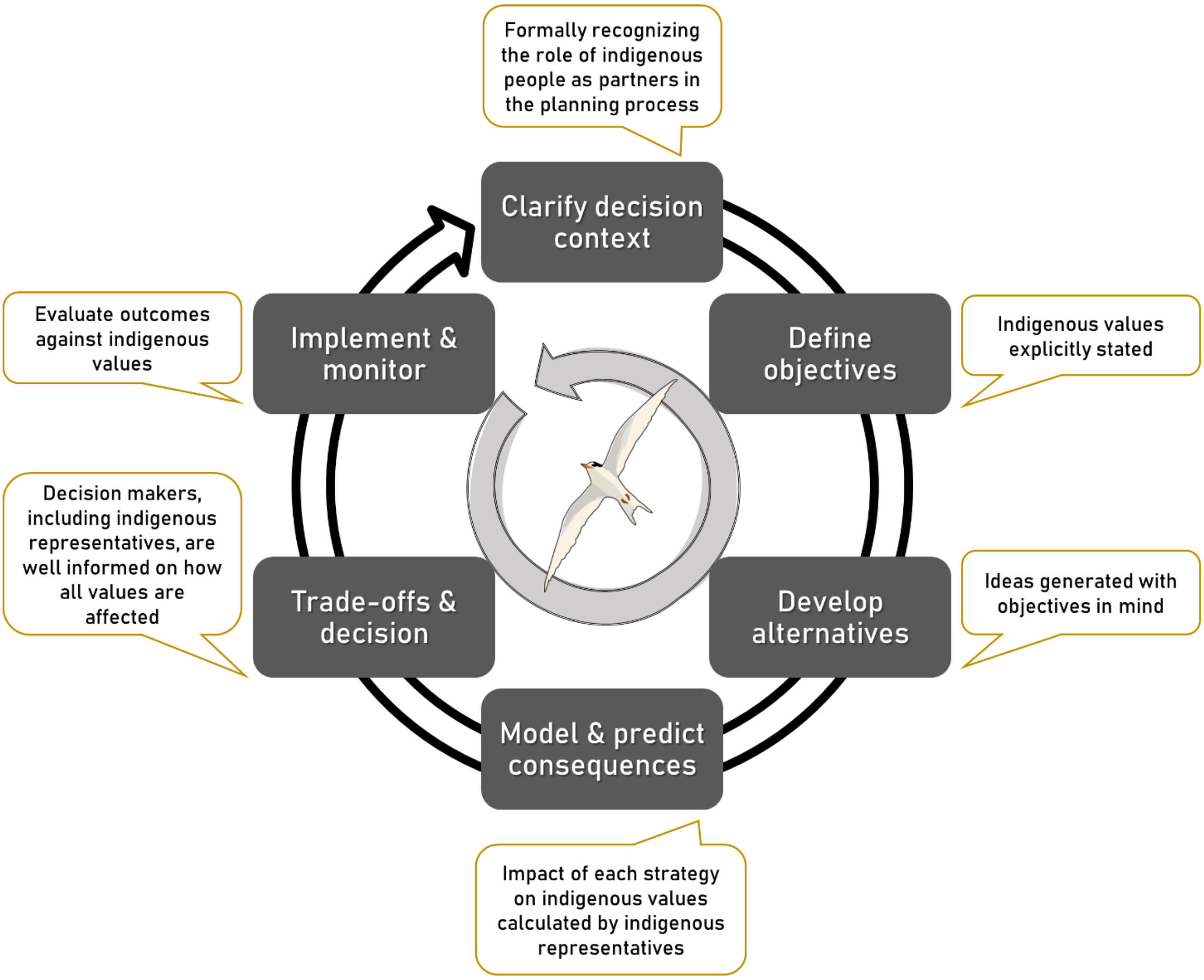

Figure 2.1. The decision-making cycle (adapted from Gregory et al. (2012a)). Anticlockwise arrow, steps may be reiterated if required; bubbles, interpretation of integration of indigenous values relative to Māori as an example, transferable to any indigenous group as that group culture demands. Illustration by J. Wold.

and sixth, monitor to track outcomes (Fig. 2.1) (Gregory et al. 2012b). Because SDM acknowledges that values (objectives) are the main driver of decisions (Keeney 1996), it is ideal for diverse stakeholder groups. Planners have used SDM in many environmental decisions, from controlling invasive willow in Australia (Moore & Runge 2012) to improving resilience of tidal marshes to climate change (Thorne et al. 2015).

Structured decision making offers opportunities to legitimately integrate indigenous values into conservation decisions. It emphasises transparency and inclusion of multiple partners, allowing for shared solutions and their implementation (Bennett et al. 2019). Crucially, values are meaningfully described and integrated into each step of a decision (Fig. 2.1). SDM has been used in this way to conserve boreal woodland caribou (*Rangifer tarandus caribou*) in western Canada (Hayek et al. 2016), to control non-native fish below Glen Canyon Dam in Arizona

(U.S.A), (Runge et al. 2011a) and to devise water-use plans in British Columbia (Canada) (Gregory et al. 2008). In these examples, SDM successfully increased process transparency, participation, and shared solutions. However, published examples beyond the United States and Canada are scarce (but see Arvai & Post (2012)).

Structured decision making could be particularly useful in a country like Aotearoa-New Zealand , where there is a strong desire to rectify a history of cultural bias in environmental management (Wright et al. 1995; Department of Conservation 2000). Te Tiritiri o Waitangi (The Treaty of Waitangi, 1840) is an agreement between representatives of Māori (the indigenous people of Aotearoa-New Zealand) and the British Crown. Its second article and its subsequent interpretation in the Conservation Act (New Zealand Government 1975) and Resource Management Act (New Zealand Government 1991) mandate that decision making consider both *mātauranga Māori* **(**Māori knowledge, perspective, and culture) and *Pākehā* (New Zealander of European descent) values. This has led to some positive steps. For example, *iwi* (tribe) management plans support valid influence of iwi on planning processes (Thompson-Fawcett et al. 2017), yet consideration of mātauranga Māori remains largely unfulfilled and invisible in conservation practice (McAllister et al. 2019; Wehi et al. 2019; Rayne et al. 2020). For example, in our experience, partners, such as iwi, are commonly asked to endorse proposals only *after* they have been developed.

We examined how SDM provides a way forward from current problematic practice to provide a tool for developing co-management of threatened species' recovery plans, integrating mātauranga Māori and promoting *ako* (teaching and learning through knowledge exchange) through open and transparent definition and assessment of management objectives, alternatives, and trade-offs. We use the term *co-management* to mean meaningful "partnerships between Māori and Crown agencies in the management of biodiversity, consistent with the principles of the Treaty of Waitangi," as defined by the Aotearoa-New Zealand government's Department of Conservation (DOC) (Department of Conservation 2000). We applied SDM to two *taonga* (culturally valuable) species' recovery programmes.

2.2 Methods

Case studies in recovery planning for taonga species

The *pekapeka* (*Mystacina tuberculata*) (short-tailed bat) is endemic to Aotearoa-New Zealand and comprises three subspecies (northern, central and southern) (Lloyd 2003). Translocations have been suggested as a potential recovery strategy but have been unsuccessful to date. A first attempt failed when all translocated pekapeka left the release site within minutes. A second attempt was aborted after translocated bats developed an unidentified infectious disease (Gartrell 2007). No further translocations have been attempted. In 2014, DOC chose to use SDM to plan a translocation of the northern pekapeka subspecies from their only remnant population on *Te Hauturu-o-Toi* (Little Barrier Island) in the *rohe* **(**territory) of the iwi Ngāti Manuhiri.

The *tara iti* (*Sternula nereis davisae*) (New Zealand fairy tern) is Aotearoa-New Zealand's rarest indigenous breeding bird, breeding at only a few beaches across the rohe of iwi Te Uri o Hau, Ngāti Whātua o Kaipara, and Ngāti Manuhiri in the Auckland and Northland regions. Despite intensive management with close community involvement, in 2020 fewer than 12 tara iti breeding pairs remain. In 2017, an internal review reported a communication breakdown between DOC and the wider community involved in tara iti recovery, including iwi partners. Although many partners' thoughts were recorded, all field management recommendations in the report came from the four scientist authors and focused on a single biological value (tara iti population recovery). Recognizing these problems, and after the successful pekapeka process, DOC suggested using SDM to restart tara iti recovery planning and implementation.

Preparing for Structured Decision Making

We applied the same SDM process for both examples unless otherwise stated. To select participants, we consulted DOC on known stakeholders engaged with or affected by pekapeka and tara iti conservation, including DOC managers, scientists, and field staff, iwi who were *kaitiaki* (guardians) to the pekapeka and tara iti populations involved, community volunteer groups, trusts, landowners, and researchers, and asked them all to send a representative to workshops (n = 16 people for pekapeka; n = 42 people for tara iti). Participants committed to working together to seek a feasible solution (Gregory et al. 2012a). Meetings were held in non-academic spaces, such as *marae* (Māori meeting houses or complexes), iwi offices, and a sports complex (except for two rounds of expert elicitation run at a local DOC office). Ground rules were in place to ensure all voices were heard. Our first step was to develop a shared description of the decision context, identifying scale, scope, and roles in the process.

Elicitation of Values and Alternatives

To identify fundamental objectives, we started by asking participants individually about their values in the decision context (Appendix A). Anonymous responses were collected by the facilitators, then summarised and shared with the group. Participants then worked in small subgroups to refine and structure their objectives, isolating those objectives that were fundamentally important (Gregory et al. 2012a). Subgroups edited these into concise statements describing the objective and desired direction of change and developed ideas of how the objectives might be measured using natural, constructed, or proxy metrics (Gregory et al. (2012a)). Sub-groups provided feedback on their candidate objectives and performance measures to the entire group; similar objectives were combined to generate a final agreed list.

To elicit alternative management strategies, the groups brainstormed possible actions with an influence diagram to show key relationships between threats and management (Gregory et al. 2012a). Participants then worked in small subgroups to combine individual actions into complex strategies. Subgroups described their chosen strategies to the entire group, which discussed them and developed a set of comprehensive strategies. Facilitators further consulted stakeholders about strategies afterward.

Prediction of Consequences and Evaluation of Trade-Offs

To predict the consequences of alternative strategies, expert working groups for each objective self-identified. Facilitators and expert groups used multiple data sources and analyses to estimate consequences (Table 2.1). To estimate biological consequences, we used empirical data and formal expert elicitation to parameterize demographic models. For economic and social objectives, expert groups shared knowledge, conducted research, and finalized outcomes through deliberation. For detailed methods, see Appendix A.

For the mātauranga Māori objective, Ngāti Manuhiri representatives led a *kōrero* (conversation) with facilitators during the initial pekapeka workshop, whereas nominated representatives from Te Uri o Hau, Ngāti Whātua o Kaipara, and Ngāti Manuhiri hosted an expert meeting at the Ngāti Whātua o Kaipara office for kōrero about tara iti. Kōrero sought to address linguistic and biological uncertainties: first, that mātauranga Māori in the respective decision contexts had been captured appropriately, second, that all elements in the proposed

Table 2.1. Fundamental objectives and their performance measures as defined by the stakeholders involved in pekapeka translocation planning from te hauturu-o-toi and tara iti recovery planning and data sources and analyses used to predict consequences for each objective.

Objective	Performance measure	Data source and analysis
Pekapeka translocation planning		
Increase persistence of the translocated subspecies	probability of extinction in 50 years	Probabilities derived from an age-structured population model (Dennis, 2019) using expert elicited vital rate parameter estimates. Elicitation followed a modified Delphi approach as recommended by Hemming et al. (2018).
Reduce impact on the source bat population	probability of extinction in 50 years	
Minimize cost of translocation	total cost of translocation (NZ$)	Costs were obtained from experts with prior experience with different components of bat monitoring and translocation.
Enhance mātauranga Māori (Māori knowledge and perspectives)	subjective scale (from bad to much better)	Assessment of alternatives with regards to mātauranga Māori was carried out by iwi (Māori tribe) experts representing Ngāti Manuhiri
Increase advocacy for species	number of visitors to site per year	Participant knowledge and internet searches of visitor numbers to proposed sites in the 12 months before the workshop.
Tara iti recovery planning		
Increase viability of wild tara iti population	probability of extinction in 50 years (extinction defined as number of adult females is ≤2) population size of Tara Iti in wild	Probabilities and population sizes derived from an age-structured population model developed by T.M.H. using expert-elicited vital-rate parameter estimates. Elicitation followed a modified Delphi approach as recommended by Hemming et al. (2018).
Integrate mātauranga Māori	How well incorporated, and therefore how acceptable (not acceptable, acceptable if certain actions removed [partly acceptable], acceptable)	Assessment of alternatives with regards to mātauranga Māori as interpreted by iwi experts representing Te Uri o Hau, Ngāti Whātua o Kaipara, and Ngāti Manuhiri .
Increase wider ecosystem benefits from tara iti management	number of breeding pairs of local key bird species	Estimates derived from expert-elicited data. Elicitation followed a modified Delphi approach as before and with wider ecosystem experts.
Reduce cost of management	NZ$ per annum	Costs were obtained from an expert group with prior experience costing tara iti and species management.
Increase awareness/respect of tara iti amongst New Zealanders	Media stories count	Expert group concluded that strategies (and thus consequences) would be common across all alternatives, so this objective was not pursued.

strategies were clear to all. Then, iwi explained how they saw each alternative strategy affecting mātauranga Māori and how they would like to compare the alternatives and communicate this to the wider group.

The predicted outcomes for each strategy against all objectives were then summarised in a consequence table for evaluation. There are multiple tools to assist trade-off choices (Gregory et al. 2012a). The pekapeka group decided to use simple multi-attribute rating technique (Keeney & Raiffa (1993); Appendix A) to identify the strategy that provided the best outcomes across objectives. All participants expressed their preferences as 0–100 weights on individual objectives. Quantitative predictions for each action (Table 2.2) were normalised and weighted by the elicited preferences of objectives to obtain an aggregate score across all objectives for each representative and an aggregate group score based on the average of the group weights (the group agreed this would only be used to help the discussion). Because Ngāti Manuhiri preferred to express their assessment of mātauranga Māori verbally rather than numerically, their qualitative judgments for different strategies could not be included in the aggregate scores; instead, they were placed alongside the numerical analysis, and the group then discussed results. The tara iti group chose a simpler approach, simplifying the consequence table with hard constraints that the group agreed on. Alternatives that did not meet certain criteria or fell below certain thresholds were eliminated, leaving a few alternatives to be selected.

2.3 Results

Decision context

The pekapeka group agreed that four decisions were needed regarding pekapeka translocation: which subspecies to translocate, where to source individuals, where to release individuals, and what methods to employ. Decisions made during the SDM process would form the basis of a permit application to DOC. The working group tasked with developing the decisions included the key stakeholders normally approached during permit evaluation and consultation by DOC and as such provided a collective view to submit for approval. This process would allow the DOC director to make decisions based on advice provided by a wider group of stakeholders.

The tara iti group agreed a decision was needed about which management strategy to employ for tara iti within the current range of the remnant population. Like the pekapeka example, the SDM process included the key stakeholders normally approached by DOC for management strategy consultation and approvals (detailed in Methods above). In this way the group viewed the process as empowering and efficient, allowing DOC (the decision maker) to fully endorse the recovery plans recommended by the group and all stakeholders to coordinate action as quickly as possible.

Table 2.2. Indicative consequence table with expected outcomes of a subset of proposed strategies for three of final ten pekapeka translocation strategies.

Alternative	**Objective**					
	Persistence of subspecies	Persistence of source	Cost	Advocacy	Aggregate score (SMART excluding mātauranga Māori)	mātauranga Māori
	P(ext) t=50 years[a]	P(ext) t=50 years[a]	Total NZ$	N of annual visitors		Scale (see text)
1. Hen Island (Hauturu-o-Toi to Taranga): capture pregnant females and pup in captivity at source, release females at source, move juveniles to destination and hold for a period plus supplementary feed to anchor them	0.000†	0.006†	124,540	811	0.57†	Good
2. Codfish Island to Secretary Island (southern subspecies‡). Capture pre- and postflight juveniles, move and release at destination as in 1	0.000†	0.007	103,620	235	0.55	Much better†
3. Hauturu-o-Toi to Hunua Ranges: capture preflight juveniles and move and release as destination as in 1	0.007	0.007	96,540†	70775†	0.22	Bad

†Best outcome for each objective respectively and for the aggregate score weighted by objective preferences (using simple multi-attribute rating technique). ‡Strategy ultimately supported by the group. [a] Probability of population extinction after 50 years.

Objectives and values

The pekapeka group identified five fundamental objectives (Table 2.1). They recognised the importance of establishing a new population while avoiding harm to the source population. They also recognised three non-biological fundamental objectives: minimise management costs, increase advocacy for the species, and enhance mātauranga Māori values. Mātauranga Māori, as viewed by Ngāti Manuhiri, was shared with Pākehā participants through kōrero, to ensure understanding across the group. Through this process of ako, the group learned that Ngāti Manuhiri values of mātauranga Māori are centred around *mauri*, a life principle that reflects vital essence of life or well-being. Mauri is influenced by at least four major factors, *whakapapa* (genealogy), *tapu* (the sacred or prohibited), *noa* (the ordinary or unrestricted, opposite of tapu), and *kaitiakitanga* (guardianship). This kōrero clarified links between these values and conservation management actions. For example, whakapapa is about connections and location: translocations that move animals within the rohe of an iwi would be viewed as better than those that move them outside. Moves outside of the rohe could also be good if they enhanced known ties between *hapū* (subtribes) or iwi. Alternatively, they could be good as a form of *utu* (paying it forward) by creating ties between hapū or iwi (i.e., *toro mai, toro atū* [reciprocity where good actions encourage an appropriate response for balance]). Similarly, whakapapa influenced the view of alternative methods of moving individual animals. For example, keeping family units together may be viewed as better than splitting parents from young or males from females. Tapu and noa were presented as rules of good behaviour, and the group learned to see them as advice for health and safety, normally with binary (i.e., yes or no) answers. For example, some translocation options may be tapu if entry to destination sites is forbidden.

The group recognised mātauranga Māori as a fundamental objective of tara iti recovery planning, alongside four other environmental, economic, and social objectives (Table 2.1). Iwi representatives outlined cornerstone values of mātauranga Māori to the whole group, described the Māori view for tara iti, and defined key terms relevant to recovery planning (Table 2.4). We found that this recognition set a positive tone for workshop conversations among all partners, providing opportunities at each step for meaningful, open kōrero of ideas against all values to build (or renew) relationships and promote ako. Participants at the workshop noted it was the first time they had “shared information” in almost a decade.

Alternative strategies and consequences

The pekapeka group developed ten alternative translocation strategies, each with a source population, destination site, and translocation method (subset shown in Table 2.2). When filling the consequence table, Ngāti Manuhiri representatives chose a subjective scale, from bad to much better, to formally assess the strategies. For example, a strategy was viewed as bad (relative to others) because animals would be moved outside of the rohe to a release site without close ties to the receiving iwi. By comparison, good alternatives would release juveniles and mothers together (enhancing whakapapa by keeping family units together) at destinations within the rohe or with close ties to neighbouring hapū.

The tara iti group initially developed six alternative strategies (which grew to eight, Chapter 4; a subset shown in Table 2.3). As with the pekapeka example, iwi partners preferred a qualitative description of how well the alternatives integrated with mātauranga Māori. They explained that the way they think about them is more of a feeling and cannot be ranked on a scale. In this case, they stated that most alternatives were acceptable, except the ones that contained disagreeable actions. If disagreeable actions were removed, however, then the strategies would become acceptable. For example, they explained that use of herbicides was of concern because "everything is connected," and there could be unknown negative impacts on other living things. Breaking up pairs and bringing infertile individuals into captivity permanently was not agreeable because it obstructed whakapapa (e.g., alternative 1, Table 2.3). In contrast, use of foster pairs aligned well due to its similarity with *whāngai* (adoption) in Māori culture.

Trade-offs and decision making

The pekapeka group used the weights and aggregate scores as a guide to stimulate kōrero and rank alternatives across objectives, except mātauranga Māori and then to compare them to this latter objective. We found no major trade-offs between mātauranga Māori and the aggregate score (Table 2.2). Therefore, the discussion was relatively straightforward; no further analysis was required. The alternative with the highest aggregate score was to translocate the northern subspecies from Te Hauturu-o-Toi. However, risk aversion by iwi and other representatives meant the group did not select this alternative. Instead, they selected the action with the second-highest aggregate score (alternative 2, Table 2.2), which was the preferred choice in terms of mātauranga Māori and focused on a more abundant subspecies. Translocation would still benefit this subspecies but incur less disease-related risk, while further developing translocation techniques that would eventually benefit the northern subspecies. This shift from the initial focus means the SDM process must be repeated with additional iwi who are kaitiaki for the suggested sub-species.

Table 2.3. Indicative consequence table with expected outcomes of a subset of proposed strategies for three of eight final tara iti recovery strategies.

Alternative	**Objectives**				
	Persistence of tara iti in wild	Population size of tara iti in wild	Cost	Change in northern NZ dotterel breeding population	Mātauranga Māori
	P(ext) t=50 years[a]	Mean N (females) t=50 years[b]	Annual NZ$ spent (millions)	% change in number of breeding pairs	Scale (see text)
1. Field 1 + captive 2: lower-intensity field management, lower intensity harvest, captive rear and release within current range, infertile males brought into captivity	0.12	20	0.78	+15	Part acceptable
2. Field 2 + captive 3 + keeping infertile males: Higher-intensity field management, higher-intensity harvest, captive rear and release inside and outside current range, infertile males remain available as foster parents	0.04	31†	1.29	+36†	Acceptable†
3. Field 2 + OZFT‡: Higher-intensity field management, single and infertile clutches supplemented with Australian fairy tern eggs	0.02†	31†	0.47†	+27	Not acceptable

†Best outcome for each objective respectively. ‡Australian fairy tern. [a] Probability of population extinction after 50 years. [b] Mean number of adult females in population after 50 years.

The tara iti group agreed to use acceptability as a hard constraint on mātauranga Māori to simplify the decision. For example, when it became apparent that removing infertile males was biologically favourable yet was deemed unacceptable by iwi, the entire group agreed to add a modified strategy that left infertile males in place (all other actions were unchanged) in the consequence table (alternative 2, Table 2.3). Partners acknowledged that Western science and mātauranga Māori had not always worked well together previously but would if the group continued in this way. Critically, iwi partners were able to have input in the decision making process at each step (Fig. 2.1), leading to shared understanding of objectives, co-development of alternatives, and simple resolution of trade-offs. This process is ongoing but has initiated a lot of positive kōrero and through this ako**.** All partners have demonstrated a willingness to work together in a *mana*-enhancing way (enhancing authority, prestige, influence) that addresses cultural imbalance.

2.4 Discussion

In our case studies, SDM helped the planning process move away from traditional unilateral methods, overcome barriers to inclusivity, and explicitly include diverse value systems such as mātauranga Māori in decision making for conservation of threatened species. This makes for fairer, inclusive decisions, which realises the legal mandate set out in the Treaty of Waitangi and in the Conservation Act (New Zealand Government 1975). We thoroughly recommend SDM for providing the space and support for meaningful kōrero and ako, vital components of good relationships and inclusive decision making.

Simply collaborating with indigenous people or recognising indigenous values does not mean their values are automatically incorporated in decisions as effectively as others (Jackson 2006; Wheeler et al. 2020; Zafra-Calvo et al. 2020). In resource management, progressive steps are being taken to recognise mātauranga Māori, such as iwi management plans as starting points for engagement (Thompson-Fawcett et al. 2017) and the legal personhood granted to the Whanganui River, reflecting its relation to Māori (*Ko au te Awa, ko te Awa ko au,* [I am the river, and the river is me]) (Whanganui River Maori Trust Board 2014). Yet, these still address indigenous values separately from scientific or Pākehā ones. We found SDM helpful because it required a clear, initial expression of values as objectives. In both case studies, the articulation and discussion of iwi values allowed us to co-develop a set of alternatives that already considered scientific, social, and cultural values. This contrasts with the traditional model of scientists preparing alternatives to be judged a posteriori. Ignoring fundamental values at the outset risks developing a set of alternatives, and ultimately decisions, that may be insensitive to social or cultural values. In this sense, during kōrero the pekapeka group realized that it was critically important not only which plan was developed, but also how it was developed. Ngāti Manuhiri viewed the SDM methods as enhancing mana and enabling people to enact their responsibilities as kaitiaki. All partners involved expressed hope that future steps would continue the mana-enhancing process.

A major challenge in our case studies was to express spiritual or cultural feelings to allow comparison with science-based metrics. Facilitators and groups listened to iwi and co-developed qualitative, verbal expressions to judge how well alternative strategies incorporated mātauranga Māori (Table 2.2, Table 2.3). Similarly, Ngāi Tahu and Aotearoa-New Zealand's Ministry for the Environment developed a cultural health index to evaluate river health that encompasses both physical and spiritual values in land and water to be integrated into decision-making with water managers (Tipa & Teirney 2006). Taking time to develop performance measures with partners is critical to inclusivity because it allows cultural values to be described appropriately and treated in the same way as common values, such as species persistence or cost (Gregory et al. 2012a).

In both case studies, we assessed alternatives against all objectives in parallel. This helped eliminate the implicit sense of ranking that would result if, for example, actions were first selected based on biological analyses and then submitted for approval from indigenous groups. Instead, our decision making clearly presented the impacts of each alternative on all stakeholder values (Table 2.2, Table 2.3). Scientific evidence predicting biological consequences of management was presented alongside, not before, consequence assessments for the other objectives. Done this way, evidence-based conservation is much more inclusive.

Co-development and parallel assessment of alternatives yielded another considerable benefit. Before engaging in SDM, some biological experts presumed that opening the co-management process might require considerable compromises, such as accepting sub-optimal biological outcomes to accommodate mātauranga Māori. In the pekapeka case, such conflict did not materialise because the assessment based on mātauranga Māori largely overlapped the aggregate scores based on more traditional Western science-dominated values. For example, sustainability, valued by both conservation science and mātauranga Māori perspectives, was captured in biological viability models and in kaitiaki principles of sustainable use. For tara iti, some compromise was necessary, and was facilitated by co-development of alternatives. Again, the process provided a secure platform for all stakeholders to grow in understanding and seek shared vision despite different backgrounds. Considering both value systems like this improves long-term planning and highlights the nuances of their complementarity. This was captured well in the statement *taha wairua taha tangata* (bringing both worlds together to achieve the objective) (Table 2.4). It echoes the sentiment of the Mi'kmaq people of Eastern

Canada when they found the “two-eyed seeing” (i.e., “learning to use both these eyes together, for the benefit of all”) approach to decision-making to be beneficial (Giles et al. 2016).

Table 2.4. Descriptions and interpretations (right column) of the mātauranga Māori (Māori knowledge and perspective view for tara iti (left column), defined by Te Uri o Hau, Ngāti Whātua o Kaipara, and Ngāti Manuhiri participants at the first Tara iti recovery planning workshop.

Te Ao Māori	Holistic Māori world view
Mauri	The binding force or essence that holds together the physical and spiritual components of a being or thing. The mauri of tara iti is diminished and needs to be rebalanced.
Whakapapa	The spiritual connections, lineage, genealogy, direction. It is the connection between humans and the natural world, ecosystems, all flora and fauna etc. We are part of the system, not separate. Everything has whakapapa, our world is built on it. Everything comes from somewhere. It is holistic and integrated and applied to many aspects of life.
Kotahitanga	The oneness, unity of relationships. For Tara iti, it means support and connection with community, schools, conservation groups (planned activities). It is collaborating to achieve objectives.
Kaitiakitanga	A combination of kaitiaki and tikanga and the processes and practices of protecting and looking after the natural environment, the taonga. It involves a set of obligations and responsibilities to those who come before you and those who come after. Kaitiaki are the guardians and the caregivers – everyone has the role of kaitiaki.
Maramataka	To restore systems and knowledge of agricultural productivity, marine and forest gathering, resource management, health, healing, and daily practices that provide sustenance for well-being.
Rāhui	A form of tapu (sacredness) (*tapu*), the practice of protecting or applying restrictions to an area to let resources recover.
Ako	A 2-way learning relationship, transmission of knowledge. Combining science and education with mātauranga Māori (knowledge of both tangible and intangible). Emerging ideas are shared, both are learning and teaching for the benefit of tara iti.
Taha wairua taha tangata	Bringing both worlds together to achieve the objective, the survival of the tara iti.
Urutau	The earth is shifting, things are changing, and we must change with it (i.e., climate change). Evolving the practice - create new *karakia* (prayer) for tara iti with the new unity, upgrades and changes within our time. Acknowledge our relationship with the tara iti.

Through its focus on values, SDM also encourages recognition of context-specific differences, rather than a one-size-fits all approach. Mātauranga Māori is a dynamic belief system, with diverse values among and within hapū and iwi (Whaanga et al. 2017). We already found slightly different interpretations and emphasis of elements of mātauranga Māori between the pekapeka and tara iti cases. We encourage managers not to make broad assumptions about how mātauranga Māori, or any indigenous belief system, is expressed or judged within a given decision.

Structured decision making provides a space for kōrero and ako. This openness improves alignment and inclusivity (Gregory 2016) and improves thinking about threatened species by enabling distillation of the best information available. Co-management meant partners could ask and answer each other's questions, as opposed to simply presenting information. This could be a non-expert Pākehā asking for clarity on the mātauranga Māori objective from iwi experts or a non-expert DOC representative asking a population ecologist to explain extinction probability. In the tara iti case, discussions helped break down long-standing relationship barriers. For example, expressing concerns about negative impacts of tara iti egg management on whakapapa resolved confusion around language used between DOC and other partners. Such relationship building and shared language use are known critical components of successful resource management (Thompson-Fawcett et al. 2017; Boiral et al. 2020).

The relevance and utility to conservationists of the benefits brought about from values-based decision frameworks cannot be understated. More conservationists across many realms are adopting these tenets and finding them to be fair and effective and to deliver robust outcomes for conservation (Bennett et al. 2019; Collier-Robinson et al. 2019; Rayne et al. 2020; Wheeler & Root-Bernstein 2020). Despite this, challenges remain. Well-meaning managers may be confused about what stakeholder values are or how they could be integrated (Jackson 2006) or may be afraid of upsetting partners and so avoid action or become over-cautious (Meek et al. 2015). At worst, managers may consider others' values irrelevant or unnecessary hurdles to species recovery or ignore them completely (Fox et al. 2006; Chapman et al. 2020).

There is still a need for "transformative change" (Wheeler et al. 2020), and although more researchers are recognising and eliciting values, there is still a scarcity of examples demonstrating their integration in decision-making outside North America (Dam Lam et al. 2019; Zafra-Calvo et al. 2020). We encourage managers to recognise the complexity of decision making in conservation and embrace value pluralism by using relevant expertise because it generates a much deeper understanding of a system and promotes shared, well-supported decisions (Bennett et al. 2019). Our results highlight that inclusivity need not compromise use of the best available scientific evidence. However, relationships with indigenous groups require time to be built (or mended), to share information and accommodate different ways of working together. Financially supporting indigenous representatives and allowing time to participate is essential (Cisternas et al. 2019; Wheeler et al. 2020) and was echoed by iwi representatives. Both would improve capacity for communities to engage meaningfully in decision-making processes (Thompson-Fawcett et al. 2017). Finally, co-management is an ongoing process and in some cases will require continual dialogue and participation from all partners to maintain relationships and efficacy (Gorris 2019).

Conservation continually seeks to become fairer and better, giving indigenous communities more defined, prominent roles in decision-making (Turner et al. 2008; Augustine & Dearden 2014). It also increasingly recognizes that to improve decision-making, meaningful, values-based approaches are needed (Gregory 2016; Mukherjee et al. 2019). Achieving this requires an interdisciplinary approach to clearly express values and identify the best way of achieving them. We are finding SDM provides such a framework. While there is still far to go in reaching widespread successful co-management in Aotearoa-New Zealand and elsewhere, there are reasons to be optimistic. The result will be better outcomes both for species and for all interested partners.

Chapter 3: Challenging assumptions – can seabirds with post-fledging parental care survive after chick translocation?

Abstract

Seabirds with complex foraging behaviours, such as terns and noddies, often have prolonged periods of post-fledging parental care. It is widely believed that chick conservation translocations are unsuitable *a priori* for these taxa, since that period of care would not be replaced and thus birds could not learn to forage. This limits options for species recovery when other accepted techniques, such as social attraction, would be more challenging. Challenging if this assumption is valid is therefore important. We analyse the first phase of a trial conducted in Mauritius that tested whether chick translocation techniques can be used for two common species with post-fledgling parental care: sooty tern (*Onychoprion fuscatus*) and common noddy (*Anous stolidus*). Translocated chicks were trained to feed independently from trays in response to a feeding stimulus prior to fledging, then this stimulus was used to attract fledged birds to supplementary food. We found that birds fledged successfully, and some birds were surviving and using supplementary food for up to a month after fledging. This study highlights the importance of challenging assumptions and provides an important first step in further developing chick translocation techniques for seabirds with post-fledge parental care.

3.1 Introduction

When it comes to thinking about management alternatives for threatened species, sometimes conservationists limit themselves unnecessarily or without realising. Conservationists might use a dogmatic approach and only consider management choices that are anecdotally ‘known’ to work, which could result in unexpected or negative outcomes (Martínez-Abraín & Oro 2013). Alternatively, they might consult available evidence but anchor on prescribed choices. Where evidence is extensive this is undoubtedly a better option, but without thinking creatively about all possible alternatives it risks missing the best one. If practitioners instead use evidence whilst also thinking creatively, challenging presumed constraints and resisting business as usual, they can increase the chance of creating novel solutions that better meet their objectives (Keeney 1996).

To recover seabird populations, for example, conservationists often attempt to establish new breeding colonies (Jones & Kress 2012). The best action is thought to depend on species’ behaviour, ecology, and the broader management context. Social attraction (e.g., decoys, acoustic lures) is recommended for surface nesters that exhibit post-fledging parental care, (Gummer 2003; Jones & Kress 2012), when there is a nearby colony (< 25 km) to attract birds from, or the source population is increasing (Gummer 2003; Buxton et al. 2016). This approach has been successfully used for species such as Australasian gannets (*Morus serrator*) (Sawyer & Fogle 2013). Chick translocation (active movement of chicks to a new site within indigenous range) is considered appropriate for burrow-nesters or species without post-fledging parental care. It is applied extensively for petrels (Procellariforms) (Miskelly et al. 2009), and can be used in combination with social attraction, e.g. for Atlantic puffins (*Fratercula arctica*) (Kress & Nettleship 1988).

Conversely, conservation experts state that chick translocation techniques are unsuitable for taxa such as terns and noddies (Gummer 2003; Jones & Kress 2012). These taxa have complex foraging behaviours, hunt by hovering, splash-diving or skimming (Ashmole & Tovar 1968; Brown 1976a; Higgins & Davies 1996; Cabot & Nisbet 2013), and often have prolonged periods of post-fledging parental care (Ashmole & Tovar 1968). After fledging, juveniles are thought only to be capable of developing complex foraging skills, and surviving whilst doing so, with parental help (Gummer 2003), for example being accompanied to good foraging spots or being fed (Ainley & Boekelheide 1986; Stienen & Brenninkmeijer 2002; Preddey 2008).

Perhaps as a consequence, chick translocation techniques have almost never been tested, with no published examples known to the authors.

Conservationists managing threatened seabird species must think carefully (Buxton et al. 2016; Moehrenschlager & Lloyd 2016) and expansively (Game et al. 2014) about which set of tools to use. In an example of creative alternatives development, the Mauritian Wildlife Foundation (MWF) began trialling chick translocations for two common tropical seabird species that are thought to exhibit post-fledging parental care: sooty tern (*Onychoprion fuscatus*) and common noddy (*Anous stolidus*), neither of which have been translocated anywhere else before. Using these common species, MWF wanted to test a novel action to replace parental feeds in free flying birds, for later use on rarer species in the region, such as roseate tern *Sterna dougllii* and Abbott's booby *Papasula abbotti* (Mauritian Wildlife Foundation, 2013). Learning whether this constraint could be overcome using supportive management could also inform planning for tara iti (New Zealand fairy tern, *Sternula nereis davisae),* for which captive rearing and releases is often advocated (Chapter 4; Hansen 2006; Maloney et al. 2017). Here, we investigate whether the first phase of this trial was successful in using pre-release training and supplementary food to allow translocated birds to survive and feed post-fledging without parental care.

3.2 Methods

Study species

Sooty tern and common noddy are long-lived tropical seabirds with a global distribution. Their conservation status is Least Concern (BirdLife International 2021). Sooty tern adults weigh 170–285 g and have a wingspan of 82–94 cm, and adult common noddy weigh approximately 171–185 g and have a wingspan of 79–86 cm (Chardine & Morris 1989; Higgins & Davies 1996). Both have a carnivorous diet, and feed by dipping or plunging immediately below the sea surface (Higgins & Davies 1996).

We used published data from other populations (outside Mauritius) on chick development, fledging and post-fledging behaviour to provide context to the results of the translocation trials (Table 3.1). Fledged sooty terns are fed at their natal colony for up to 21 days (i.e. until they are approximately 73–97 days old; Feare (1975); Brown (1976)). In this period, they spend the day at sea and return to be fed at night (Brown 1976a). After this, birds depart the colony with at least one adult and do not return. Birds have been observed in adult-juvenile groups hundreds of kilometres away from colonies, but juveniles have not been observed being fed by parents at sea (Feare 1975; Ainley & Boekelheide 1986). Fledged common noddy young appear to continue attending the colony for longer. Young have been seen receiving parental feeds at their natal colony as late as 100 days after fledging (approximately 143–156 days old), mostly at night (Brown 1976a). No data are available on parental care after this period.

Table 3.1. Known data on biology and behaviour of sooty tern and common noddy chicks and fledglings. Older chicks are defined as aged 30 days to fledging.

Species	**Chick asymptotic weight (g)**	**Age at asymptotic weight (days)**	**Chick fledge weight (g)**	**Parental care of older chicks (feed frequency, weight)**	**Flying age (days)**	**Fledging age (days)**	**Length of PFPC[1] (days, est. age in days)**
Sooty tern	180-190 [3, a]	40-50 [6, a]	188-196 [5, c]	0.79/day, 25.1 g (± 9.6, 1 S.D.) /feed [2, a]	> 28 [3, a, 7]	> 56 [6] 56 – 70 [5, c]	17, 77 – 91 [5,c] 21, 73 – 97 [3,a]
Common noddy	200 [8, a] 171-182 [4, b]	34 [8, a] 30 [9, b]	195.8 (± 18.4, 1 S.D.) [8, a]	0.7-0.9/day, 15.5 – 20.3 g /feed [2, a]	> 35 [7]	43 – 49 [9,b] 49 – 56 [7]	100, 143 – 156 [3,a] 33 – 88, 76 – 144 [8,a]

[1] PFPC denotes post-fledging parental care. References: 2. Brown (1975); 3. Brown (1976a, 1976b); 4. Chardine & Morris (1989); 5. Feare (1975); 6. Feare et al. (2013); 7. Higgins & Davies (1996); 8. Megyesi & Griffin (1996); 9. Morris & Chardine (1992). Letters denote location, if specified: a. Hawaii; b. Puerto Rico; c. Seychelles.

Translocation

We analysed the outcomes for two translocations in 2013 and 2015 (Table 3.2). The source colonies were on Île aux Serpents, a 19-ha volcanic cone with an elevation of 162 m. It is the most northerly and remote of Mauritius' offshore islands and it supports approximately 200,000–300,000 sooty terns and 20,000–30,000 common noddies (Mauritian Wildlife Foundation, 2013). The destination site was 26-ha Île aux Aigrettes , managed by MWF since 1984. It is a low-lying coral island with dry coastal forest, 850 m off the south-east coast of Mauritius and 67 km from Île aux Serpents. Île aux Aigrettes is considered a suitable destination site due to the presence of seabird sub-fossils and historical records (Mauritian

Wildlife Foundation, 2013). Chicks were harvested from Île aux Serpents based on healthy appearance, approximate size and feather development. Time was limited at the source colony, so exact chick age at harvest was varied and unknown. Harvested birds were transported in aerated boxes, by boat then jeep, taking approximately 4 hours.

Chick monitoring and rearing

On arrival to Île aux Aigrettes, translocated birds were rehydrated and released into a small enclosure with shelters (except smaller chicks, which are initially fed more frequently). Birds were fed twice a day on squid, fish, and octopus. Field staff initially fed individual birds by hand, until birds could feed communally from trays. Birds were conditioned to associate feeds with a whistle blown immediately prior to each hand-feed, then before communal feeds. Once the birds fledged, the whistle was blown loudly to attract birds in flight to the food trays until no birds were seen returning. Field staff monitored pre-feed chick weights and development (methods varied between years), and in 2013, they also estimated age of chicks. We calculated the average amount of feed taken daily as the total mass of food taken (including morning and afternoon, plus individual feeds) divided by the count of birds recorded. In 2013, because noddies and terns mixed at food trays, the average was calculated for both species combined. Birds were counted each day and individuals were accounted for when possible.

3.3 Results

In March 2013 MWF translocated 17 common noddy and 21 sooty tern. The cohort were estimated to be between 18–50 (sooty tern) and 12–39 (common noddy) days old, except two sooty tern chicks and three common noddy smaller than the rest, which were estimated age 10–13 and 3–8 days old, respectively. In September 2015, MWF translocated a further 29 sooty tern (ages were not estimated in this year). Across both these translocations, 95–100% of translocated individuals fledged successfully (Table 3.2). We note a failed translocation of these species in 2014, that we have not analysed further due to poor data (only seven (24%) sooty tern and one (9%) common noddy reached fledging age (Mauritian Wildlife Foundation 2015)).

Table 3.2. Summary data of translocated common noddy chicks in 2013 and sooty tern chicks in 2013 and 2015.

Species (Year)	**Cohort size**	**No. fledged (percent of cohort)**	**Mean first weight (g) (CI*)**	**Mean fledge weight (g) (CI)**	**Mean estimated trans.† age in days (CI)**	**Mean days on IAA before fledging (CI)**	**Mean estimated fledging age in days (CI)**	**Mean no. days using communal feeds post-fledge (CI)**
Common noddy (2013)	17	17 (100%)	98.2 (49 – 148)	189.6 (172 – 213.8)	17.6 (4.2 – 36.6)	44.5 (29.5 – 59.4)	62.2 (57.2 – 67.6)	14.6 (0 – 33)
Sooty tern (2013)	21	20 (95%)	104.3 (72.5 – 130)	166.5 (131.9 – 195.2)	29.5 (11.5 – 48)	39.5 (22.5 – 52.2)	68 (53.5 – 79)	NA‡
Sooty tern (2015)	29	28 (97%)	141.3 (96.68 – 225.6)	161.5 (151 – 176)	NA ‡	46 (39.1 – 50.7)	NA‡	17.4 (8.4 – 27)

* CI denotes non-parametric confidence interval of 2.5th - 97.5th percentile range. † translocated. ‡ Data not available.

Feeding

Individuals were hand-fed for 15 days in 2013, and for 8 days in 2015. After this and depending on developmental stage, birds transitioned to communal feeds. The mean daily consumption per bird in 2013 (both species) was 84.7 g ± 52.8 S.D (n = 120) and in 2015 (sooty tern) was 70.7 g ± 32.9 S.D. (n = 74). This is higher than observed in the wild for older chicks (Table 3.1) but is confounded as our data includes fledged birds (for which there are no known data for wild birds). Daily consumption was relatively constant with a slight increase towards birds departing, and in 2013, some of the last common noddies to leave the island were taking much more food when offered (Fig. 3.1). In 2013, communal feeds were offered (whistle blown) for 6 days after the last common noddy departed the island (no data for 2015).

Common noddies appeared to visit the feeds intermittently, with counts dropping (sometimes to zero) for several days in a row then increasing again, indicating some birds had begun spending days away from the island (Fig. 3.2). This broadly matches behaviour of wild birds, which may learn to forage away from the colony and return to be fed by parents over a protracted period. Sooty terns seemed to display two different behaviours. When reared with noddies in 2013, some also took trips away from Île aux Aigrettes for a few days at a time (although fewer birds did this; Fig. 3.3). However, in 2015, sooty terns were mostly present every day until numbers began dropping off gradually (Fig. 3.3); this pattern more closely matches wild behaviour, where sooty terns leave the colony permanently after a shorter phase of being fed by parents.

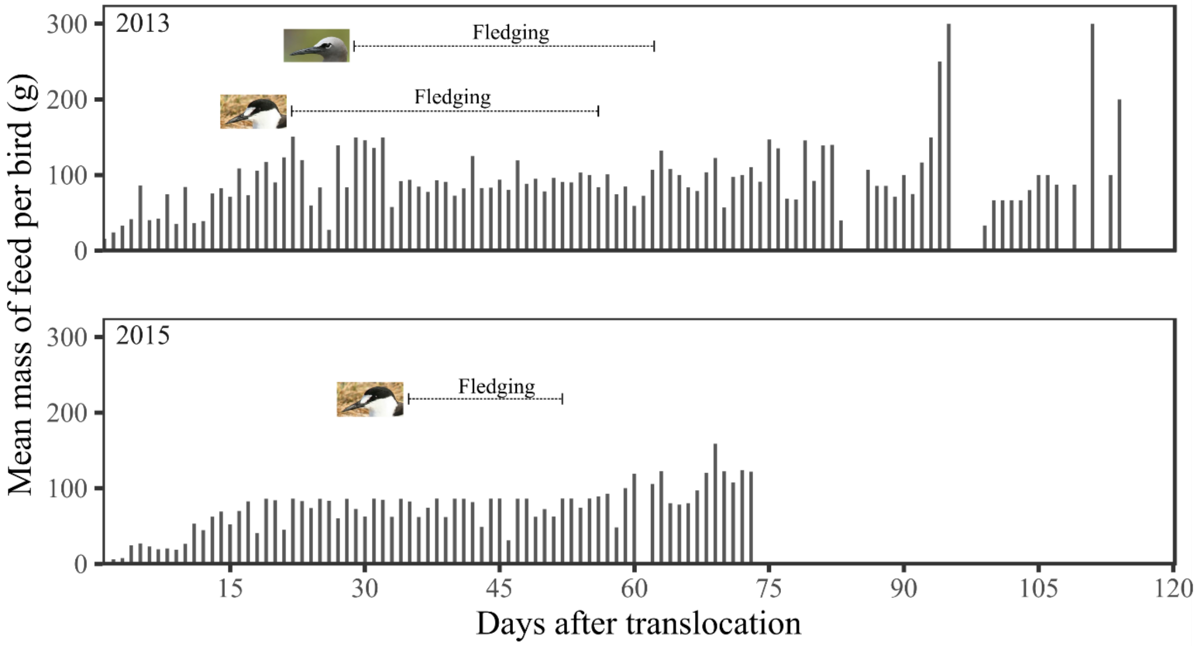

Figure 3.1. Mean mass (grams) of food consumed per bird (includes hand-feeds and communal feeds), per day, in each season. In 2013, common noddy and sooty terns fed together so the amounts are combined; 2015 represents sooty terns only. Birds transitioned to communal feeds after 15 days in 2013, and after 8 days in 2015. In 2013, fledging period for common noddies was 29–63, and for sooty terns was 22–56 days after translocation. In 2015, fledging period for sooty terns was 35–52 days after translocation.

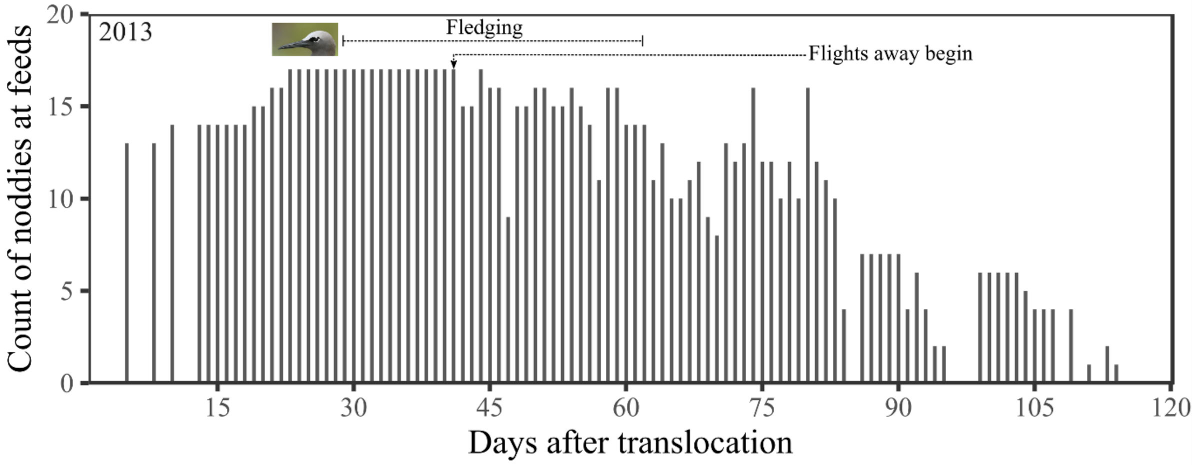

Figure 3.2. Count of common noddies at the communal feeds on Île aux Aigrettes after translocation in 2013. Fledging period was 29–63 days after translocation. Birds start taking flights away from Île aux Aigrettes for more than a day 41 days after translocation.

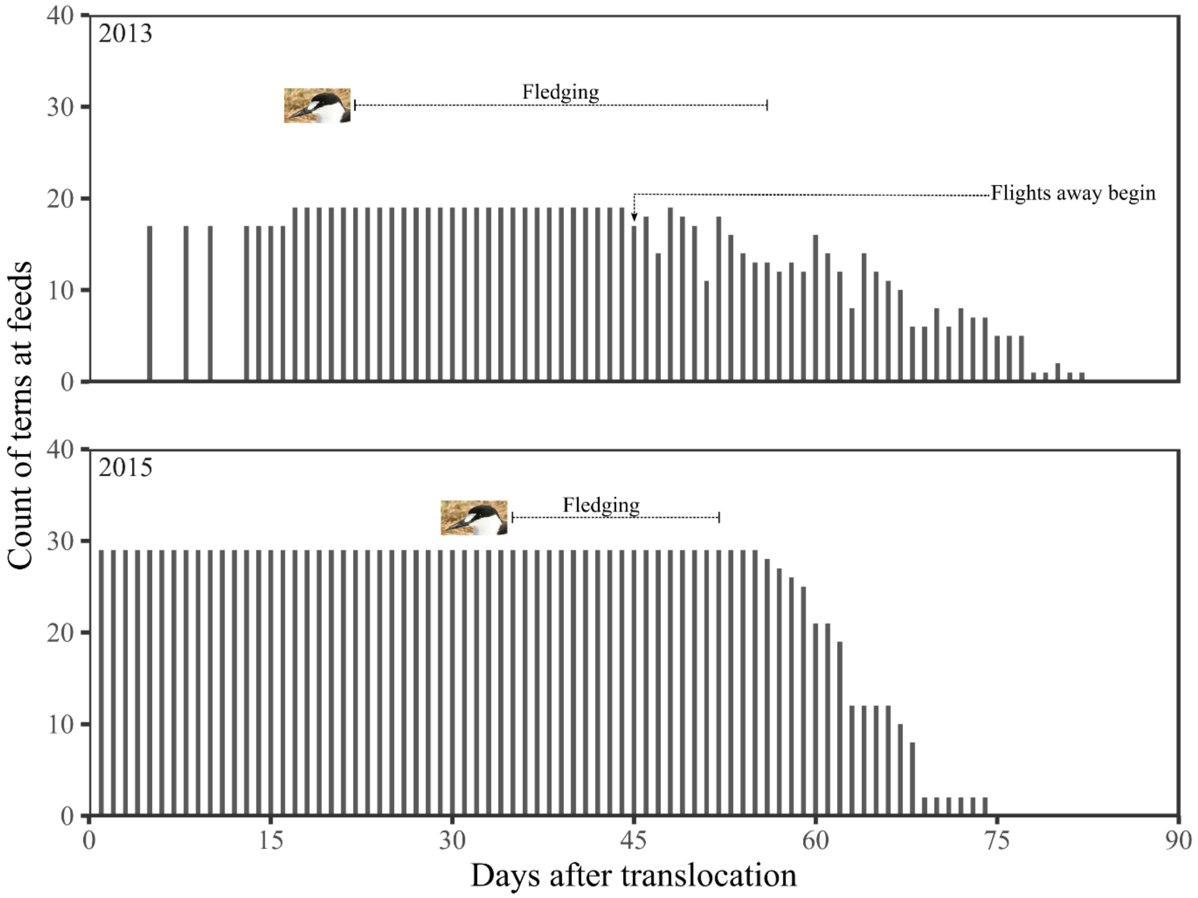

Figure 3.3. Count of sooty terns at the communal feeds on Île aux Aigrettes after translocation in 2013 and 2015. In 2013, fledging period was 22–56 days after translocation, and in 2015, it was 35–52 days after translocation. In 2013, birds start taking flights away from Île aux Aigrettes for more than a day 45 days after translocation but did not do this in 2015.

After fledging, sooty terns used the communal feed for an average of 17.4 days (2.5–97.5% range = 8.4–27 days; Table 3.2, Fig. 3.4) and for a maximum of 29 days (2015 data only). This matches the period wild birds get fed by parents at the natal colony (Table 3.1). Fledged common noddies used the communal feed for an average of 14.6 days (0–33; Table 3.2, Fig. 3.4) and for a maximum of 35 days (2013). This value is at the lower range of the period that wild birds get fed by parents at the natal colony (Table 3.1).

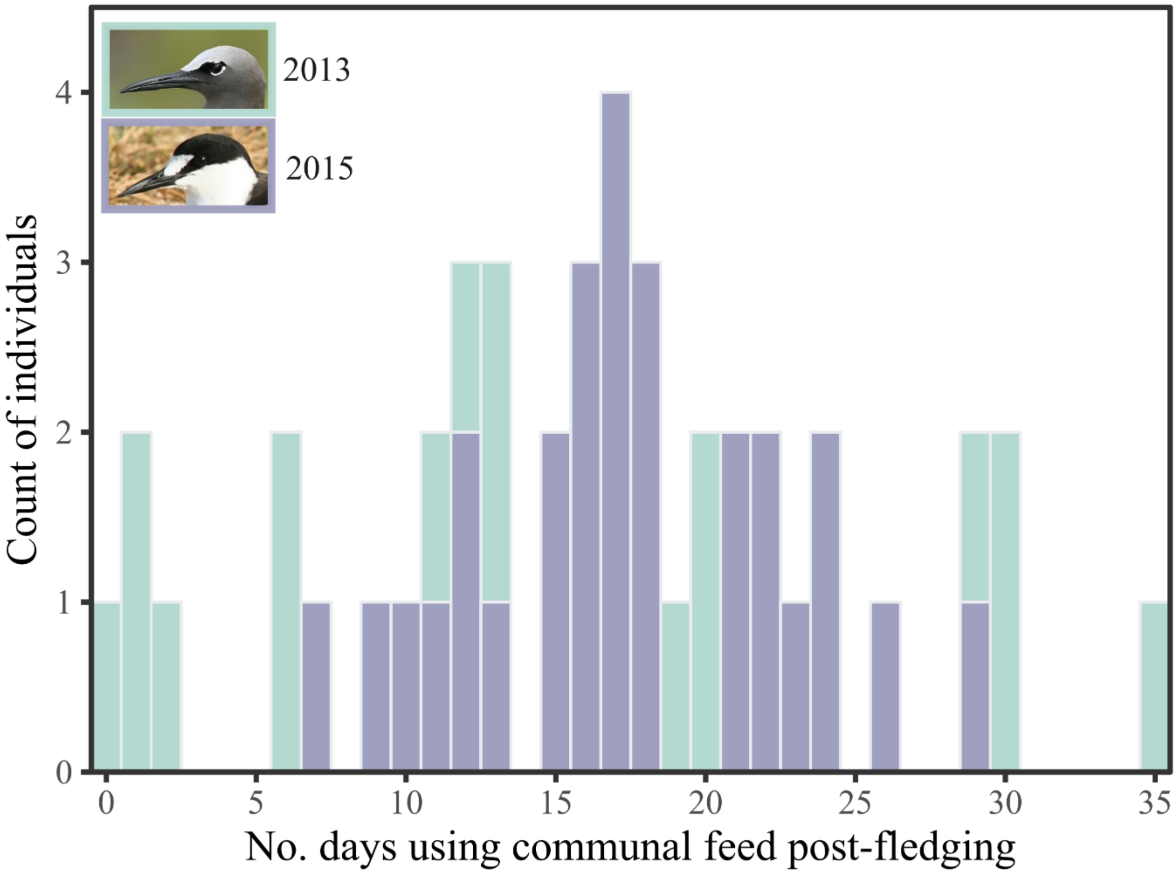

Figure 3.4. Frequency distribution of the total number of days individuals that used the communal feeds for after fledging. Common noddy data from 2013 (pale bars, *n* = 17) and sooty tern data from 2015 (dark bars, *n* = 28). For example, 4 sooty terns used the feeds for 17 days after fledging, 2 common noddies used the feeds for 30 days after fledging.

Growth and fledging

The growth curves of both species in 2013 show that chicks reached asymptotic weight at an older age than wild chicks observed in other locations (Tables 3.1, 3.2, Fig. 3.5). In both years, asymptotic and fledging weight for sooty tern was lower than wild chicks observed elsewhere (Tables 3.1, 3.2, Figs. 3.5, 3.6), but in 2013, common noddies asymptotic and fledging weights were within the range observed in wild chicks (Tables 3.1, 3.2; Fig. 3.5). Mean estimated fledging ages of common noddies (62.2 days old) and sooty terns (68 days old, 2015 data only) were similar to wild chicks studied elsewhere (Tables 3.1, 3.2).

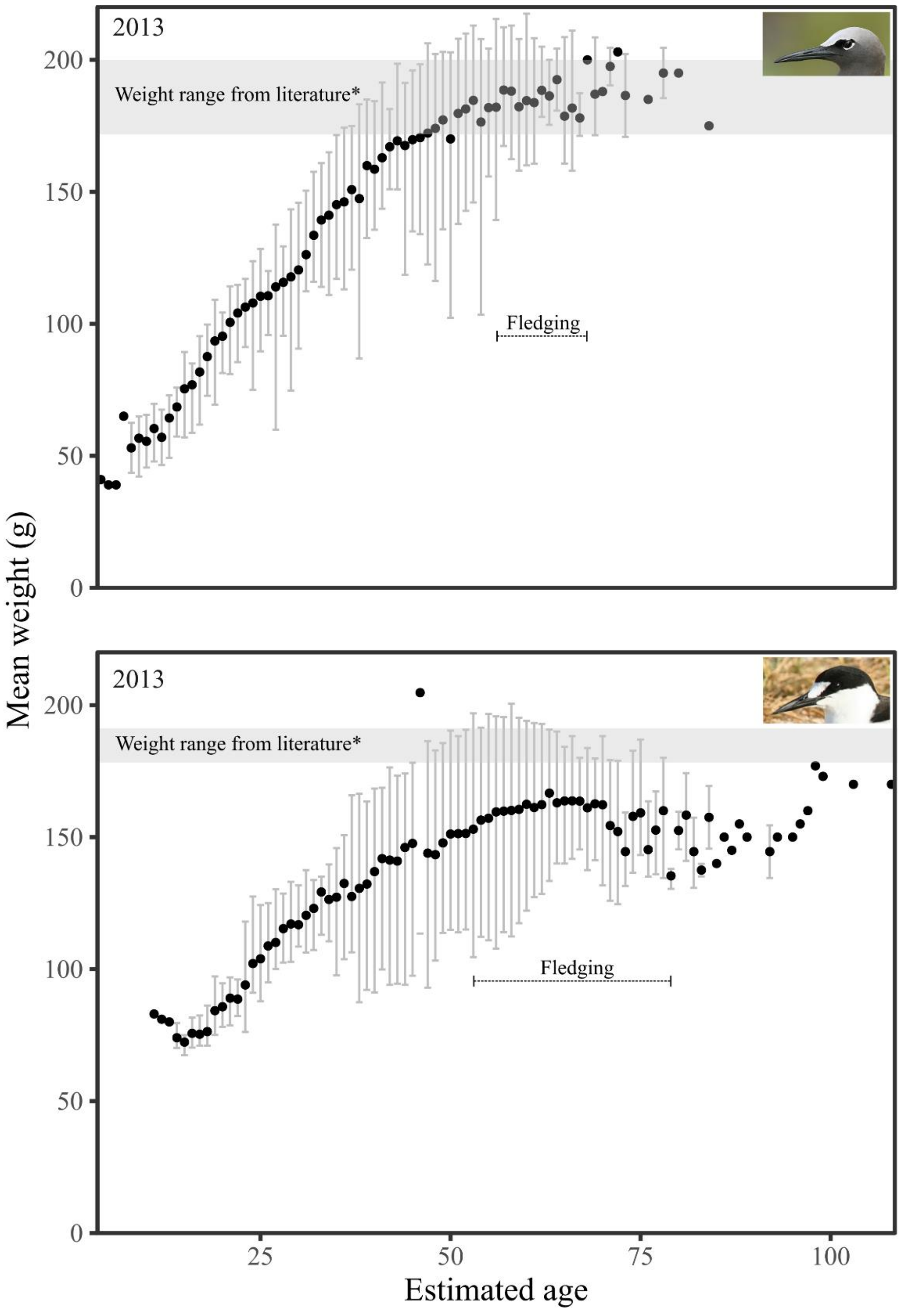

Figure 3.5. Change in body weight post-translocation of common noddies and sooty terns in 2013 as a function of estimated age in days. Points indicate mean weight and error bars represent 2.5th to 97.5th percentile range. In 2013, fledging period for common noddies was 56–68 days old, and for sooty terns was 53–79 days old. *common noddy asymptote weight data from Chardine & Morris (1989) and Megyesi & Griffin (1996), sooty tern data from Brown (1976b), see Table 3.2.

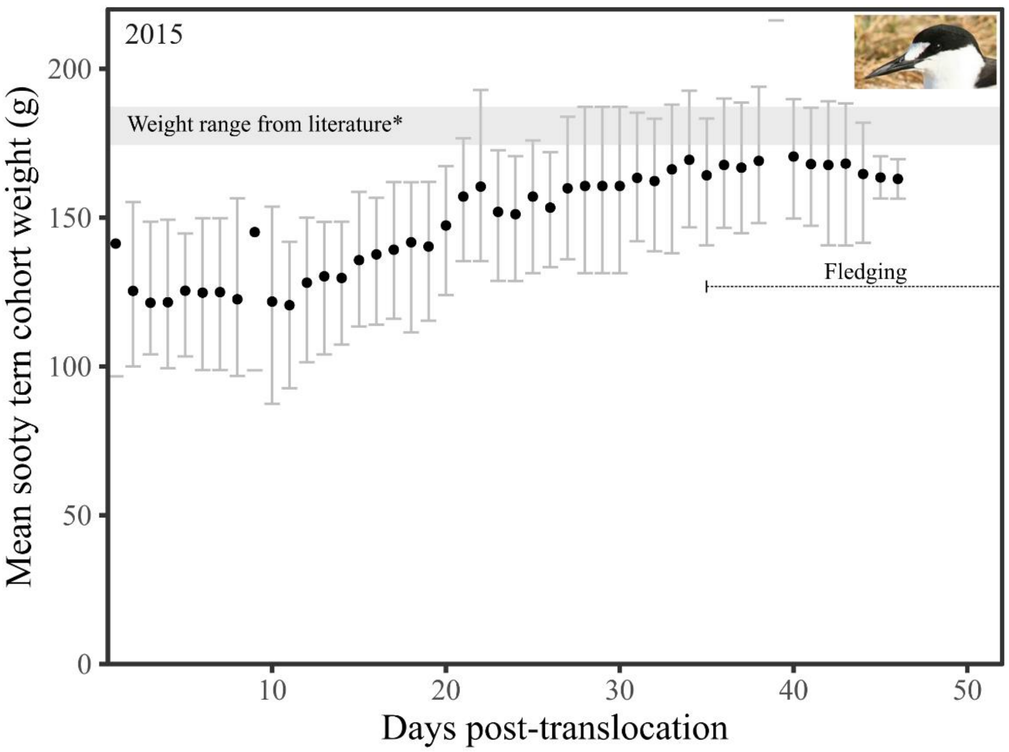

Figure 3.6. Change in body weight post-translocation of sooty terns in 2015 as a function of days post-translocation (estimated age not available). Points indicate mean weight and error bars represent 2.5th to 97.5th percentile range. In 2015, fledging period for sooty terns was 35-52 days after translocation. * sooty tern asymptote weight data from Brown (1976b), see Table 3.2.

Observations of post-fledging behaviour and condition

Birds of both species used communal feeds regularly for up to 35 days post-fledging (Fig. 3.4). A few birds were opportunistically caught post-fledging and were maintaining their weight within the range of fledged weights (Figs. 3.5, 3.6). A few fledglings were observed practicing foraging dives in the lagoon, and some returned with wet feathers. Both species displayed patterns of colony departure similar to what has been observed elsewhere in the wild, but the alternate hypothesis is that they died before full independence. Between August 2015 and April 2016, field staff counted 109 terns and noddies flying around the island, however no individuals have returned to nest on Île aux Aigrettes as of 2021.

3.4 Discussion

Chick translocation is commonly thought to be unfeasible for seabird taxa that display post-fledging parental care, yet our findings challenge this assumption, at least in the initial translocation phase. These results are important to help develop methods for chick translocation of endangered species like tara iti, for which other techniques may be more difficult. We found that chick translocation techniques could be used to successfully fledge two common tropical seabird species that exhibit post-fledging parental care (although we note that this is difficult as the failed 2014 translocation showed). Furthermore, some individuals survived and used supplementary food for up to five weeks after fledging, maintaining their bodyweight. These results suggest translocation is not unfeasible *a priori*; the important next step will be to ascertain and ensure longer term survival of translocated birds. Although no colony has established on Île aux Aigrettes for either species, there has been more sightings of both sooty terns and common noddies nearby the island, including one individual that was ringed (neither species is ringed elsewhere in Mauritius), perhaps indicating some longer-term survival and interest in the destination site.

In this trial, translocated sooty tern and common noddy chicks developed, behaved and survived during a critical period post-fledging similarly to wild birds. We note that while published data from outside Mauritius provided important context in our study, care must be taken, since fledging weight, length of parental care and juvenile survival are affected by local conditions, e.g., intra- and inter-seasonal food availability and weather (Feare 2002; Stienen & Brenninkmeijer 2002). Birds of both species were observed taking trips away from Île aux Aigrettes for several days, without dramatic drops in weight or dramatic increases in consumption, which may suggest they were able to partially self-feed. Nonetheless, translocated sooty tern fledged at lower weights than their wild counterparts, which could have impacted their survival after departing the colony (Feare 2002; Feare & Bristol 2013). Common noddies fledged at similar weights to their wild counterparts, which may improve their chances of survival. Yet, they stopped using communal feeds earlier than we might have expected, in contrast with sooty terns.

For this management alternative to be useful, fledglings not only need to learn how to feed but have to survive and recruit to the destination site. Investigating survival and dispersal after birds depart the destination site will be a critically important next step, which is more accessible now that tracking technology is light enough to use on these taxa (Soanes et al. 2015; Maxwell et al. 2016). Both species take relatively long to start breeding (sooty tern, 5–6 years, Feare & Doherty (2011); common noddy 3–6 years, Morris & Chardine (1995)) and juvenile recruitment is low (e.g., 26% for sooty terns, Feare & Doherty (2011)). Therefore, improving chances of colony establishment might require larger releases and implementation of social attractants (e.g. decoys and acoustic lures) on Île aux Aigrettes (Jones & Kress 2012; Mauritian Wildlife Foundation 2016). Translocation programmes in such conditions require long-term commitment.

The technique of training translocated chicks to a stimulus during feeding, then using the stimulus to supplementary feed fledged young could be relevant for managers of endangered terns and noddies, where small or single populations mean that social attraction techniques in isolation might not be enough to establish new colonies. For example, MWF trialled it for potential use with roseate tern and Abbott's booby, and both tara iti and the Critically Endangered Chinese crested tern *Thalasseus bernsteini* both have estimated population sizes of fewer than 50 (BirdLife International 2021). Tara iti managers previously trialled ad hoc captive-rearing and release of fledged young, yet it failed since none were seen again after release (Gummer 2003). However, chicks have taken supplementary food from trays in the field, and managers may consider a more systematic trial of releases in future (Chapter 4), building on the MWF study presented here. A recent study on Chinese crested tern seemed to rule out chick translocation based on the assumptions we have described earlier (Lu et al. 2020). However, conservation managers also took a novel approach: since social attraction would be difficult, they successfully attracted a common relative, greater crested tern (*T. bergii).* This subsequently attracted the endangered tern, leading to a revised population estimation of ≤ 100 individuals. Since hybridisation of the two species now threatens the rarer tern (Lu et al. 2020), our trials may still be of interest in future.

Challenging assumptions and thinking creatively is valuable for generating alternative management actions without dismissing them *a priori* (Keeney 1996). For example, the translocation of migratory birds was perceived to be impossible without parents present to escort young on migratory routes. Managers of whooping crane (*Grus americana*) reintroductions used a novel approach to overcoming this problem. Trialling first on the more common sandhill cranes *Antigone canadensis* (Urbanek et al. 2005), cranes were imprinted onto field staff and ultralight aircraft, who then flew the migratory route with the cranes (Urbanek et al. 2010). This has been applied successfully in reintroductions for other migratory bird species, such as lesser white-fronted goose (*Anser erythropus*) (Mooij et al. 2008). Of course, successfully challenging specific assumptions does not guarantee success. For whooping cranes, uncertainty still surrounds other associatively learned behaviour and their long-term impacts on breeding ability, possibly jeopardising recovery (Runge et al. 2011b).

Conservationists must celebrate short-term successes and learning but continue to focus on future learning needs to meet their fundamental objectives.

Developing creative alternatives, phased learning and trials using common species are important aspects of challenging assumptions in conservation. The seabird translocations we analysed here are the first of their kind, testing the assumption that seabird chicks with post-fledging parental care cannot be translocated. Our results help advance knowledge and show that it is possible to keep animals alive during a critical developmental stage. Future learning can now focus on post-fledge survival and dispersal. It is important that conservationists do not limit their thinking on alternative management options unnecessarily: as well as utilising and generating scientific evidence, thinking expansively about management alternatives can improve the chance of finding the best option.

Chapter 4: Navigating action, inaction, and conflict in recovery planning for extremely small populations

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Abstract

Decision making in conservation is arguably at its hardest when choosing management for species on the brink of extinction, yet this is when rigorous process is most critical. Conservationists must face the complexity of the situation if they want to avoid management responses that over-estimate success, or that lead to inaction or even entrenched conflict. This can hinder progress and have unintended outcomes. Structured decision making, a framework from the decision sciences, is highly effective in managing complexity as it incorporates multiple stakeholder values and uses predictions that clarify uncertainty. Management planning for tara iti (New Zealand fairy tern, *Sternula nereis davisae*), a Nationally Critical seabird, was at a standstill for ten years. Stakeholders had entrenched, divergent beliefs about the best way forward, with some suggesting novel actions and others concerned that the 'wrong' decision could cause extinction. Using SDM, the tara iti stakeholder group articulated their four fundamental objectives, described eight management alternatives and jointly learned the predicted outcomes (consequences) with uncertainty across these. Rebuilding group unity through participation and information sharing helped move the group away from inaction, and the predicted biological consequences with uncertainty prevented over-optimistic expectations from a 'do everything' approach. Ultimately, recognising multiple objectives changed the decision from that which would have been made under a single-objective, biology-focused approach. The group worked in an inclusive way to make a shared management recommendation after a longstanding period of conflict and inaction. Addressing complexity in a transparent and structured way using decision analysis can deliver clear insight and an agreed, rational, path forward.

4.1 Introduction

Species on the brink of extinction because of extremely small population size embody the 'crisis' discipline that is conservation science. Conservationists need to make difficult decisions and must often choose between intensive management actions with uncertain outcomes and considerable commitments, yet with limited resources (McCarthy et al. 2012; Gerber et al. 2018) and in highly complex socio-ecological contexts (Evans et al. 2017; Law et al. 2018; Martin et al. 2018). Despite inspiring examples (Butchart et al. 2006; Bolam et al. 2021), the complexity of these decisions means that successfully recovering species remains challenging (Butchart et al., 2006, Catalano et al 2020).

Conservationists face major difficulties incorporating complexity when they are planning recovery for extremely small populations. First, they need to look beyond biological considerations and include other management objectives that are fundamental to the decision (Ewen et al. 2014a). This multi-objective view might seem obvious in this context, yet neglecting non-biological considerations risks poorly allocated resources and conflict with other unstated objectives, such as cost (Canessa et al. 2014; Iacona et al. 2018) or cultural values (McMurdo Hamilton et al. 2020). A second challenge is the need to consider a full range of alternative management actions (Dolman et al. 2015; Taylor et al. 2017), predict the outcomes of those actions against key objectives and acknowledge the uncertainty around those predictions (Regan et al. 2005; Converse et al. 2013; Milner-Gulland & Shea 2017). If predictions and uncertainty are not made explicit, it is difficult to understand the consequences of actions, conduct risk analyses, or make trade-offs between objectives, and so decision making becomes vulnerable to biases and hidden value judgements (Keeney 2002; Game et al. 2013). This can lead to irrational decisions or amplify conflict between team or community members on what should be done.

Failure to tackle the full complexity and uncertainty in a decision can be linked to two kinds of management responses. One is an alternatives-focused (Keeney 1996), action-orientated response in which actions, such as reintroduction or supplementary feeding, are implemented broadly without thorough consideration of the problem. Such dogmatic approaches have been criticised in conservation, since they can result in unexpected, negative biological or economic outcomes (Sutherland et al. 2004; Martínez-Abraín & Oro 2013). The other, contrasting kind of response is a slow response, where uncertainty and the desire to avoid management interventions that could hasten extinction lead to managers taking no action, or perpetuating the status quo (Meek et al. 2015). Hammond et al. (1998) explain that a slow response is common because it feels psychologically ‘safer’ than making a decision that might be regretted; it might also feel safer to make a decision once there is more information (e.g. a call for more data, Canessa et al. (2015); Buxton et al. (2020)).

Both responses individually have major ramifications for project success, species conservation status and societal perceptions of conservation utility (Jachowski et al. 2016). When individuals within the same team are responding in these two opposing ways, further conflict between members can occur. This can be hard to resolve if they have opposing beliefs or hypotheses about the ‘right’ management response, and those beliefs are based on vaguely specified predictions against an unstated mix of objectives. This entrenched, complex conflict, if not tackled using the appropriate tools, can hinder recovery planning and implementation for years (Mason et al. 2018). It is important to recognise and limit these common, inherent cognitive responses to complexity in order to improve decision making (Papworth 2017). Structured decision making (SDM) is a particularly useful approach in such situations (Gregory et al. 2012b; Ewen et al. 2014b; Milner-Gulland & Shea 2017; Edwards et al. 2019; Canessa et al. 2020).

Many conservationists advocate for the use of SDM when facing complex problems for small populations, such as whether to implement intensive conservation actions like supplementary feeding (Ewen et al. 2014b; Ferrière et al. 2020), egg management (Edwards et al. 2019) and reintroductions (Smith et al. 2011). SDM is a decision-analytic approach that starts with a clear definition of objectives (values-focused thinking, Keeney (1996)) and explicitly predicts the outcomes (consequences) as well as the uncertainty for each of these, across a range of alternatives (Gregory et al. 2012a) to inform trade-offs. Decision-analytic approaches can be carried out in a transparent, participatory way which provides space for dialogue to address complexity and uncertainty. Embracing SDM can therefore reduce the likelihood of dogmatic responses or inaction.

In this study, we used SDM, including a bespoke population model, to plan recovery of tara iti (New Zealand fairy tern, *Sternula nereis davisae*). Tara iti persist as an extremely small population in the north of Aotearoa-New Zealand. The species recovery team was disbanded in the mid-2000s and a communication breakdown ensued. When it was reformed in 2018, the team decided to choose the best management alternative by explicitly recognising plural values and thereby hoping to neutralise entrenched conflict. We show how considering multiple objectives and predicting consequences with uncertainty across management alternatives influenced the choice of a preferred management, and improved group function and agreement.

4.2 Methods

Tara iti and historical management

Tara iti is a threatened, Nationally Critical (Robertson et al. 2017) non-migratory seabird of Aoteroa-New Zealand that breeds on sandy coastal beaches in Auckland and Northland, and winters in Kaipara Harbour in Northland. Aotearoa-New Zealand’s Department of Conservation (DOC) began management in 1990, starting with monitoring, then adding actions such as predator control (in 1995) and active nest and egg management (e.g. artificial incubation, in 1998; Chapter 5; Ferreira et al. (2005)). The recovery group disbanded in 2006 and the breeding population plateaued at only 8-11 pairs (since 2010). Concurrently, relationships between various stakeholders became acrimonious and communication between DOC, community-based volunteers, charitable trusts and local iwi (Māori indigenous tribes) partners began to deteriorate. In 2017, a DOC-commissioned review concluded that there had been a failure of DOC leadership and called for a reformed recovery group and changed management structure (Maloney et al. 2017). The review proposed a number of field management actions, but their potential outcomes were unstated, and overall were received with a mix of positive and negative responses from stakeholders and local managers. Because of the sensitive social environment and mixed reactions to management solutions provided by Maloney et al. (2017), the reformed recovery group decided to re-start recovery planning using SDM.

Decision making context and objectives

A wide stakeholder group of government, scientists, community, iwi and charitable trust representatives (*n* = 42 people; hereafter, the stakeholder group) was convened to work through the SDM process and to co-develop the basis for a new recovery plan. Based on this, the recovery group (a subset of the stakeholder group and made up of 5 DOC staff, 3 iwi and 1 independent species specialist) would then make a final recommendation for approval by senior decision makers within DOC.

Four fundamental objectives and a way to measure their consequences (performance measure) were identified by the stakeholder group: (1) increase viability of the wild tara iti population (probability of persistence and adult population size after 50 years); (2) increase integration of mātauranga Māori (acceptability); (3) increase wider ecosystem benefits of tara iti management (change in size of breeding population of key species); (4) minimize cost (annualised cost in millions NZ$).

Table 4.1. Summarised set of eight management alternatives for tara iti. Letter Y denotes action is part of alternative. Full descriptions of each alternative are provided in Appendix B, Table S1. All alternatives included the following suite of nest protection actions: deterrents at poor nesting locations, raised nest platforms, storm chick & egg care.

Alternative	**Predator management**			**Habitat management**				**Ex situ management and release**	
	breeding grounds, non-native species	breeding grounds, native species	winter grounds & winter control	shell patches	vegetation control	foraging habitat man.	new site creation	remove infertile males	multi-clutch, harvest† & release
Field 1	Y	Y		Y	Y				
Field 2	Y	Y	Y	Y	Y	Y			
Field 1 + Captive 1	Y	Y		Y	Y				x 1
Field 1 + Captive 2	Y	Y		Y	Y			Y	x 1
Field 2 + Captive 3	Y	Y	Y	Y	Y		Y	Y	x 2
Field 2 + Captive 4	Y	Y	Y	Y	Y		Y		x 2
Field 2 + OZFT‡ supp.	Y	Y	Y	Y	Y	Y			add OZFT eggs
Field 2 + new sites	Y	Y	Y	Y	Y	Y	Y		

† Values indicate number of harvests. ‡ OZFT denotes Australian fairy tern.

Alternatives

The stakeholder group specified a number of management actions that were assembled into eight alternative management strategies, including three based exclusively on field management, and five combining field and captive management (Table 4.1). Several proposed actions overlapped with those identified in the review by Maloney et al (2017).

Predicting consequences

We modelled the structure of the tara iti population (Fig. 4.1A) to predict consequences under management alternatives. All analyses were carried out in R (R Core Team 2020). We built a stage-based, female-only, post-breeding population model (adapted from Kéry & Schaub 2021) to predict population size and persistence probability under status quo management over a 50-year period, aligning with DOC's species management timeframe (Joseph et al. 2009). We parameterised a status quo model using DOC field data, collected between 1997 and 2017. Productivity, the probability of an egg hatching and a chick fledging, was obtained using Generalised Linear Mixed Models (with package 'lmer', Bates et al. (2015)), and survival rates were obtained using mark-recapture analysis (with package 'RMark', Laake (2014)), respectively (further methods, see Appendix B).

Since the actions in all proposed management alternatives were untested, we derived vital rate parameters from expert elicitation using a modified Delphi method (Hemming et al. 2018a). We asked experts (*n* = 10) for four values: their best guess, their lowest and highest guesses, and an estimate of how confident they were that the true value would fall within this range (Appendix B). We elicited a range of vital rates for each alternative including survival, productivity and carrying capacity, defined as the total number of territories that could fit into sites and thus the maximum number of females that could breed in the population (Appendix B, Table S4). In addition, the biological experts believed that environmental conditions had worsened in recent times and that survival estimates derived from monitoring data were overly optimistic. The status quo population model therefore used elicited values for juvenile, immature and adult survival. All population models incorporated parametric uncertainty and temporal and demographic stochasticity (*sensu* McGowan et al. 2011; Kéry & Schaub 2021). Parametric uncertainty bounds were captured as the mean lowest and highest expert estimates (standardised to 100% confidence) using a beta-PERT distribution (Vose 1996); we propagated parametric uncertainty by randomly drawing values from this distribution in each simulated run of the model (*n* = 10,000 runs of the population model for each alternative; McGowan et al. (2011). Demographic stochasticity was simulated using Binomial and Poisson distributions around mean parameter values (Kéry & Schaub 2021). Probability of persistence (quasi-extinction defined here as adult females N > 3) and number of adult females after 50 years (with uncertainty; 2.5th – 97.5th percentile range) based on the expert opinions were summarised for each alternative in the consequence table (further methods, Appendix B).

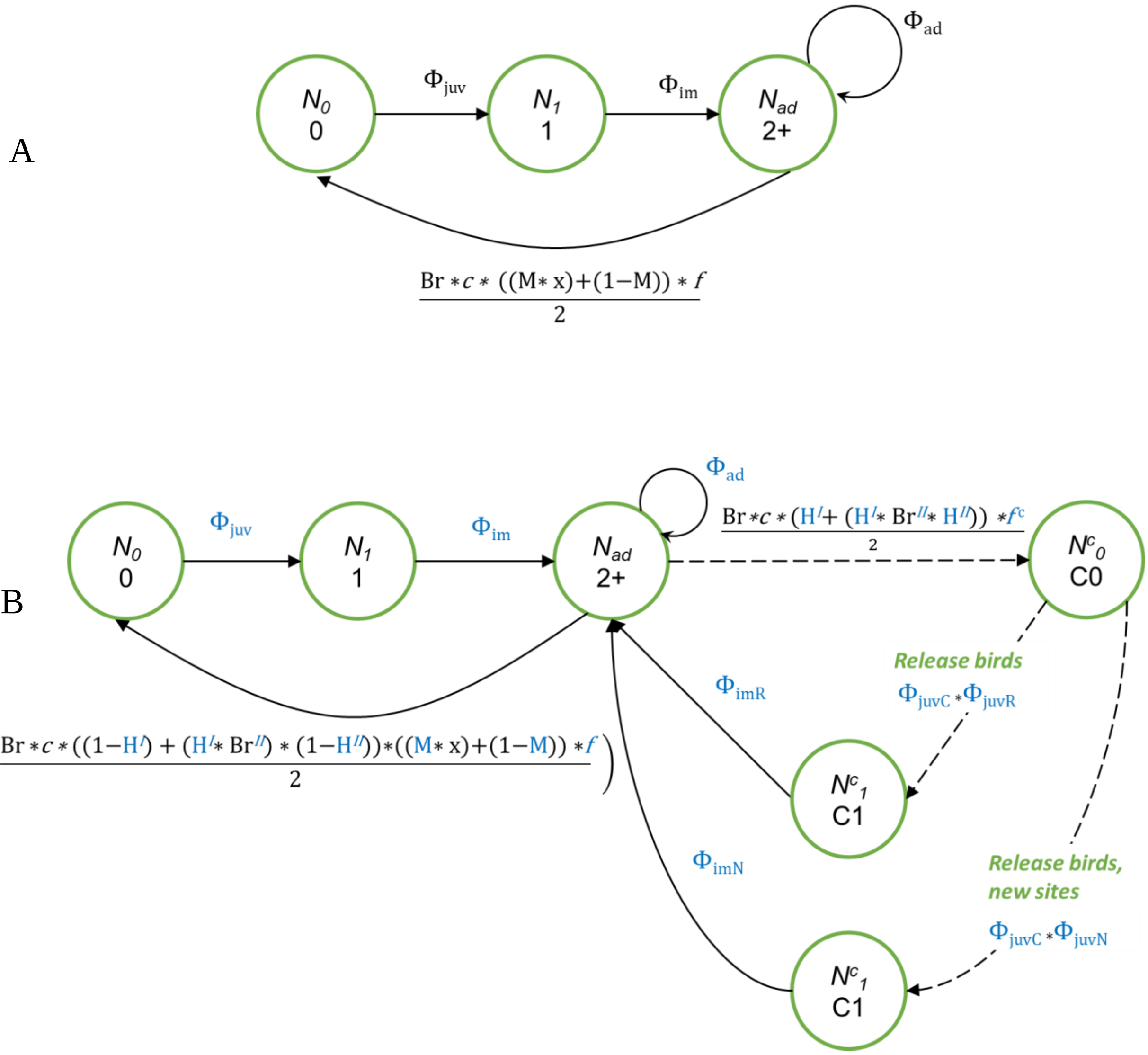

Figure 4.1. Life cycle diagrams for tara iti population under A, status quo management and B, Field 2 + Captive 3 management. The models are female-based and consider a post-breeding census. Parameters are colour-coded according to the data types that contribute to their estimation; black parameters are estimated using empirical data, blue parameters are informed by expert elicitation. Population size at different life stages defined as: number of fledged juveniles age 0 N_0; number of immatures aged 1 N_1; number of adults aged 2+ N_{ad}, number of captive fledged juveniles aged 0 N^C_0; number of captive-born immatures aged 1 N^C_1. Estimated demographic parameters are: juvenile survival Φjuv, juvC, juvR, juvN (wild, captive, released in current range, released outside range); immature survival Φim, imR, imN (wild, released in current range, released outside range); adult survival Φad; proportion of females attempting to breed Br, $Br^{//}$ (first, second attempt); clutch size *c*; proportion of nests harvested $H^{/}$, $H^{//}$ (first, second harvest); proportion of nests managed M; proportional reduction in breeding success of managed nests x, fecundity f (product of hatching and fledging probabilities).

For the wider ecosystem objective, we again used expert judgement ($n = 5$ experts) to predict the change in number of breeding pairs at each site expected under tara iti management alternatives for two key species (as a proxy for wider ecosystem health): variable oystercatcher (*Haematopus unicolor*) and Northern New Zealand dotterel (*Charadrius obscurus aquilonius*). We used the same elicitation method as described above and summarised the mean of expert estimates in the consequences table (Table 4.2). To estimate consequences for Mātauranga Māori, we had facilitated discussions with iwi representatives in the stakeholder group ($n = 4$) (McMurdo Hamilton et al. 2020; Chapter 3). They explained which actions, and thus alternatives, were acceptable or not using mātauranga Māori. Finally, cost was estimated, without uncertainty, as the average annual cost (in millions of NZ$) spread over the total 50-year management timeframe based on known costs of component actions from other species management programmes. This included: staff costs (salary, admin, vehicles, leave, training); contractors; equipment; large one-off costs such as aviaries. Excluded were: flights for staff; uncertainty contingency; corporate overheads; indirect costs; third-party funding contributions.

Decision making

We compiled the predicted consequences under each objective for each alternative into a consequences table (Table 4.2), which we presented to the recovery group alongside graphical representations of population outcomes (Fig. 4.2; Appendix B, Fig. S4). We then explored the decision space with hypothetical but desirable constraints on important objectives by highlighting alternatives that failed to meet three conditions: probability of persistence $p \geq 0.95$, population size $N \geq 50$ in the 2.5–97.5th percentile uncertainty range, and alignment with mātauranga Māori. In addition, we highlighted the relationship between increasing cost of management and biological benefits for tara iti (whilst keeping clear indication of acceptability under mātauranga Māori; Fig. 4.3). Given that there was no known budget against which to optimise, we instead held two rounds of anonymous voting. First, recovery group members voted for all the alternatives they considered to be acceptable. Second, each recovery group member voted for their single recommended alternative. In both rounds of voting, recovery group members based their votes on personal weighting across objectives, with the added step of judging how likely it would be to obtain funding. Votes were tallied in each round to identify a preferred choice for recommendation.

4.3 Results

Consequences

We predicted the consequences of each of the eight management alternatives with regard to the four management objectives, and captured uncertainty for the tara iti and the wider ecosystem objectives (Table 4.2). Under status quo management, the 50-year probability of tara iti persistence was estimated as $p = 0.60$ and mean adult female abundance as $N = 10$ (0–36; Table 4.2). The best-performing alternative for the tara iti objective also had the most intensive set of actions, ‘Field 2 + Captive 3’: $p = 0.99$ and $N = 42$ (12–77). Alternative ‘Field 2 + OZFT’ had the same probability of persistence, but a much lower predicted population size ($N = 24$, 3–49). All alternatives showed improved tara iti outcomes compared to status quo management, although with considerable overlapping uncertainty around predicted population sizes (Fig. 4.3, Table 4.2). Alternatives with ‘new sites’ (Field 2 + Captive 3/Captive 4, Field 2 + new sites) performed best for the wider ecosystem objective (Table 4.2). Cost ranged between 0.35 and 0.94 million NZ$ annually; the most expensive alternatives were Field 2 + Captive 3/Captive 4, and all were relatively expensive compared to the status quo annual budget. Five alternatives were assessed as acceptable based on mātauranga Māori (Table 4.2). Those that required the removal of infertile males from the wild population or the introduction of Australian fairy tern eggs were deemed unacceptable. No single alternative performed consistently well across all objectives. For example, there was a tendency for better performance on biological objectives (tara iti and wider ecosystem) to be more expensive and risk not being acceptable based on mātauranga Māori.

Table 4.2. Consequences table showing predicted outcomes of all alternatives with regard to the management objectives identified by the group: wild tara iti population viability, cost, wider ecosystem, Mātauranga Māori.

Alternative	**Objectives and performance measures**					
	Population persistence in 50 years (p)‡	**Population size in 50 years (mean N**, 2.5th – 97.5th percentile range)**	**Cost (annualised 50-year cost, millions NZ$)**	**Change in number of breeding pairs of northern NZ dotterel (percent, lwr – upr estimate) ††**	**Change in number of breeding pairs of variable oystercatcher (percent, lwr – upr estimate)**	**Alignment with mātauranga Māori (scale of acceptability)**
Status quo*	0.60	10 (0 – 36)	0.09	n/a	n/a	n/a
Field 1	0.70	12 (0 – 34)	0.35	15 (10 – 22)	6 (-1 – 9)	Acceptable
Field 2	0.95	23 (1 – 47)	0.41	27 (10 – 39)	7 (-4 – 20)	Acceptable
Field 1 + Captive 1 + learning†	0.74	12 (0 – 34)	0.50	15 (10 – 22)	6 (-1 – 9)	Acceptable
Field 1 + Captive 2 + learning	0.88	20 (0 – 45)	0.60	15 (10 – 22)	6 (-1 – 9)	Not acceptable
Field 2 + Captive 3 + learning	0.99	42 (12 – 77)	0.94	36 (12 – 46)	16 (-1 – 24)	Not acceptable
Field 2 + Captive 4 + learning	0.95	30 (2 – 60)	0.94	36 (12 – 46)	16 (-1 – 24)	Acceptable
Field 2 + OZFT	0.99	24 (3 – 49)	0.41	27 (10 – 39)	7 (-4 – 20)	Not acceptable
Field 2 + new sites	0.94	29 (1 –59)	0.64	36 (12 – 46)	16 (-1 – 24)	Acceptable

* status quo with Field 1 elicited values for survival. † learning added in second iteration of alternatives development represents 50% reduction in captive rearing success for three years. ‡ p = probability of persistence, quasi-extinction defined as adult female $N>3$. ** N = number of adult females †† minimum, maximum of mean elicited value scaled to 100% confidence.

The predicted biological consequences highlighted some key points. Alternatives based on 'Field 2' management at their base outperformed the rest, reflecting expert belief that survival and productivity would be increased due to increased predator control and reduced egg management, respectively. Alternatives with 'new sites' (Field 2 + Captive 3/Captive 4, Field 2 + new sites) also outperformed the rest, reflecting expert belief that the current breeding range has a modest carrying capacity (mean K, K = 17 pairs) before new sites were added (K = 24; Appendix B, Table S4). This suggests that increasing territories through development of new sites might be pivotal to increasing population sizes. The next two top-performing alternatives for population size, ‘Field 2 + Captive 4’ (*N* = 30, 2 – 60) and ‘Field 2 + new sites’ (*N* = 29, 1 – 59), had similar projections, especially when considering uncertainty (Fig. 4.3, Table 4.2). This was despite the fact that ‘Field 2 + new sites’ did not contain any captive rearing and releases, indicating that captive rearing may only provide marginal benefit. In summary, the model predictions using expert data indicated that captive rearing would only be more effective than other alternatives in combination with two harvests, higher productivity of wild pairs from increased egg fertility after removal of infertile males, and increased number of territories available from new site creation.

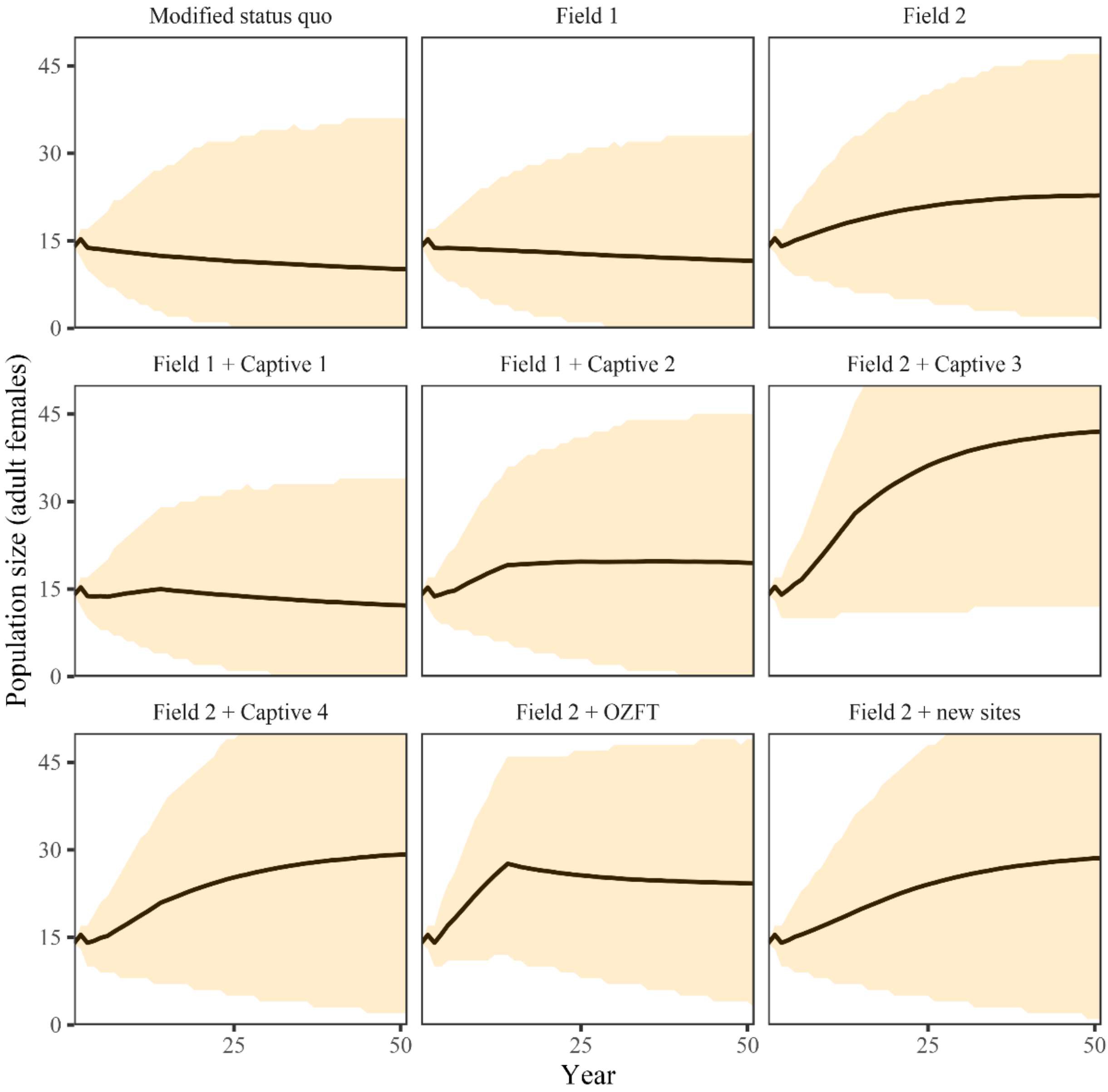

Figure 4.2. Tara iti population projections over 50 years for all alternatives. Black line denotes the mean number of adult females at each time step calculated from 10,000 simulations. Shaded area indicates 2.5th–97.5th percentiles of the simulated population sizes at each timestep. Projections are based on expert elicited vital rate parameters for all alternatives, except status quo, which had a combination of expert guesses and data derived parameter estimates.

Trade-offs and solution

The predictions indicated that there was no obvious best-performing alternative, making it challenging for the recovery team to make a recommendation. To explore the trade-offs in more detail, we compared both tara iti persistence and tara iti population size against cost and acceptability based on mātauranga Māori (Fig. 4.3). For these visualisations of the decision space, we did not include wider ecosystem impacts, since they were considered of secondary importance by the recovery group due to all alternatives having predicted positive outcomes. An important trade-off was that alternative 'Field 2 + Captive 3' had the highest tara iti population viability but also the greatest cost; and, crucially, it did not align with mātauranga Māori (Fig. 4.3). If this alternative was excluded, the remaining captive-rearing option, Field 2 + Captive 4, gave approximately the same biological results as 'Field 2 + new sites,' but was considerably more expensive (Table 4.2, Fig. 4.3). It was useful to look at explicit predictions of performance, given that this 'doing everything' approach (Field 2 + Captive 3) may have provided the highest predicted viability but not substantially, such that it would be chosen over a multiple-objective-compliant alternative.

The two rounds of voting showed some consensus across the recovery group. For example, in both rounds of voting there was near unanimous agreement that alternatives with 'Field 2' as a base were preferred (only 4 of 28 allocated votes went to 'Field 1' based alternatives). There was still diversity, however, in how individual recovery group members saw the trade-offs. In the first vote, 'Field 2 + Captive 3', 'Field 2 + Captive 4' and 'Field 2 + new sites' were joint top approved (each getting 7 of 9 peoples' votes; no other alternatives received support from more than three people). In the second vote for a single preferred strategy, 'Field 2 + new sites' got the most votes (5 of 9 people), whereas alternative 'Field 2 + Captive 3' received all the remaining votes (4 of 9 people). Based on this, the group recommended doing 'Field 2 + new sites' but would seek continued discussions with iwi and more funding in future to eventually include captive breeding, as is detailed in the alternative 'Field 2 + Captive 3'.

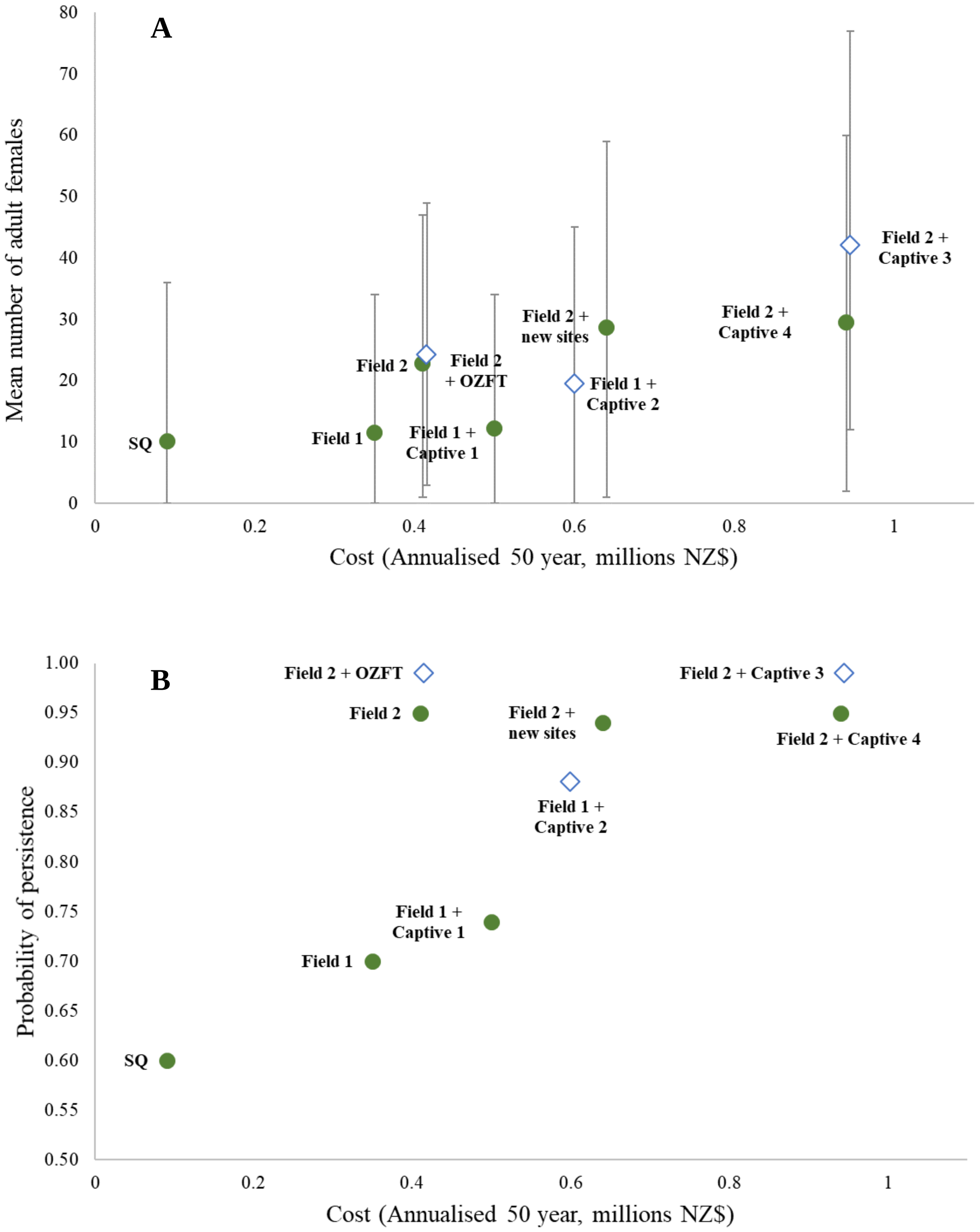

Figure 4.3. Predicted cost with A, mean estimated population size after 50 years, and B, probability of persistence after 50 years, plotted for all alternatives. In A, the y-axis indicates the mean number of adult females in year 50, over 10,000 simulation runs for a given strategy. Bars indicate 2.5th–97.5th percentile uncertainty range. Filled circles (acceptable) and open diamonds (not acceptable) indicate alignment of alternative with mātauranga Māori. Note that the data points for Field 2 + Captive 3 and Field 2 + Captive 4 have been staggered deliberately along the x-axis to view them more clearly, but they have the same predicted cost (see Table 4.2). In B, probability of persistence indicates the proportion of 10,000 simulation runs that results in tara iti extinction (less than 3 adult females) for a given strategy. SQ denotes status quo productivity and Field 1 survival.

4.4 Discussion

While it can be tempting to bypass the complexity of difficult decisions for small populations, doing so can hinder conservation efforts when they are most needed (Rout et al. 2013; Evans et al. 2017). Recognising diverse stakeholder values and using decision-relevant predictions that incorporate uncertainty not only improves rational planning, it can also strengthen group unity. Here, we found that after many years without an agreement on whether and how to change tara iti management, the SDM process succeeded in helping the group break a deadlock. All the stakeholders were able to comprehend the uncertainty in the biological outcomes of management alternatives, and how they related to other key objectives that were previously unstated. This moved them away from perpetuating the status quo, yet also avoided a dogmatic application of a ‘do everything’ approach. Given the complexity of this problem and the longstanding deadlock on recovery planning, agreeing on a management alternative was an important achievement for the tara iti recovery group.

Rebuilding group unity was integral to moving away from status quo perpetuation. In particular, building bespoke biological models together with the tara iti experts helped to build trust, both in the models and interpersonal relationships (Chee et al. 2017). First, the group were able to participate and jointly learn about uncertain knowledge through summaries of monitoring data (e.g. Chapter 5) and consequences from expert elicitation. Second, we built models that were complex enough that the group felt they would be useful and had integrity. Our experience echoes others who have found that conflict can be eased and trust can be built through effective participation and information sharing (especially between different stakeholders) (Redpath et al. 2013; Meek et al. 2015) and when system models are complex enough to be believable yet not unwieldy (Converse & Armstrong 2016; McGowan et al. 2020). Further to this, developing believable models for management alternatives also helps managers consider the risks and benefits of inaction. The po’o-uli (*Melamprosops phaeosoma*), baiji (*Lipotes vexillifer*) and Christmas island pipistrelle (*Pipistrellus murrayi*) are all thought to have gone extinct in part due to delayed action (Groombridge et al. 2004; Turvey et al. 2007; Martin et al. 2012b). Inaction can stem from different causes, including reluctance to deviate from the status quo (Ferrière et al. 2020) and risk aversion (Canessa et al. 2019), and can be overcome when functioning leadership or recovery groups confront the status quo bias by using trusted models to compare it explicitly to alternatives.

Being confronted with an expression of uncertainty around predictions allowed the group to agree on a non-captive breeding alternative and prevented a dogmatic ‘do everything’ response. Systems containing small populations are typically poorly known, and yet species experts often have strong, differing beliefs about what management must be done and how a species will respond. Individual expert beliefs derived in an unspecified way are prone to being biased and inaccurate (Speirs-Bridge et al. 2010; McCarthy 2014; Hemming et al. 2018b). Without acknowledging and evaluating these beliefs, they can become entrenched and cause division within groups, and bring planning to a standstill (Meek et al. 2015; Canessa et al. 2019). In many cases, the divergent beliefs from experts can be resolved through exploration of linguistic and epistemic uncertainty (McCarthy 2014; Milner-Gulland & Shea 2017), and through formal approaches to expert elicitation (Hemming et al. 2018a). For example, some tara iti biological experts strongly believed the way to recover the species was through captive breeding and releases. These are common responses for endangered species with extreme small populations, but conservation has been criticised for over-reliance on such intensive activities without looking at a wide range alternatives or making explicit predictions (Sutherland et al. 2004; Taylor et al. 2017). For example, in our case, little is known on captive breeding for terns (Chapter 3). A thorough evaluation of these beliefs showed that the group actually expected little added benefit of captive breeding in absolute terms, perhaps because of the low number of nests available for harvest, or perceived low carrying capacity that constrains tara iti recovery.

The explicit assessment of alternatives against management objectives then helped the group make more transparent trade-offs, recognising multiple objectives instead of focusing only on biological objectives. Many of the actions proposed for tara iti prior to the SDM were intensive (e.g., reinforcement, removing infertile individuals, supplementing nests with Australian fairy tern eggs; Maloney et al. 2017). Experts believed those actions could benefit population recovery, but not enough to gain universal recommendation when trade-offs with all management objectives were considered. By recognising multiple objectives, a different alternative was recommended (a change from ‘Field 2 + Captive 3’ to ‘Field 2 + new sites’). Furthermore, the deliberative process clarified why some alternatives were not favoured, allowing more focused dialogue or shared learning in future decisions. For example, due to explicit evaluation of alternatives against mātauranga Māori, the steps required to include management of infertile males in the future are now clear to both iwi partners and population biologists alike (McMurdo Hamilton et al. 2020). Encouraging groups to approach decisions in this inclusive way relieves conflict by ensuring the values that matter are used to build and select the best management (Gregory et al. 2012b; Bennett et al. 2017; McMurdo Hamilton et al. 2020; Zafra-Calvo et al. 2020). Participants feel validated when their values and perspectives are recognised (McGowan et al. 2014), which promotes group unity further.

Although SDM facilitated the overall process of making trade-offs, we still faced the challenge of not knowing the available budget. Cost is a common objective for recovery planning, yet one that is often not explicitly considered, potentially creating problems for project success (Iacona et al. 2018). In our case, the recovery team was required to make a recommendation that had to account for a *feeling* of how likely funding would be available, as well as balancing how much they would each invest for a unit of recovery of tara iti (e.g. breeding pair) and how much they favoured alignment with mātauranga Māori. The second preferred alternative in our study was one that clearly favoured biological gains despite expense and poor alignment with mātauranga Māori, indicating that some recovery experts still valued tara iti recovery at any cost. Interestingly, in light of potential biological gains, iwi representatives tentatively considered discussing the acceptability of breaking infertile pairs and this remains to be explored in the future. Indeed, despite not having a known budget, we note the benefit of a fully specified consequence table in providing an informed decision space to the ultimate decision makers (senior managers in DOC in our case). Decision makers can readily view recovery team recommendations against alternatives and balance these within their broader budget allocations.

Tackling complexity is critical for decision making for small populations that typically involves diverse stakeholders working in complex and uncertain systems. Our tara iti case study provides a positive example of a formal decision analysis that was highly effective in this situation. SDM is cognitively demanding, but by clarifying objectives (such as indigenous values) and evaluating consequences under uncertainty, it improved trust, supported learning and dialogue between group members and progressed the group to a shared management recommendation. The group moved towards action (avoiding status quo bias) but in a rational way that focused on objectives first (avoiding an alternatives-focused approach from a narrow biological focus). The best decisions are the ones that are made and supported (Bennett et al. 2019): addressing complexity in a transparent and structured way using decision analysis can deliver clear insight and an agreed, rational, path forward, and did so in this case.

Chapter 5: Rational interpretation of messy monitoring data to improve future conservation decisions

Abstract

Successful recovery of species and ecosystems often needs decades of close-order management. Monitoring allows managers to evaluate and adjust management to changing conditions over time. However, monitoring data are often harder to interpret than anticipated because of uncertainty, lack of experimental controls and hidden bias. In this study, we show a way to combine decision-support tools and counterfactuals to evaluate and improve management, countering bias in ad-hoc monitoring data. Tara iti (New Zealand fairy tern, *Sternula nereis davisae*) is Aotearoa–New Zealand's rarest bird and its eggs have been managed intensively to prevent nest failure since 1997. There is a need to measure management efficacy, yet there is no experimental control and data for managed nests are biased toward nests that were already threatened. We used a decision tree and multiple counterfactuals with uncertainty to explore effectiveness to date and future improvements to egg management. A naïve interpretation of the results, without accounting for bias, might suggest that active nest management performs poorly and even worse than no management. However, our counterfactual analysis showed that if managers' accuracy in identifying imminent threats to eggs is high, egg management is effective in meeting its objective. Our results also suggest there are limited opportunities to improve egg management within the current context. Our results clearly illustrate the pitfalls of naïvely interpreting messy monitoring data without addressing bias. Decision support tools, and the appropriate use of multiple counterfactuals with uncertainty, can help overcome those pitfalls and leverage critical information on management efficacy from imperfect monitoring data.

5.1 Introduction

Recovery of endangered species may require decades of close-order management (Young et al. 2014). During that time, managers will need to assess and adjust actions in response to improved knowledge, or adapt to changing conditions in the species, environment, and planning context. To know what is working and what must be changed, large quantities of monitoring data are usually accumulated, often at great expense and effort (Buxton et al. 2020). However, monitoring data may be harder to interpret than managers anticipate: many programmes either make incorrect inferences about management efficacy (Christie et al. 2019) or fail to use those data at all (Lindenmayer et al. 2013; Buxton et al. 2020).

Conservation monitoring data and their interpretation can be ‘messy’ for several reasons. Management often evolves ad hoc, because initial expectations are uncertain, resource limitations vary, and speedy intervention might initially be prioritised over thorough decision-making. For endangered species conservation, where every individual matters, robust experimental approaches including controls – such as Before-After Control-Impact (Christie et al. 2019) – may be considered too risky (Runge 2011). Managers might be reluctant to deviate from current actions (this is known as status quo bias; Samuelson & Zeckhauser (1988)), or might feel that changing actions would waste time and resources invested in previous management (sunk cost fallacy; Arkes & Blumer (1985)). These problems are compounded by the fact that conservation decisions rely heavily on intuitive judgment by experts (Martínez-Abraín & Oro 2013; Walsh et al. 2019), often with cursory and informal examination of outcomes.

To counter such challenges, conservation can adopt formal decision-support methods. For example, counterfactuals — if-then statements where one examines critically what might have happened under alternative choices — are increasingly used to assess conservation outcomes where controls are not available (Ferraro 2009; Bull et al. 2020; Jellesmark et al. 2021). Counterfactuals can help reduce bias, but should be built carefully to avoid ‘just-so’ storytelling, where one implicitly justifies their original choice and thus biases hindsight (Roese & Olson 1996). In most cases, counterfactuals remain uncertain due to heterogeneity in the ecological system and because their interpretation is ultimately subjective (Baylis et al. 2016; Bull et al. 2020; Coetzee & Gaston 2021). However, used correctly, counterfactuals can improve decision making and contribute towards conservation impact evaluation to inform management elsewhere (Baylis et al. 2016; Coetzee & Gaston 2021).

Recovery plans for endangered birds often involve intensive nest management to improve reproductive success by mitigating threats like predation, disease, human disturbance, or poor weather. Actions include nest enhancement (Jansen 2005; Booker & Moxom 2019), egg or brood manipulation (Jones et al. 1995; Lobo & Marini 2013) and head-starting (Robertson et al. 2010; Collins et al. 2016). Nest management decisions are typically made under time pressure and great uncertainty. Consequently, they incur many of the evaluation challenges we described above, which has led to recent calls for better decision making (Edwards et al. 2019).

Aotearoa-New Zealand's most endangered indigenous breeding bird, tara iti (New Zealand fairy tern, *Sternula nereis davisae*) is an extreme example of such challenges. Since 1997, all tara iti eggs have been monitored closely and, whenever threats are perceived, eggs are managed intensively in an attempt to prevent nest failure. Managed and unmanaged eggs are therefore not comparable, there is no control and the monitoring data are biased. Whilst fledging rates improved during 1991-2002, it is unknown whether this was attributable to egg management (Ferreira et al. 2005). In this study, to evaluate tara iti management efficacy from long-term monitoring data and to identify possible improvements, we used a decision tree and a novel post-hoc counterfactual analysis.

5.2 Methods

Study system and decision context

Tara iti is the smallest and rarest breeding tern in Aotearoa-New Zealand, listed as 'Nationally Critical' (Robertson et al. 2017). From October to February, the population of approximately 30 birds (DOC, unpubl. data) breeds at five locations in the northern North Island: Waipu (≤3 pairs); Mangawhai Spit (≤6 pairs); Te Arai (one pair); Pakiri, (one pair) and Papakanui Spit (≤4 pairs) (Hansen 2006; DOC, unpubl. data). Each of these sites is close to estuarine and oceanic waters, and is on open, sandy areas often with little vegetation and patches of shell distributed throughout. Tara iti lay 1-2 eggs in a 'scrape' on the sand, primarily on these low-lying shell patches (Treadgold 2000). In the past ten years, on average, only 9-10 females have been breeding. Tara iti is vulnerable to a range of threats during its breeding season, such as predation of eggs, chicks or adults by invasive mammalian predators; predation of eggs or chicks by native avian predators; disturbance or nest destruction by humans; and nest failure due to chilling, sanding, or tidal flooding of eggs, often in combination with poor weather.

Their productivity is also inherently affected by high rates of (presumed) infertility and embryo death, which typically reflects a genetic bottleneck (Baling 2008; Assersohn et al. 2021).

Tara iti management and monitoring data collection are coordinated by Aotearoa-New Zealand's Department of Conservation (DOC) with support from DOC rangers and scientific staff, community volunteers and Auckland Zoo. Breeding habitat is managed by DOC at the four biggest sites and aims to maximise safe nesting locations and limit human disturbance with fences, ranger presence and creating additional shell patches. Furthermore, if a nest is at risk from flooding events, sandbag walls and trenches are occasionally used *in situ* to stem the tide (Ferreira et al. 2005; Hansen 2006). Habitat management also aims to protect the population by lethally controlling native and non-native predators with traps and poison. The objective of direct egg management, therefore, is to maximise reproductive success when a threat to a nest - such as a nearby predator or flooding tide - is detected, in spite of habitat management.

Decision analysis

In our study, we investigated efficacy of threat management by summarising the survival of managed and unmanaged eggs since 1997, with a focus on those lost to threats that the programme controls for (storms, tides and predators). We used a decision tree to define the decision making context for egg management, and structured and analysed the raw breeding monitoring dataset to evaluate efficacy. Eggs are not managed randomly or in a balanced way: the recovery programme aims to manage every egg that is perceived to be threatened, and therefore the decision not to manage is made when no threat has been perceived. Therefore, after the initial naïve analysis, we used counterfactuals to assess the possible true impact of management and explored ways of improving outcomes in the current context.

Efficacy of nest and threat management

To better understand the decision faced when considering egg management and the sequence of events that follows, we constructed a decision tree (Fig. 5.1): a simple way to frame a decision formally by showing objectives, alternative actions and their probabilistic expected outcomes with risk (Behn 1982). The tree starts with a decision node (square) and has branches representing the alternative management actions available to managers: (1) not managing, (2) shifting nests away from tidal inundation, (3) transferring imperilled eggs directly to foster pairs, (4) bringing eggs into temporary captivity (at Auckland Zoo) until the threat has passed and returning them to either natal parents or foster pairs (details in Table 5.1). This approach and its suite of management actions has remained relatively unchanged since 1997.

Table 5.1. Description of alternative egg management actions available to managers when a threat has been detected that is insurmountable by other forms of management.

Management action	**Description**
None	No management of eggs
Shift	Nest is gradually shifted away from incoming tide or sand over a period of a few hours
Donate	Eggs removed and either directly transferred to foster parents or via Auckland Zoo first (for ≤1 day)
Artificial incubation – natal return	Eggs removed and incubated - sometimes *in situ* but usually at Auckland Zoo (for >1 day); dummy eggs put in natal nest. Eggs returned to parents later
Artificial incubation – donate	Eggs removed and incubated at Auckland Zoo (for >1 day); natal parents abandon. Eggs donated to foster parents later

On each branch of the tree, uncertainty nodes (ovals) represent the possible egg or chick fates after a given decision: survive (egg hatch or chick fledge), die due to threats preventable by management (predator, tide or storm), die due to other or unknown causes. The artificial incubation branch includes an additional uncertainty node, since some eggs are dead on arrival to captivity or die in captivity, and a subsequent decision node to choose between returning surviving eggs to natal parents or donating them to foster parents. For all branches, the outcome of the tree is whether or not an egg survives to fledging stage, which is the ultimate management objective.

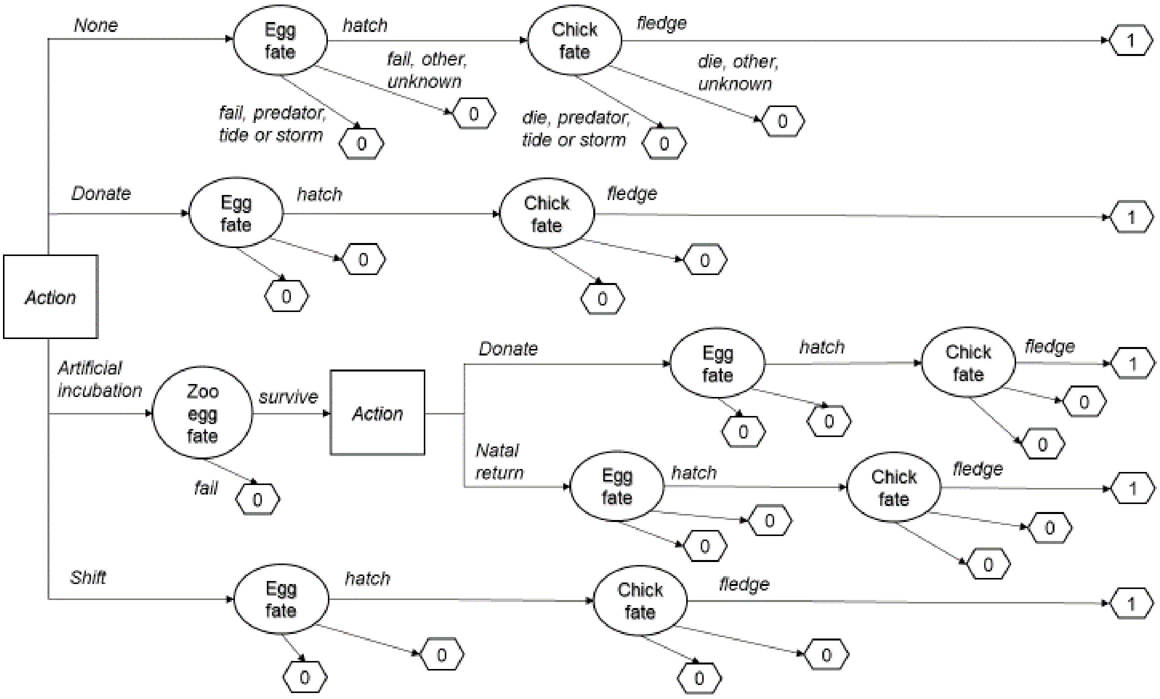

Figure 5.1. Decision tree showing the options for tara iti egg management after a threat has been detected. Each branch represents an egg management alternative and the temporal sequence of events from left to right. Squares represent decision points, ovals represent uncertainty nodes, and hexagons represent outcome in terms of in terms of hatching or fledging (1 = success, 0 = failure). Outcomes described on first branch apply to all branches but are not repeated across other branches to ease visualisation.

To estimate the overall fledging probability along each branch, we needed to estimate probabilities of different outcomes at each uncertainty node. We analysed DOC's historical breeding data from 1997 – 2017 (n = 505 eggs, of which n = 305 were fertile at candling), and categorised nest management types (Table 5.1), outcomes, and causes of failure (Table 5.2). We included only clutches with eggs deemed to be fertile, because infertile eggs are deliberately not managed and ultimately fail. We estimated multinomial probabilities for all nodes using JAGS (Plummer 2003) through the 'jagsUI' package in R (Kellner, 2015). We obtained 10,000 posterior samples for each probability using an uninformative Dirichlet prior (code available at github.com/ThalassaMH/thesis-examination). We then entered the probability estimates in the corresponding tree nodes and calculated each branch's probability of success (fledging) by multiplying the estimated probabilities along that branch. For example, the probability of an egg surviving to fledging after being donated to another pair was calculated as $p(success|donate) = \big(p(hatch|donate)\big)\big(p(fledge|donate)\big)$. To propagate uncertainty, we repeated calculations for each set of posterior samples, resulting in a full distribution of estimated fledging probability for each branch, rather than a single point estimate. We plotted all distributions to illustrate the optimal management option for managers in this situation, noting that these results were biased and would need further interpretation using counterfactuals.

Table 5.2. Description of apparent causes of egg or chick failure, as categorised using the notes in the historical dataset.

Outcome recorded	Apparent cause	Description
Abandon	Embryo death, tide/storm, one parent, human disturbance, unknown	Parents have abandoned the clutch, cause inferred. Embryo death is discovered at time of post-mortem. One parent indicates that one of the pair left or died, leaving the final one alone that also subsequently abandoned.
No hatch (egg)	Embryo death, tide/storm, damage, unknown	Like above, but field notes do not make it clear whether the pair abandoned, did not incubate enough or field staff removed egg.
Removed (egg)	Infertile, embryo death, tide/storm, unknown	Eggs were removed either because the eggs were known infertile, parents had stopped incubating or eggs had gone over date they could feasibly hatch, releasing the pair to lay again.
Predated	Invasive mammalian, aerial, or unknown predator, human removal	Mammalian predators are cats and rats, aerial predators are the native Australasian harrier and native gull species, red-billed gull, and Southern black backed gull, or recorded as an unknown predator
Die hatch	Tide/storm, unknown	Chick attempted to hatch but died.
Chick death	Health, human, tide/storm, unknown	Chick hatched but then died due to any suspected health reason, human trampling, or disturbance, or because they were starved/chilled in storms, or for unknown reasons.

Counterfactuals to evaluate and improve outcomes

To understand how effective management was, we compared the (biased) observed outcomes with two hypothetical, extreme counterfactuals. Nests are managed or not depending on whether a threat is perceived (predation, storm flooding or tides); however, due to partial observability, threat perceptions may or may not be accurate (Cohen et al. 2016). We do not know true accuracy of management, in other words, we do not know the proportion of eggs whose fate was accurately predicted by managers. To understand how effective management was, we compared the (biased) observed outcomes with two hypothetical, extreme counterfactuals. In Counterfactual 1, we countered that managers were 100% accurate in classifying nests as under threat. Thus if those eggs had not been managed (*n* = 156), egg survival would have been zero. In Counterfactual 2, we countered that managers were 50% accurate in classifying nests as under threat, reflecting random selection of nests for management. Thus, in this scenario assume both managed and unmanaged nests have the same overall probability of fledging as observed unmanaged eggs (i.e. the same as the 'no nest management' branch, the background rate of success in good, managed habitat). The range between these two counterfactuals captures all reasonably realistic classification types, and the range of a manager's ability to judge the presence of a threat from no better than random to perfect accuracy. To compare both counterfactuals to the observed outcomes, we calculated their hypothetical number of fledglings. For the first counterfactual, we simply subtracted the fledged young of managed eggs from the overall totals; for the second, we multiplied the number of managed eggs by the background rate of success (estimated fledging probability for the 'no management' branch), repeating the calculation for each posterior sample. As a result, we obtained a point estimate for the first counterfactual, and a distribution of estimates for the second one. We then calculated the difference between each counterfactual and the actual observed outcome.

After counterfactually assessing past management, we considered possible future improvements in two areas. In our dataset, some eggs that were not managed still failed due to predation, storm, or tide (*n* = 12). First, therefore, we tested how much fledging success could increase if these threats had been detected and classified as present, and thus managed (i.e., no false negatives). We multiplied those eggs by the fledging probability of the management branches 'donate' and 'artificial incubation' (repeating for each posterior sample to obtain uncertainty), then added the result to the total number of fledglings in the observed data.

Second, we tested how much fledging success could increase by improving artificial incubation options to the equivalent of the best performing management action. Artificial incubation is an attractive option for managers because donation to foster pair is strongly constrained. There are only two foster pairs in the current population, and these might not be available at the right time (although other pairs are sometimes used as fosters, this causes an undesirable knock-on effect of eggs needing captive management). We multiplied the eggs that underwent artificial incubation management in our dataset (*n* = 99) by the fledging probability from the best-performing management action, then added the result to the total number of fledglings in the observed data.

5.3 Results

Efficacy of nest and threat management

Of 305 fertile eggs laid, 156 (50.6%) received egg management in response to a perceived threat, and out of these, 14 eggs (9%) were shifted, 43 (27.5%) were donated, and 99 (63.5%) were artificially incubated. Of those surviving artificial incubation, 44 were donated (71%) and 18 (29%) were returned to natal nests. Of the unmanaged eggs (*n* = 149, 49.4% of all fertile eggs), 12 (8.1%) failed at egg stage due to threats that the conservation programme controls for through management (predators, 2.7%; tides and storms, 5.4%). After chicks hatched, or after eggs were managed, no further targeted intervention was possible. Of chicks hatched from unmanaged eggs (*n* = 114), 17 (14.9%) subsequently died due to predators (7%) or tides and storms (7.9%). Of managed eggs post-intervention (*n* = 121, excluding those that died while in artificial incubation), 9 eggs (7%) subsequently failed due to predators (4%) or tides and storms (5%). Sixteen chicks hatched from managed eggs (13%) subsequently died due to predators (4%) or tides and storms (9%).

As expected, when we modelled the biased data, nests that were not managed (because they were never perceived to be under threat) had the highest probability of fledging (mean *p* = 0.55, 2.5th – 97.5th percentile range = 0.47 - 0.63: Fig. 5.2). Ignoring bias, no management would be selected as the best decision in 99.6% of decision tree solutions. When eggs were judged to require management, donating them to a foster pair had the highest fledging success (*p* = 0.32, 0.19 – 0.46), followed by artificial incubation and donation (*p* = 0.26, 0.17 – 0.36), artificial incubation and natal return (*p* = 0.17, 0.06 – 0.31), and shifting (*p* = 0.14, 0.02 – 0.35). Eggs rescued for artificial incubation had a probability of surviving *ex situ* before being returned to either natal or foster nests of *p* = 0.64 (0.55 – 0.74).

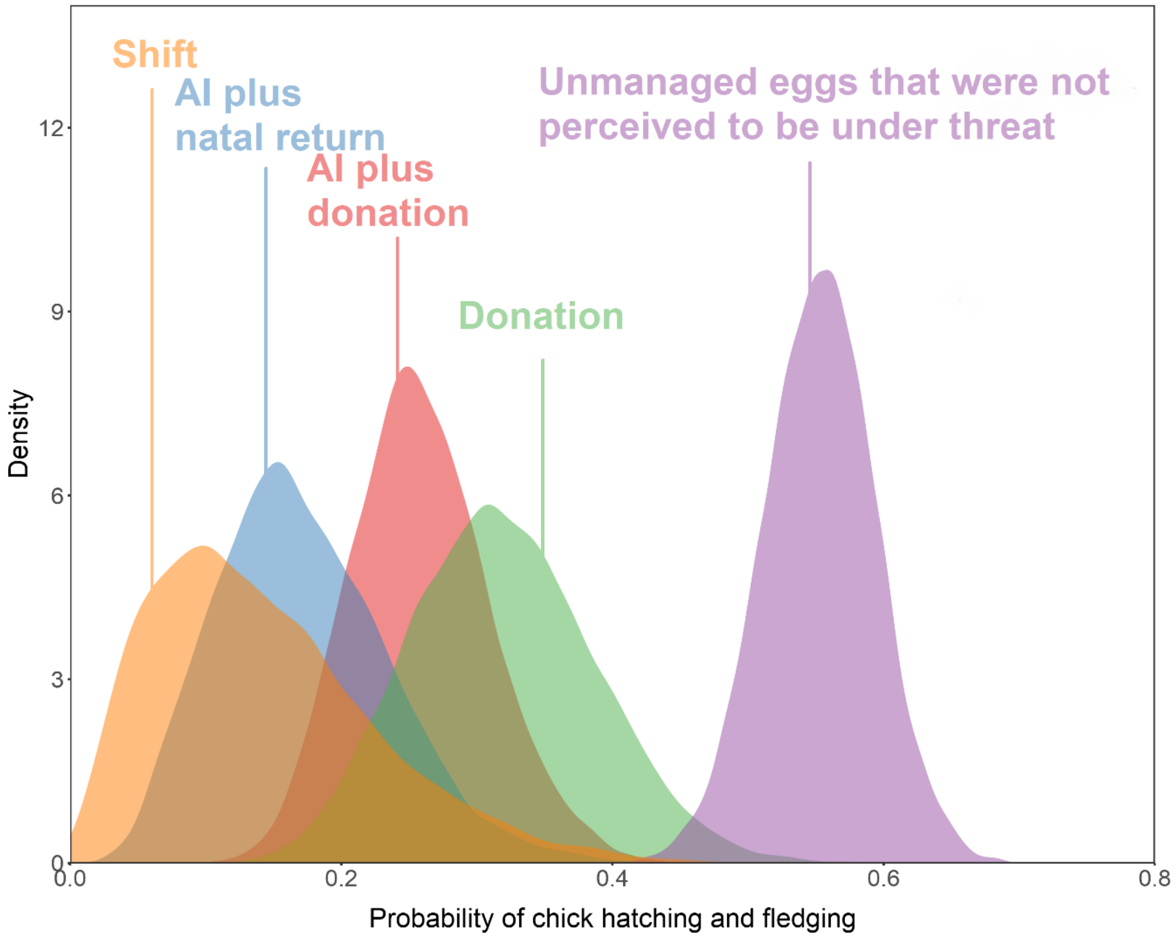

Figure 5.2. Frequency distributions of the probability of successful fledging under different egg management alternatives, from 10,000 simulations of the decision tree using estimated probabilities. This is a naïve analysis of the biased dataset: as all nests that are judged to be under threat are subsequently managed, unmanaged eggs (purple curve) do not represent a true control group. They only indicate the probability of fledging from nests that managers have judged to be safe from tides or predators.

Counterfactuals to evaluate and improve outcomes

Under the first counterfactual (Counterfactual 1: 100% accurate threat detection), where no management means no survival, over twenty years there would have been 39 fewer fledglings than observed (a 13% loss; Table 5.3, Fig. 5.3). Under the second counterfactual (Counterfactual 2: 50% accurate threat detection), where no management means the same survival as observed unmanaged eggs, there would have been on average 47.3 more fledglings than observed (a 16% gain; Table 5.3, Fig. 5.3). In other words, if managers were accurately detecting real threats,, managing eggs was the best decision and provided up to 13% more fledglings than not managing. By contrast, if managers were perceiving threats inaccurately (i.e., equal to random classification of nests needing management) and managing nests unnecessarily, they could have lost out on providing up to 16% extra fledglings than observed, meaning it would have been better not to manage. If both scenarios are equally likely, the gains of not managing eggs are only marginally higher than the losses. Thus, if the accuracy with which managers perceive imminent threats is substantially better than random, they should manage eggs perceived to be under threat.

Looking at future improvements, increasing correct classification of nests as threatened during the egg stage (i.e., no false negatives) would provide a 1% increase in fledging success, equivalent to 3 fledglings over the last 20 years (Table 5.3, Fig. 5.3). Improving the success rate of artificial incubation to the same level as the best performing alternative, donate, would provide a 3% (-2 – 7%, Table 5.3) increase in fledging numbers, corresponding to between 4 fewer and 22.5 more fledglings over the last 20 years (Fig. 5.3).

Table 5.3. Net benefits and losses in fledglings from two counterfactuals and pursuing different scenarios of improvement on detecting threats to eggs and on egg management tools, compared to observed outcomes from current management of eggs (*n* = 305). The scenarios assume the number of fledged young from unmanaged eggs (*n* = 83/149) remains constant. AI means artificial incubation. CRI are 2.5th - 97.5th intervals.

Scenario	*Number of chicks fledged from managed eggs (CRI)*	*Total number of chicks fledged (CRI)*	*Proportion of eggs that fledged (CRI)*	*Proportional change from status quo (CRI)*
Observed outcomes	39	122	0.4	*n/a*
Counterfactual 1: 100% accurate detection	0	83	0.27	-13%
Counterfactual 2: 50% accurate detection	86.3 (73.8 – 98.7)	169.3 (156.8 – 181.7)	0.56 (0.51 – 0.6)	+16% (11 – 20)
Improve accuracy through no false negatives	42 (40.7 – 43.5)	125 (123.7 – 126.5)	0.41 (0.41 – 0.41)	+1% (1 – 1)
Improve AI efficacy to equal best performing management alternative	47.4 (35 – 61.5)	130.4 (118 – 144.5)	0.43 (0.39 – 0.47)	+3% (-2 – 7)

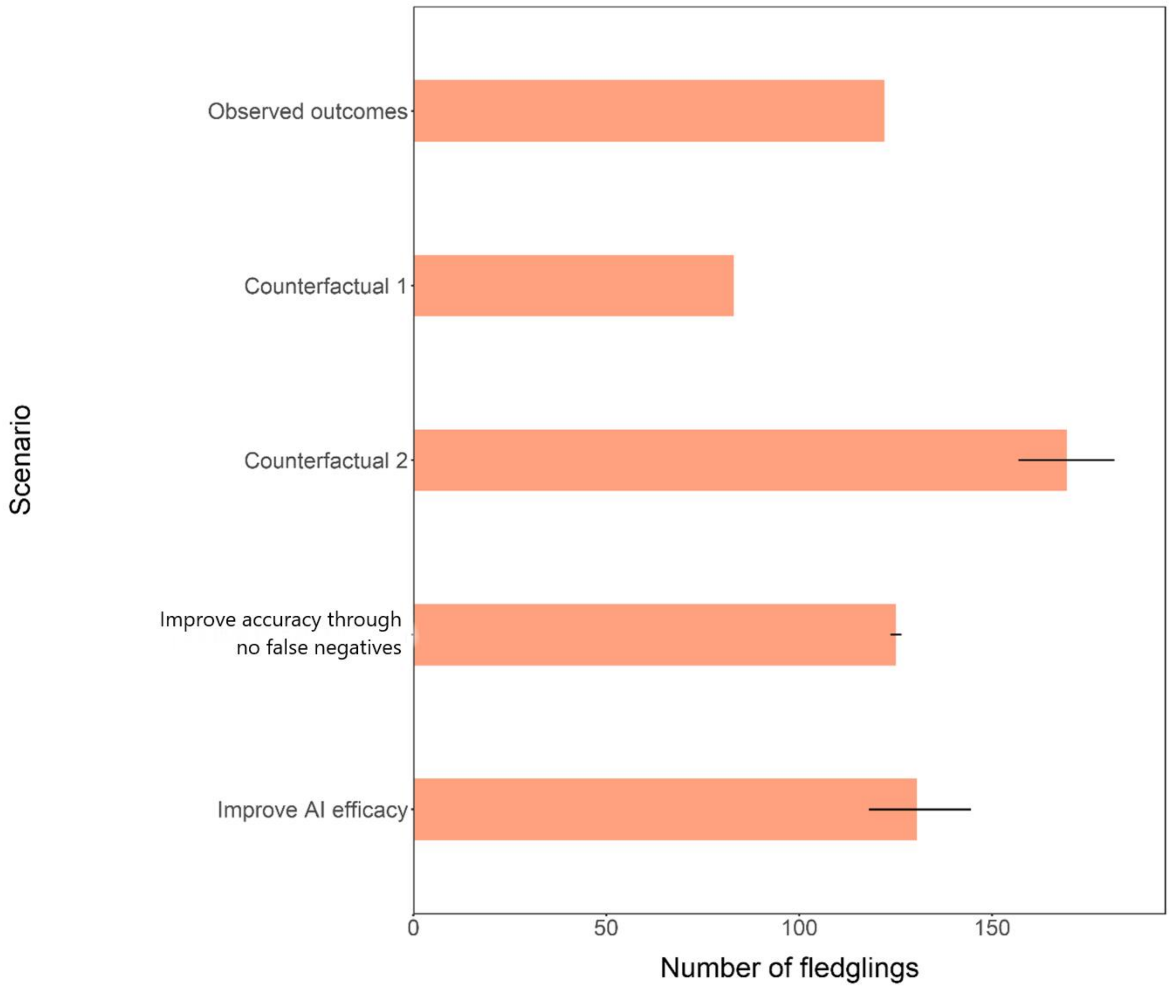

Figure 5.3. Fledgling production observed between 1997-2017 (Observed outcomes) and under other theoretical scenarios: counterfactual with 100% accurate detection (Counterfactual 1); counterfactual with 50% accurate detection (Counterfactual 2), improve false negatives to 0% (Improve accuracy through no false negatives), improve artificial incubation efficacy to equal best performing management alternative (Improve AI efficacy). Black error bars indicate 2.5th - 97.5th range.

5.4 Discussion

Tara iti nest management is representative of conservation interventions that evolve ad hoc in the absence of true experimental controls, but still require good evaluation of efficacy. Our results clearly illustrate the pitfalls of naïvely interpreting messy monitoring data. Overall, our counterfactual analysis and evaluation of hypothetical improvements indicated that if detection accuracy of imminent threats is substantially better than random, then egg management is effective in meeting its objective, though perhaps with limited opportunities for significant improvements within the current context. This interpretation is in clear contrast with intuitive interpretation of raw monitoring data, which show a much lower fledging success for managed eggs.

Without a counterfactual analysis, one might have interpreted observed outcomes to date as suggesting that active nest management performs poorly and worse than no management. On the other hand, if one simply assumed management to be always justified, as in the counterfactual where all detected threats were true threats, it would have obviously confirmed initial guesses (‘just-so’ storytelling). While counterfactuals are often portrayed as static and without uncertainty (Butchart et al. 2006; Hoffmann et al. 2010; Young et al. 2014), using multiple counterfactuals better reflects their subjective and uncertain nature (Bull et al. 2020; Coetzee & Gaston 2021). To avoid subjective or biased interpretations, we used two opposing counterfactuals at the reasonable extremes of accurate threat detection, thus creating a reference range against which to quantify the losses or gains from managing eggs. Presenting counterfactuals with uncertainty, as we did, may also better account for risk aversion (Tulloch et al. 2015; Canessa et al. 2019).

Our two counterfactuals provided a reference range of potential management efficacy (fledgling production) based on accurate threat detection, managers’ ability to accurately judge that a threat is truly present and will destroy eggs. However, we have little information on where managers’ current ability falls along this range, and therefore where to improve. On one hand, managers could try to increase their detection of threats in the field to reduce unmanaged nest failures. However, few eggs that did not undergo management failed to a preventable cause, and accordingly, this future improvement analysis showed there is little to be gained.

On the other hand, managers could seek to improve management by judging threats more conservatively to avoid managing nests when they are not under threat. Because the probability distribution for no management overlapped little with those of other actions (Fig. 5.2), and few eggs that did not undergo management failed to a preventable cause, our results suggest that if a threat is not judged to be present, not managing is the best action. However, if they reduce the number of nests managed and only intervene when they believe eggs would be certainly lost otherwise, the improvement in fledging rate achieved by being more conservative could be offset by an increase in the proportion of nests being lost to a preventable cause. Whilst we don’t know where manager’s accuracy lies, these hypothetical trade-offs remain hard to solve, further illustrating the complexity and subjectivity of whether to intervene. Using live nest cameras or increasing surveillance could increase manager understanding of the true underlying risk of a nest being lost to a preventable cause, which in turn could increase their overall accuracy in classifying a nest as requiring management or not.

Alternatively, managers might seek to improve management to increase success for managed eggs. For example, overall efficacy of artificial incubation was low, countering its general perception as a ‘safe’ option (Assersohn et al. 2021). Conservation programmes for other taxa like American oystercatcher (Thompson et al. 2015) and North American sage grouse (Collins et al. 2016) have achieved higher hatching success in captivity. This suggests there may be room for improvement in tara iti, for example by using better equipment or strict harvest protocols instead of rescuing potentially jeopardised nests (Dolman et al. 2015). Our analyses of management optimisation, however, indicated that this may provide minimal benefit to fledgling production. Furthermore, threats that egg intervention mitigates for still caused 14% of egg and chick mortality after the point when further targeted intervention was possible. This mirrors findings by Collins et al. (2016), who ultimately concluded that head-starting oystercatchers was only appropriate if most reproductive failure happened at incubation.

Managers’ ability to select some alternatives was constrained by availability of nests at any given time. This means, first, that estimates of the effectiveness for active management options might be further biased, and second, that under current management the best-performing alternative (donation) is not always available, whilst the most commonly used alternative (artificial incubation) is more accessible but less effective at producing fledglings. Considering these points, managers may wish to explore more creative alternative management actions beyond our candidate set, for example improving efficacy of ‘shift’ management, which in some cases can outperform other alternatives and can be easily implemented. Indeed, other programmes have successfully kept tern eggs *in situ* by enhancing or securing nests in different ways to shifting, for example using nest platforms (Palestis 2014) or shell patches (Booker & Moxom 2019). This could be combined with improving habitat quality by keeping nest sites safe from predators and tides. Habitat quality is a critical limiting factor for tern populations (Raynor et al. 2012), and will form a central part of future tara iti management (Chapter 4).

Imperfect data challenge a manager’s ability to learn and adapt management. Incorrect assessments of the effectiveness of actions, compounded by uncertainty, can promote status quo bias (Schwartz 2020) and sunk cost fallacies, where managers pursue continuation of ineffective management simply to justify past investments (Samuelson & Zeckhauser 1988; Gregory & Keeney 2002). Deliberative thinking can reduce the risk of biased decisions but requires time and effort. We found even the simple counterfactual scenarios in our study cognitively challenging at times, with intricate hypotheticals and trade-offs. This “slow thinking” process can be greatly assisted by tools like decision trees to harness conservation monitoring data effectively and avoid biases (Hammond et al. 1998).

At the same time, decision support tools should be combined with increased acceptability of learning and changing course when actions were not effective (Catalano et al. 2019) as well as close collaborations between researchers and practitioners (Baylis et al. 2016). Framing management as adaptive from the beginning can also pre-empt messy data, setting up easier evaluation of outcomes. Adaptive options might involve different levels of risk. A more conservative option would be to choose the action that is believed to be most effective, but to use expert elicitation to formally state prior beliefs that can later be used as counterfactuals (passive adaptive management; Martin et al. (2012); Canessa et al. (2016)). A more risk-seeking option might be to deliberately choose not to manage some nests under threat and compare the treatment-control outcomes, taking some short-term risk to achieve greater long-term gains (active adaptive management; Runge (2011)).

Managing endangered species to recovery is a long process that requires ongoing decisions as populations change and programmes evolve. Leveraging information from imperfect data is the challenging reality of much conservation decision making. Even abundant data can be difficult to interpret, and psychological (Papworth 2017) and external barriers (Walsh et al. 2019) can hinder managers in making robust, informed decisions. Nonetheless it is imperative that programmes use their data to evaluate their management, to prevent poor decisions based on instinct, and thus wasting resources. For careful, deliberative evaluation of biased data, we recommend using decision support tools and multiple counterfactuals that incorporate uncertainty to promote rational thinking.

Chapter 6: Discussion

Key findings

Threatened species recovery is a major part of conservation but it has not always been effective as it could be (Clark et al. 1994; Dee Boersma et al. 2001; Bottrill et al. 2011). Species recovery groups often underestimate or ignore the complexities of the challenges they face (Clark et al. 1994; Milner-Gulland & Shea 2017). Ignoring social values and multiple objectives can lead to planning deadlock or to unsustainable outcomes, while uncertain and messy data can lead to biased inferences and sub-optimal decisions. In the last 20 years, uptake of interdisciplinary decision support tools to address these complexities has increased (Wright et al. 2020). Yet, applications of arguably one of the most powerful frameworks, structured decision making (SDM), are still mostly limited to environmental management in the United States and Canada. In this thesis, I have demonstrated the efficacy of a range of SDM principles and tools to address the complexities of species recovery.

With a population of under 40 individuals, tara iti is an example of extreme rarity. In the years prior to my PhD, planning by the tara iti recovery group and surrounding community was in a state of deadlock. This was at least in part due to unresolved complexities. Using decision analysis and SDM, I helped the tara iti recovery group solve these common problems in a collaborative, transparent and inclusive way. I led the community through to making their first management recommendation in over ten years, demonstrated how to successfully integrate the community's values into recovery planning, and investigated the efficacy and the potential of nest management and captive breeding, contributing towards the future conservation of tara iti.

For conservation to finally achieve its widely sought aim of increased inclusivity, recovery groups need clear, applied examples of interdisciplinary tools that integrate multiple values and support meaningful co-management. In Chapter 2, I demonstrated how SDM facilitates the integration of *mātauranga Māori* (Māori knowledge and perspectives) into decision making, using two case studies: tara iti and pekapeka. SDM is a values-focused approach, where decisions start with a clear, initial expression of group values as fundamental objectives. Once objectives were stated, it promoted information sharing, about mātauranga Māori and species biology alike, which in turn fed into the co-development of alternative management strategies that considered multiple objectives. Facilitated deliberation resolved linguistic uncertainties which were causing tension in the community, for example, around tara iti egg management. The Iwi representatives designed their own way of describing the impact of management on their values, which was then presented alongside, not after, other information such as biological consequences and cost. Done this way, evidence-based conservation becomes more inclusive. This chapter provided important methodological details on how to facilitate co-management of threatened species between different peoples. Embracing value pluralism using the appropriate tools generates a deeper understanding of the whole socio-ecological system and promotes shared and sustainable solutions (Bennett et al. 2019).

Captive rearing and release has been suggested as a recovery tool for tara iti. Yet, several conservationists consider it unsuitable *a priori* for a species that exhibits post-fledging parental care (Gummer 2003; Jones & Kress 2012). In Chapter 3, to inform conservation recovery options for tara iti and for other rare tern species, I evaluated lessons learned from a translocation trial of sooty tern and common noddy in Mauritius. I found that training terns and noddies to feed at a whistle, then providing supplementary feeding with the whistle when birds were free-flying, seemed to allow at least some individuals to survive for up to five weeks after fledging. This research provided an important first step in learning the utility of chick translocations for these taxa, and the logical next step would be to plan trials to assess and support long-term survival, in order to reach the fundamental objective of population establishment. These outcomes generated new evidence for practitioners and challenged a common but untested assumption. This ‘tension’ between best practice and creative thinking has been recognised as a key part of effective decision making: to ultimately select the best alternatives, one should first think creatively and without limitations about one’s options (Keeney 1996; Game et al. 2014).

In decision making for threatened species recovery, the failure to tackle complexity in values and scientific knowledge can lead to over-optimistic expectations, inaction or even conflict that derails planning. In Chapter 4, I demonstrated a full SDM iteration for tara iti recovery planning, showing how SDM excels at addressing the multiple complexities in such contexts. Tara iti management was in a deadlock, partly due to major uncertainty about outcomes and because stakeholder values had not been considered. With SDM, the group expressed their values, learned and deliberated together and worked with predictive models that explicitly depicted uncertainty. This combination successfully removed entrenched divisions and allowed the group to progress and recommend a strategy. They chose an alternative strategy that was a balance across the stakeholder objectives, reinforcing the idea that values drive our decisions, not science alone. The group's recommendation will now lead to the development of the first species recovery plan in over ten years. The way participation, information sharing and structured deliberation helped resolve disagreements and rebuild trust in the tara iti conservation community echoes findings elsewhere (Redpath et al. 2013; Meek et al. 2015). Because SDM is values-focused and addresses typical complexities in conservation, it ideally complements science-focused tools such as conservation evidence (Sutherland et al. 2004) in helping decision makers find solutions that work in the real world (Williams et al. 2020).

Monitoring data are inherently complex in threatened species recovery, and yet managers depend on them to make informed decisions. In Chapter 5, I demonstrated how managers can make sense of complex monitoring data to improve inferences about management efficacy. Previously, managers had to make decisions about egg management based on experience and cursory examination of biased datasets. I produced the first quantitative evaluation of more than twenty years' egg management. To visualise the problem and analyse these data against the fundamental management objective (improving productivity), I structured the complex decision process that managers undertake into a decision tree. I showed how, if bias in the data was not accounted for, it could be concluded that management was ineffective. Using a counterfactual analysis, however, I found that if managers are accurately detecting imminent threats to nests, management is in fact effective. Although this method could help managers understand the nuances of management, I concluded that there was little room to improve efficacy within the current set of alternatives. I suggested that a better, more adaptive approach would be to generate priors as counterfactuals using expert elicitation (Runge 2011; Canessa et al. 2016). This novel use of counterfactuals contributes to the growing research on non-experimental, post-hoc analyses of conservation impact evaluation, and has particular relevance for practical application since it is framed as a decision from the beginning. Furthermore, it responds to calls for better use of counterfactuals (Coetzee & Gaston 2021), since I treated them as multiple and uncertain, with different plausible outcomes (Bull et al. 2020) dependent (in this case) on threat detection.

Future directions

This thesis contributes worked examples of how decision-analytic tools can tackle complexity in decision making. Structured decisions are iterative processes; as such, this thesis opens up further opportunities of research and management improvement, for both tara iti recovery and conservation decision making in general.

Tara iti research and future management

The tara iti recovery group intends to implement its chosen management recommendation (as described in Chapter 4), but also to monitor outcomes to help assess future possible management changes. To use resources effectively and generate decision-relevant science, any further tara iti research should address key uncertainties that may affect these future decisions (Runge et al. 2011b). This can be done in a formal, adaptive management process, a natural extension of SDM (Runge 2011). To implement adaptive management, the group would extend the current decision cycle to include the following steps: 1) detail critical uncertainties that are currently affecting the decision, 2) design and implement monitoring to assess these, 3) update the predictive population models created in Chapter 4 with this information, 4) adapt future decisions based on the new information (Runge 2011). The Department of Conservation are overhauling their data management for tara iti following the SDM, and this could lay the groundwork for adaptive management, although generating decision-relevant science in this way is more likely when recovery groups have good academic connections (Ewen et al. 2013; O'Connell & White 2017; Wright et al. 2020).

During the SDM process described in this thesis, several sources of uncertainty emerged that could be explored further in future iterations of the decision cycle. First, it is uncertain whether birds will colonise newly created sites at the rate predicted by the biological experts (Chapter 4). This contributes to broader, structural uncertainty about whether the number of territories limits population growth, or whether the tern would start nesting colonially once numbers reached a critical mass.

Second, the tara iti recovery team have recommended development of new sites, but these sites are yet to be identified, and this will require a suite of considerations (Brooks et al. 2011). Applying SDM to site selection would be valuable, particularly since the consultations described in Chapter 4 highlighted strong values and beliefs in the community about site options. Climate change, related sea level rises and extreme weather pose a threat to seabirds like tara iti that nest on low-lying sandy beaches and spits (Hatfield et al. 2012; Dias et al. 2019). Incorporating predicted sea-level rises at proposed alternative sites into consequence prediction would be important. For example, Hatfield et al. (2012) produced quantitative predictions of future land loss in important seabird islands in Hawai'i by using lidar-derived landmass elevations and different scenarios of sea-level rise.

Third, the recovery team could explore uncertainty around how tara iti genetic issues might affect future management decisions by incorporating them into the predictive population models (Chapter 4). The high rate of infertility in the population could be linked to inbreeding depression (Baling 2008) and the variability of breeding success in the tara iti population (Chapter 5, Appendix B) may be indicative of variation in female fitness or relatedness of pairs. Making this component explicit in future decision cycles would help the group understand the genetic implications of alternative actions, for example the use of foster pairs (Chapter 5 – egg management) or splitting infertile pairs (Chapter 4). This would provide information for critical further *kōrero* (deliberation) between the recovery group and Iwi partners about the acceptability of splitting infertile pairs (Chapter 2).

Fourth, the implementation of a full adaptive management approach requires a long-term view on species management, and this can be challenging within institutional constraints (Clark et al. 1994; Game et al. 2014; Catalano et al. 2019). Short-term funding contracts and high staff turnover are common in conservation and limit the ability to learn and to build relationships (Hodge & Adams 2016; Catalano et al. 2019). An example in this study was the lack of clear information about long-term budgets for tara iti management (Chapter 4), which restricted the use of more advanced trade-off techniques (Chapter 2). The inability to secure long-term funding is a common reason why good decisions made using SDM may not be implemented (Wright et al. 2020). An interesting avenue of research would be to explore how recovery groups and their associated institutes could adopt long-term timeframes and adaptive management. Hodge et al. (2016), for example, propose a model of an institutional framework that could support adaptive governance for ecological restoration programmes.

Using decision science to address complexity in conservation

This thesis demonstrates the power of SDM in incorporating complex data and multiple values in a robust and logical way for decision making in conservation. It is both an approach and a set of tools which can help to overcome common biases, such as status quo bias (Chapter 4). It supports values-focused thinking, the co-production of knowledge and the integration of non-scientific information into decisions (Chapter 2), which could help redress the over-reliance on biological sciences (Kadykalo et al. 2021). There are other pieces of groundwork that can enhance the SDM approach. Structured decision making cannot completely remove biases (Gregory et al. 2012a; Canessa et al. 2019), so conservation institutes could replicate training carried out in other sectors to improve recognition of biases (Catalano et al. 2019).

Furthermore, there is room for conservationists (particularly Western scientists, such as myself) to work on inclusivity by recognising and reflecting on their own framing of conservation and latent values (Evans 2021; Pascual et al. 2021).

The SDM principles and tools described in this thesis are highly scalable: practitioners can adopt specific methods according to their needs. Apparently simple steps that may not require specific analyses, like the identification of objectives or the thoughtful generation of alternatives, might be enough to solve the bulk of problems (Keeney 2004). For the tara iti recovery group, the '*mana*-enhancing' core philosophies of SDM alone, such as group *kōrero* (deliberation) and *ako* (two-way learning, joint learning), had a visible impact on the way the community worked together (Chapter 2). Equally, the SDM framework supports the use of more technical steps, for example the tara iti recovery group needed the models to be detailed enough for their outputs to be useful and trusted (Converse & Armstrong (2016); Chapter 4). Giving practitioners a broad understanding of the SDM tool kit, and some practical real-world examples, could significantly assist clear thinking in conservation. I hope my work contributes towards this knowledge base.

In this thesis, I have demonstrated the efficacy of using interdisciplinary, decision-analytic methods to address different types of complexity in threatened species recovery. Throughout this PhD, I have worked directly with practitioners, and my work has supported the effective co-development of a new recovery plan for an endangered seabird. My aim is that this work provides a set of practical examples for conservationists seeking to improve deliberations between different peoples, co-produce information and recovery plans, and improve clear, rational thinking in the face of uncertainty.

Appendix A: Additional methods information for Chapter 2

Preparing for structured decision making

In both case studies, the SDM process began with stakeholder workshops over multiple days. The iwi Ngāti Manuhiri were involved with pekapeka as the *mana whenua* (holding authority over land) of Hauturu-o-Toi/Little Barrier Island and *kaitiaki* (guardians) of the northern pekapeka subspecies. They hosted the first two-day workshop in November 2014 on Te Kiri Ōmaha *Marae* (Māori meeting house/complex). The second two-day workshop was held in April 2015 at Auckland Zoo. The initial tara iti workshop was two days long, hosted by DOC in March 2018 in Kaiwaka sports complex, Northland. Subsequent smaller workshops with expert working groups were held throughout the following year. Two facilitators were present at each workshop. Ground rules were agreed upon with participants at the start of our meetings: acknowledgement that everyone's view was important; environment should be positive and forward moving; only one person speaking at a time; no phones. This helped to ensure that all voices present were heard.

Eliciting Values and Alternatives

We used simple brainstorming techniques to elicit values and alternative management strategies from the stakeholder groups. To elicit values and objectives, we asked participants to brainstorm individually first, to generate creative thought, encourage unconstrained thinking and capture everyone's concerns. Then in subgroups, followed by whole group deliberation. To assist brainstorming for objectives, we asked the following types of questions (Gregory et al. 2012a):

- What do you hope to achieve?
- What concerns are you trying to address?
- What would be the best outcome for you?
- What is the worst thing that could happen?
- What do you want to avoid in making your choice?

To assist the subgroups in separating their ideas into means and fundamental objectives, we asked them two questions: why is that important [and] how could we achieve that? In the open deliberation of the finalised set of fundamental objectives and their performance measures, we took time to ensure they were well understood, unambiguous and fit for purpose. To check this, we asked participants whether, if this information was collected, they would have what they needed to make an informed choice between different alternatives. To elicit possible management strategies, we asked participants to brainstorm in small groups. We encouraged participants to think innovatively without constraints other than the scope and scale defined in the goal statement, avoiding premature assumptions about efficacy of an action.

Eliciting Consequences and Trade-offs

Expert working groups were formed for each objective. We met with our expert groups in comfortable spaces, for example, meeting rooms in Department of Conservation buildings, or iwi offices. To estimate quantitative outcomes, such as biological consequences, where data were not available, we employed expert elicitation methods. To gather qualitative information, for example for indigenous values, we had facilitated discussions. For other objectives, such as social and economic values, expert groups used a mixture of elicitation, data sharing and deliberation to generate predicted outcomes. Below, we explain the methods we used to prepare the biological working groups and the elicitation method used across several of the groups.

Estimating biological consequences

Our first step was to model the current population(s). Facilitators built stage-based population models with help from biological experts. To predict how populations would respond to different alternatives, we needed to elicit vital rates from the biological working group. In advance of the workshops, facilitators prepared a list of all parameters that experts would be asked to provide estimates for. In both case studies, we first discussed existing empirical data and analyses, then trained the group in the elicitation method. Where needed, we discussed the meaning of different vital rates; for example, how model outputs translate into probabilistic statements. The group practiced the elicitation method using a trial run, where they were asked to estimate the survival rate of another species. After we were confident that the group understood the aim of the meeting, how to estimate vital rates and how to follow the elicitation procedure, the formal elicitation began. We used a modified Delphi elicitation technique (Hemming et al. (2018)). The groups followed three steps: Investigate, Deliberate, Estimate. For each parameter, each expert was initially asked to provide estimates (minimum, most likely and maximum values) individually then allowed to revise those after group discussions based on critically evaluated published and unpublished evidence. The groups repeated this process for each parameter required. The final step, Aggregate, was carried out by the facilitator after the elicitation was completed. Following best practice, we used the minimum, most likely and maximum estimates provided by experts to fit beta-PERT distributions for all elicited parameters (Vose 1996) and derived a "consensus" estimate for each parameter by linear pooling of expert estimates. We then used the consensus parameters in the model, propagating uncertainty by randomly drawing values in each simulation run (McGowan et al. 2011). We simulated the outcomes of each alternative strategy by multiple runs of the models with the respective parameters (1,000-10,000 runs depending on the objective and case study). In each case study, the ultimate result was a complete set of predictions for the different biological objectives, under each alternative strategy, which we summarized in consequence tables (examples in Table 2.2, Chapter 2).

Estimating non-biological outcomes

In both case studies, management costs for the alternative strategies were estimated by experts with prior experience costing species recovery programmes. For tara iti, cost was estimated, without uncertainty, as the average annual cost (in millions of NZ$) spread over the total 50-year management timeframe, and included: staff costs (salary, admin, vehicles, leave, training); contractors; equipment; large one-off costs such as aviaries. Excluded was: flights for staff; uncertainty contingency; corporate overheads; indirect costs; third-party funding contributions. For pekapeka, translocation costs summed , without uncertainty, and included only: staff costs (salary and travel, includes pre-translocation work); equipment; animal transport, care and food; large one-off costs such as aviaries. These were shared in the expert groups and finalised through deliberation. In the pekapeka case, we estimated the value of management strategies for advocacy by contacting the managers of candidate source/release sites, and asking them to provide (1) the known or estimated annual number of visitors to each site and (2) the known or estimated annual number of media stories covering each site.

Addressing trade-offs

In the pekapeka example, to facilitate multiple-objective decisions we aggregated outcomes across objectives with quantitative measures using the Simple Multi-Attribute Rating Technique (Keeney & Raiffa 1993; for example, see Ewen et al. 2015). Participants were asked to express their preferences about individual objectives as weights on a 0-100 scale. Objectives with weight 0 would be irrelevant; those with weight 100 would have maximum importance; those with equal weights would be equally important. Weights were then normalized for each participant. Figure S1 shows the (anonymous) individual weights, as well as an average value, which was used in the final SMART analysis; note the group agreed the average score would only be used to help the discussion and facilitate insights by the decision makers.

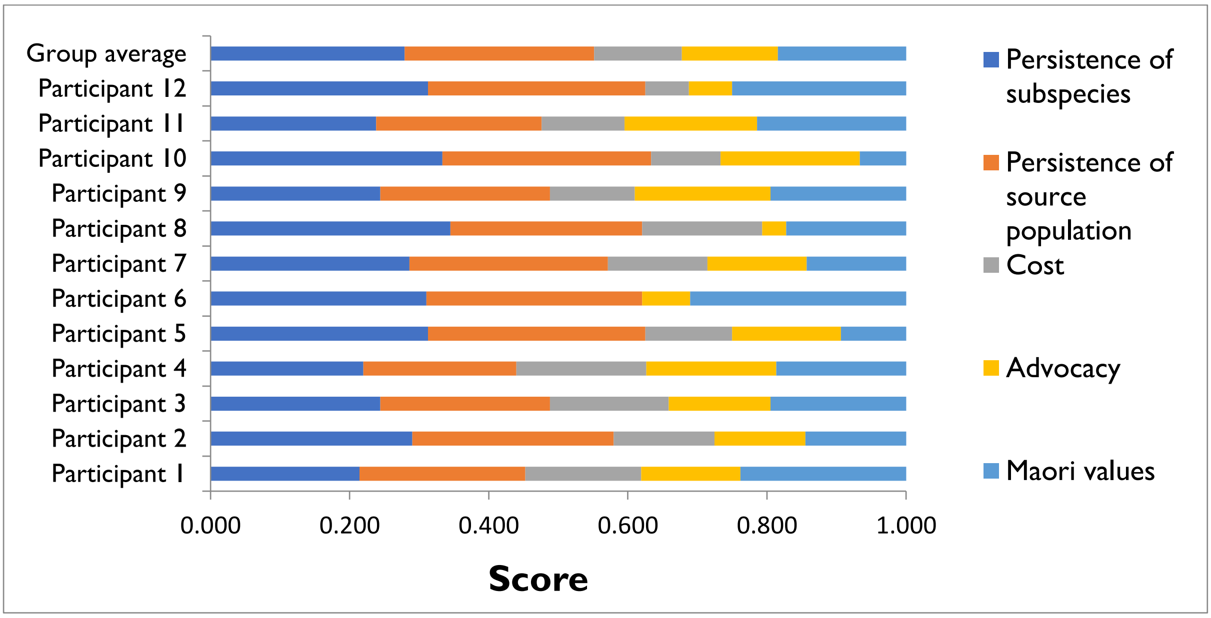

Figure S1. Subjective scores reflecting relative preferences for objectives. A greater value indicates higher importance e.g. an objective with a score of 0.4 is considered twice as important as one with a score of 0.2. The group agreed to use the average of these weights in a SMART analysis where they were multiplied against normalised predicted outcomes and summed for each strategy to give an aggregate score (Table 2.2, Chapter 2). Mātauranga Māori was not included in this numerical analysis (see Chapter 2 for an explanation).

Appendix B: Additional method information and analyses for Chapter 4

Analyses of tara iti vital rates

Productivity

Methods

Tara iti breeding productivity was estimated using DOC nest success data during 1997-2017. Clutches with fertile eggs (*n* = 170) were analysed to exclude bias (infertile eggs are deliberately not managed but still ultimately fail). To estimate the probability of an egg hatching and of a chick fledging, two Generalised Linear Mixed Models (GLMMs) were built using the package ‘lme4’ (Bates et al. 2015). Productivity was calculated as the probability of an egg producing a fledging, the product of these probabilities, to be inserted into the fecundity equation in the tara iti life cycle (Chapter 4, Fig. 4.1). We used binomial response variables for hatching probability (number of eggs hatched, number of eggs failed) and fledging probability (number of hatchlings fledged, number of hatchlings failed), a binomial error structure and a logit link function. They were fit by Laplace approximation (Bolker et al. 2009). Nest management was fitted as a fixed, binary variable (managed, unmanaged), alongside *a priori* hypotheses about other factors that affect productivity: age of breeding female and clutch attempt number (1-3). Models were fitted as global, ‘maximal’ models (Barr et al. 2013) and only simplified if there were problems with convergence and fit. The data are temporally and spatially replicated, as multiple measures are taken from the same individuals, across multiple years, and those individuals are grouped at sites. Both global models were therefore fitted with female, breeding site and season as random effects. Mixed effect models are known to be ‘data hungry’: in the hatch model, breeding site had to be removed from the hatch model, and season had to be removed from the fledging model, since the models failed to converge or gave singularity warnings when these were included. This indicates the models were overfitted, or in the case of site, the random effect not having enough levels (Harrison et al. 2018). A BOBYQA optimiser was also used to support convergence (Powell 2009). Models were validated and checked for fit using package ‘DHARMa’ (Hartig 2020). We used package ‘ggpredict’ (Lüdecke 2018) to extract probabilities and their uncertainty from the GLMM outputs. A range of other statistics were derived from the databases using the 'tidyverse' package (Wickham et al. 2019), and used to parameterise the population models (Table S3).

Results

We investigated the two global models to evaluate breeding success. The parameter estimates for egg management in both models were negative, indicating that egg management has a negative effect both on the probability of an egg hatching and of a chick fledging (Table S1). After the probabilities were extracted, we found that unmanaged eggs had a higher probability of hatch (p = 0.81, 0.68 – 0.89, 95% C.I.) than managed eggs (p = 0.6, 0.45 – 0.74; Table S3). We observed the same pattern for probability of a chick fledging, where chicks that were unmanaged at egg stage had a higher probability of fledging (p = 0.7, 0.52 – 0.84) compared to managed eggs (p = 0.44, 0.23 – 0.64; Table S3). There is a lot of uncertainty around the predictions, and the managed and unmanaged estimates' uncertainty bounds overlap. We also found an effect of age on hatch probability, with older females more likely to hatch an egg (Table S1, Fig. S2). This effect was less pronounced in the chick fledging success model, potentially linked to the female's larger role in incubation compared to chick rearing. In both models, we found a slight negative effect of clutch number, suggesting that probability of nest success declines with each breeding attempt within a season (Table S1). The annual mean frequency of nest management was 50.7% (± 4%, S.E. / ± 19%, S.D.). Overall propensity of tara iti to replace their first or second clutch was 43.3%. If the first clutch was lost, the propensity to relay was 52.2%.

Table S1. Fixed effect and random effect parameter estimates and standard errors from the maximal model for a) probability of an egg hatching, and b) probability of a chick fledging.

Model	**Fixed effect**	**Estimate**	**SE**
a) Egg hatching success ~ 1 + female age (scaled) + egg management + clutch number + (1 | female ID) + (1 | season)	intercept	2.0056	0.5097
	female age (scaled)	0.5396	0.2058
	egg management	-1.0284	0.3463
	clutch number	-0.4341	0.2877
	Random effect	**Variance**	**SD**
	female ID	1.02751	1.0137
	season	0.04188	0.2046
b) Chick fledging success ~ 1 + female age (scaled) + egg management + clutch number + (1 | female ID)+ (1| site)	**Fixed effect**	**Estimate**	**SE**
	intercept	1.0800	0.6113
	egg management	-1.0917	0.3960
	clutch number	-0.1605	0.3657
	female age (scaled)	-0.0829	0.1861
	Random effects	**Variance**	**SD**
	female ID	0.3940	0.6277
	Site	0.2372	0.4870

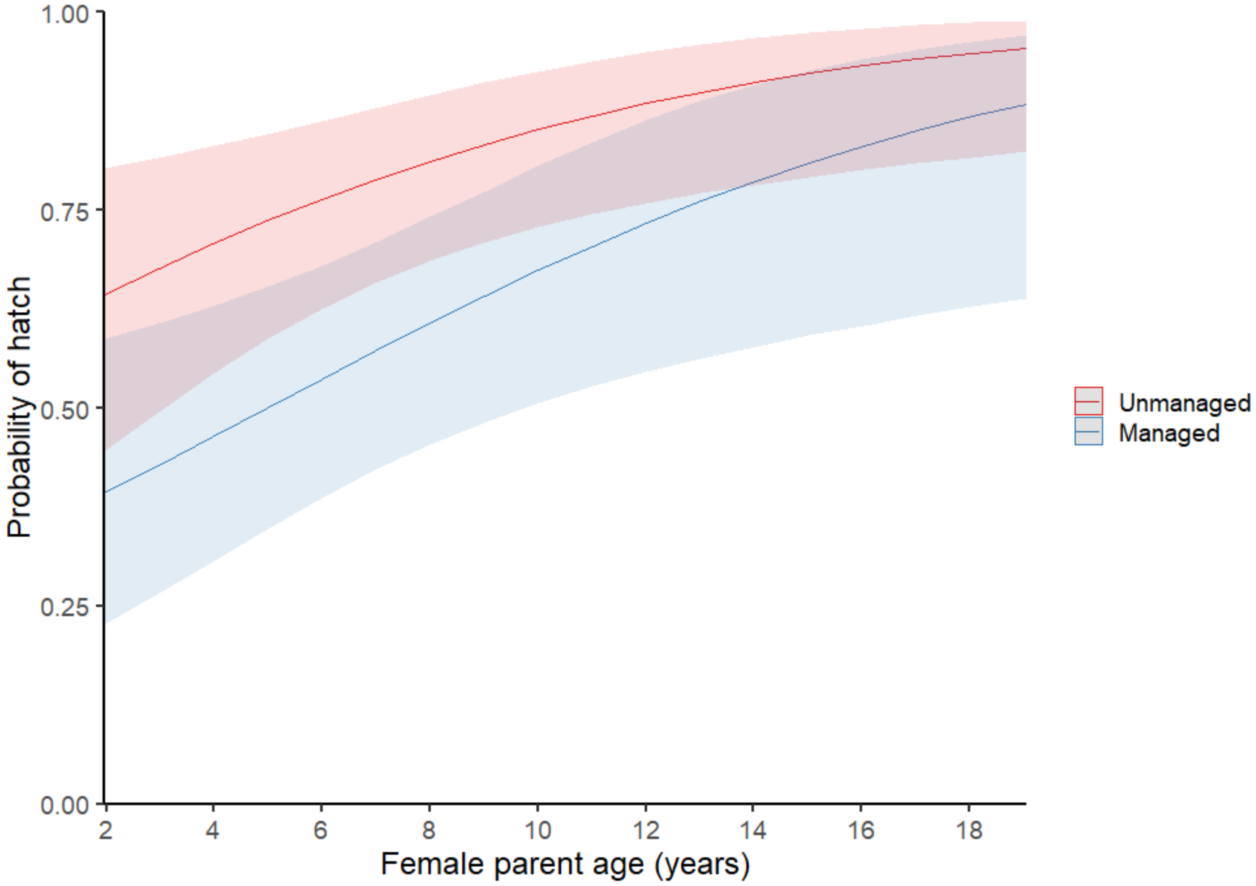

Figure S2. Probability of managed and unmanaged fertile eggs hatching as a function of female parent age.

Survival

Methods

We analysed tara iti survival using DOC sightings data for all birds ringed from 1991 (*n* = 149) in a Cormack-Jolly-Seber (CJS) model, using software programmes MARK (White and Burnham, 1999) and RMark (Laake & Rexstad 2014). The first mark for birds is when they are ringed as chicks. To generate yearly parameter estimates, sightings of juvenile, immature and adults were binned into year quarters, then only the fourth quarter was kept (October to December), to meet the assumptions of instantaneous capture in CJS models. Survival was modelled across three age classes: first time-step is fledged juveniles aged 0-1 years, second timestep is immatures aged 1-2 years then third timestep is adults, aged 2 years until senescence. These data were tested for goodness of fit using package 'R2ucare' (Gimenez et al. 2018) and the 'release' function in RMark. The global model analysed survival as a function of age, sex and time (year), and recapture as a function of age and time, giving 12 candidate models. The survival model list was then adjusted for overdispersion using the variance inflation factor (known as 'c-hat'). To generate robust annual survival probabilities that accounted for model uncertainty, the models making up the top 95% Akaike weights were averaged.

Results

No model had overwhelming support (top model Akaike weight was 0.34), so the top ten models were used in model averaging to reach the top 95% of weights (Table S2). We found a significant effect of age class on survival, but not of sex or time (year), (Fig. S3). Overall, immature survival was estimated as highest, but highly variable (female immature survival probability *p* = 0.93, 0.68 – 0.99, 95% CI). Adult survival was high and much less variable than immature or juvenile survival (*p* = 0.92, 0.86 – 0.95). This estimate was lower, however, than that estimated by Ferreira et al. (2005), who found the survival probability of adults to be 0.95 (± 0.02, S.E.). Juvenile survival was lowest (female *p* = 0.81, 0.55 – 0.93). The survival rate from fledging to adulthood was therefore higher than the estimate by Ferreira et al. (2005), who estimated that annual fledgling survival rate per annum until adulthood was 0.63 (± 0.05, S.E.). We used the estimates for female birds to parameterise the tara iti population models (Table S3).

Table S2. Cormack-Jolly-Seber model selection table, showing the top models (top 95% Akieke weights) that were used in model averaging, the null model (second to bottom row) and the global model (bottom row).

Model	K*	QAICc†	ΔQAICc	Weight	QDeviance
Phi(~age)p(~time)	30	493.0055	0	0.336725262	348.5475
Phi(~1)p(~time)	28	494.2358	1.230362	0.182014458	354.3961
Phi(~age)p(~1)	4	495.3446	2.339154	0.104552601	407.5005
Phi(~age + sex)p(~time)	32	495.9525	2.947084	0.077147977	346.8294
Phi(~1)p(~1)	2	496.0124	3.006966	0.074872316	412.235
Phi(~age)p(~time + age)	32	496.3425	3.337033	0.063481645	347.2194
Phi(~sex)p(~time)	30	497.3448	4.339367	0.038458665	352.8869
Phi(~age + sex)p(~1)	6	497.8667	4.861247	0.029625752	405.9172
Phi(~sex)p(~1)	4	498.6249	5.619458	0.020278035	410.7808
Phi(~1)p(~time + age)	30	498.79	5.784503	0.018671824	354.332
Phi(~1)p(~1)	2	496.0124	3.006966	0.074872	412.235
Phi(~age + sex + time)p(~time + age)	60	545.3716	52.36614	1.43E-12	325.5677

* number of parameters. †

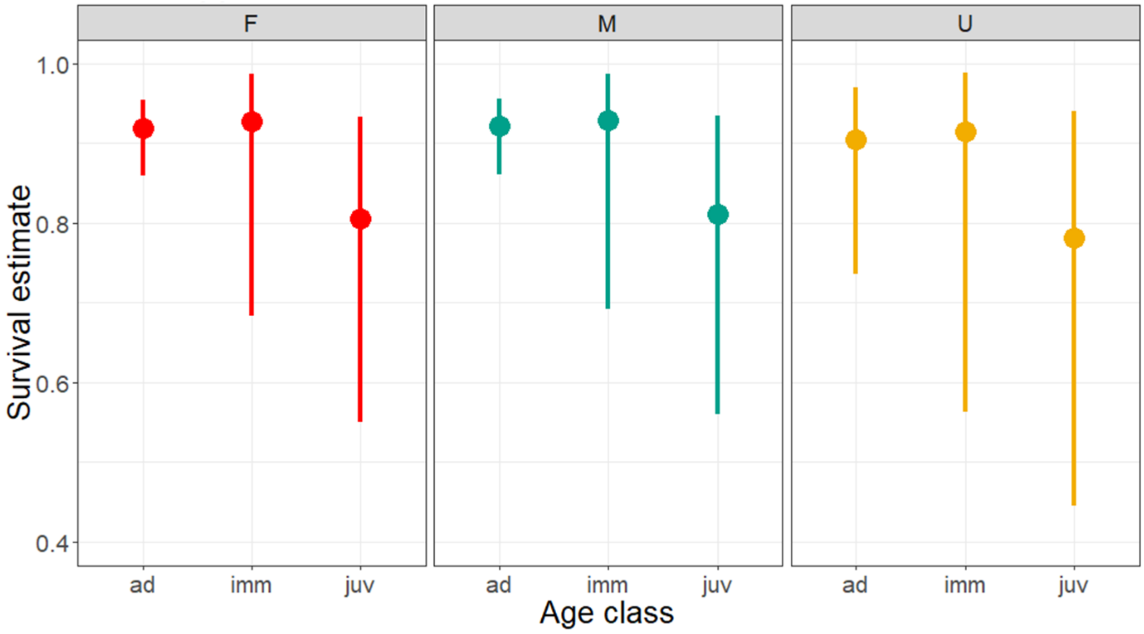

Figure S3. Model-averaged apparent survival estimates for tara iti as a function of age and sex, with 95% CI. Estimates shown for female (*n* = 60, F), male (*n* = 65, M) and unknown (*n* = 19, U) genders in age classes adult (ad), immature (imm) and juvenile (juv).

Table S3. Summary table of parameters used in the population models derived from empirical data using methods described above. Survival estimates are annual and for female birds.

Parameter / Statistic	**Lower 95% C.I.**	**Estimate**	**Upper 95% C.I.**
Juvenile survival	0.55	0.81	0.93
Immature survival	0.68	0.93	0.99
Adult survival	0.86	0.92	0.95
Hatch rate (wild)	0.68	0.81	0.89
Hatch rate (managed)	0.45	0.60	0.74
Chick fledge rate (wild)	0.52	0.70	0.84
Chick fledge rate (managed)	0.23	0.44	0.64
Overall probability of nest success (wild)	0.35	0.57	0.75
Overall probability of nest success (managed)	0.10	0.26	0.47
Mean clutch size (all breeding attempts)	1.68	1.73	1.78
Proportion of females age 2+ attempting to breed[§] (2007-2017)	0.64	0.72	0.80
Proportion of nests managed* (2007-2017)	0.41	0.55	0.69

[§] Attempting to breed defined as laying eggs. * Managed nest defined as nest shifting, use of dummy eggs, artificial incubation of eggs and transfers for fostering i.e. any movement of the eggs

Expert elicitation

Methods

Expert elicitation, an effective method for estimating consequences where no data are readily available (Martin et al. 2012a), was used to estimate quantitative outcomes for the biological and wider ecosystem objectives following the IDEA protocol, a version of the modified Delphi protocol, as described in Hemming et al. (2018). We held two day-long workshops with ten biological and five wider ecosystem experts, comprising Auckland Zoo, DOC field staff and technical experts, community volunteers and NGO volunteers. We started by discussing existing empirical data and analyses, and ensured the group understood how to estimate vital rates and how to follow the procedure.

In the workshops, the groups followed three steps: Investigate, Discuss, Estimate. For each parameter or value, experts were initially asked to 'investigate' by providing estimates individually (lower estimate, best guess and upper estimate), without discussion. They were also asked to specify how confident they were that the true value fell between the bounds they had provided (credible interval). This four-step elicitation helps to circumvent over-confidence in experts (Hemming et al. 2018a). Following best practice, we standardised the credible intervals provided by experts to 100% confidence to reflect expert uncertainty using the following equation (Hemming et al. 2018a), in which:

Lower standardised interval: $B – ((B-L) \times (S/C))$

Upper standardised interval: $B + ((U-B) \times (S/C))$

where B = most likely, L = best guess, U = upper estimate S = level of credible intervals to be standardised to, and C = level of confidence given by the participant. The groups repeated this process for each parameter required. After the first round, we collated the data and shared the results graphically, anonymising responses. In the 'discussion' step, they analysed these results, questioned them and discussed critically evaluated published and unpublished evidence, where applicable. They were then allowed to make a second, final and private estimate. The final step, 'aggregate', was carried out afterwards. The minimum, most likely and maximum estimates provided in round two, the 'estimate' round, were cleaned for mistakes and extracted to fit betaPERT distributions for all parameters (Vose 1996). A 'consensus' estimate for each parameter was derived by taking the arithmetic mean of expert estimates

(Hemming et al. 2018a). The summarised results for the biological parameters (Table S3-S4) were to parameterise all the population models, and the summarised wider ecosystem objective results were used directly in the consequences table (Table 4.2).

A second expert elicitation was carried out a year later with the biological experts to estimate different colonisation rates of tara iti depending on age class and site. This was carried out remotely: word documents with clear questions for the ‘investigate’ and ‘estimate’ step were sent by email to participants, who had two weeks to complete each of these and send them back. For the ‘discussion’ step in between, the graphical results were sent around the group by email and we held a facilitated conference call to carry out the deliberations and information-sharing. The data were cleaned, standardised and collated as before, and incorporated into the population models (Table S4).

Table S4. Elicited parameter values from methods described in above, used in the population models for alternative strategies. Uncertainty bounds are standardised to 100% confidence. Survival rates are annual and for female birds.

Parameter	**Mean minimum estimate**	**Mean best estimate**	**Mean maximum estimate**
Juvenile survival (Field 1)	0.53	0.82	0.94
Juvenile survival (Field 2)	0.56	0.86	0.96
Juvenile survival (in captivity)	0.67	0.87	0.95
Juvenile survival (captive-reared birds released at new sites)	0.30	0.53	0.76
Juvenile survival of released (captive-reared birds released at occupied sites)	0.36	0.63	0.86
Juvenile survival (OZFT)	0.48	0.82	0.91
Immature survival (Field 1)	0.66	0.91	0.98
Immature survival (Field 2)	0.64	0.93	0.98
Immature survival (captive-reared birds released at new sites)	0.47	0.73	0.84
Immature survival (captive-reared birds released at occupied sites)	0.47	0.71	0.81
Adult survival (Field 1)	0.83	0.91	0.95
Adult survival (Field 2)	0.85	0.92	0.95
Hatch rate (Field 1)	0.7	0.87	0.94
Hatch rate (Field 2)	0.71	0.89	0.96
Hatch rate (captive)	0.73	0.89	0.95

Hatch rate (OZFT egg)	0.40	0.59	0.70
Chick fledge rate (Field 1)	0.63	0.8	0.90
Chick fledge rate (Field 2)	0.66	0.83	0.92
Chick fledge rate (captive)	0.66	0.84	0.94
Chick fledge rate (OZFT chick)	0.53	0.66	0.73
Clutch size (Field 2, all breeding attempts)	1.70	1.79	1.89
Proportion of females age 2+ attempting to breed§ (Field 1)	0.64	0.74	0.78
Proportion of females age 2+ attempting to breed (Field 2)	0.64	0.77	0.83
Proportion of females age 2+ attempting to breed (Field 2 + Captive 3)	0.48	0.67	0.76
Proportion of nests managed* (all alternatives)	0.10	0.31	0.59
Number of territories in breeding sites (Field 1)	5.8	17.49	22.39
Number of territories in breeding sites (Field 2)	6.8	17.99	23.99
Number of territories in breeding sites (Field 2 plus two new sites)	8.25	23.71	31.98
Probability that adult dispersers without a territory will unlock† territories in range‡	0.36	0.59	0.79
Probability that wild/captive recruits released in range will unlock territories in range	0.34	0.56	0.74
Probability that captive recruits released outside range will unlock territories outside range**	0.1	0.39	0.6
Probability that adult dispersers without a territory will unlock territories outside range	0.04	0.2	0.37
Probability that wild/captive recruits released in range will unlock territories outside range	0.06	0.22	0.39

§ Attempting to breed defined as laying eggs. * Managed nest defined as nest shifting, use of dummy eggs, artificial incubation of eggs and transfers for fostering i.e. any movement of the eggs. † Unlock defined as a bird pairs and breeds in an empty, available territory. ‡ Inside range defined as current breeding sites or new breeding sites where birds in the living population have been seen but have not bred in the past 15 years.

Estimating biological consequences using population projection models

To estimate tara iti population outcomes under alternative strategies, we modelled the structure of the current population (Chapter 4, Fig. 4.1). All analyses and models were carried out in R (R Core Team 2020). We built a stage-structured, female-only post-breeding population model (adapted from Kéry & Schaub 2021) to predict population size and persistence probability under status quo management over a 50-year period, aligning with DOC's management strategy timeframe. We modelled three life stages as per the survival models (Appendix B). Fecundity was modelled as a function of: proportion of females attempting to breed; number of eggs laid; number hatching probability; fledging probability; and egg management (Chapter 4, Fig. 4.1). We parameterised the models using a mixture of empirical and elicited data (Table S3, S4). The initial population vector was derived from the population structure in 2017 (14 adult females, 3 immature females). We built in a function for density dependence, so that once the number of adult females reaches the maximum number of breeding territories (expert elicited estimates; Table S4), no more females could breed.

We used a hierarchical process for incorporating different types of uncertainty (temporal and demographic stochasticity, and parametric uncertainty) into the models. In the inner, individual-level loop, environmental and demographic stochasticity is incorporated into the calculation of population size in the following year (N_{t+1}) by drawing a value for each demographic rate from a probability distribution, using population size (N_t) and time-dependent life history parameter values from the time-step loop (outside it). For tara iti, the stochastic population equations are as follows:

$$N_{0,i,t+1} \sim \text{Poisson}\,(N_{\text{ad},i,t}\,\Phi_{\text{ad},i,t}\,f_{i,t})$$

$$N_{1,i,t+1} \sim \text{Binomial}\,(N_{0,i,t}\,\Phi_{\text{juv},i,t})$$

$$N_{\text{ad},t+1} \sim \text{Binomial}\,(N_{1,i,t}\,\Phi_{\text{im},i,t}),$$

where $N_{0,i,t}$, $N_{1,i,t}$ and $N_{\text{ad}\,i,t}$ is the population size of juveniles, immatures and adults respectively, in simulation i at timestep t; Φ_{juv}, Φ_{im}, Φ_{ad}, is survival probability of juveniles, immatures and adults respectively; and f is fecundity. Temporal stochasticity is estimated in the time-step loop. Demographic rates, drawn from the simulation loop, outside the time loop, are allowed to vary annually. Since the variance for survival estimates was unknown, a low value was given (0.001) to define background variation. For breeding parameters, the standard deviation of the random effect for season (time) in the GLMM models for hatch (Table S1) and fledge (not shown) was used. Finally, in the outer, simulation loop, we propagated parametric uncertainty by randomly drawing values from a statistical distribution of the mean with estimation uncertainty (e.g. standard error) in each simulated run of the model (i = 10000) (McGowan et al. 2011; Kéry & Schaub 2021). The mean and uncertainty values (Table S3, S4) were derived from empirical or expert elicited data, as described previously. We simulated this model over a 50-year period (t = 50). Population models were built for each of the alternatives and parameterised appropriately (code available at github.com/ThalassaMH/thesis-examination).

Incorporating colonisation probabilities

Models for alternative management strategies that contained the development of new breeding sites (Field 2 + Captive 3, Field 2 + Captive 3 and Field 2 + new sites) needed to incorporate probabilities of these being colonised. With the biological experts, we elicited probabilities of 'unlocking' a new territory (defined as a bird pairing and breeding in an empty, available territory) dependent on age, status (wild or captive bred), and the site in question (restored former breeding site or new breeding site; Table S4). These were used calculate the maximum number of territories available (carrying capacity, K) in each timestep (year). We let carrying capacity vary between two bounds. The lower bound is the number of territories available in the current breeding range, i.e., when none of the new breeding territories have been 'unlocked'. The upper bound is the lower of two values: a) the lower bound plus the product of number of dispersing tara iti and the probability of unlocking a territory, and b) the maximum number of territories available in the alternative (the mean elicited value, Table S4). In these calculations, we made the following assumptions: birds could not unlock new territories until the carrying capacity of the current breeding range had been filled; adults in the population only disperse when the population is above the lower carrying capacity (i.e. the current breeding range is 'full'); the probability that birds unlock a territory is calculated annually so carrying capacity can go down as well as up; juveniles do disperse but it is not relevant to the model because they cannot breed at that age. The carrying capacity was incorporated into the calculation of population fecundity as previously described.

Additional graphical representations of population outcomes

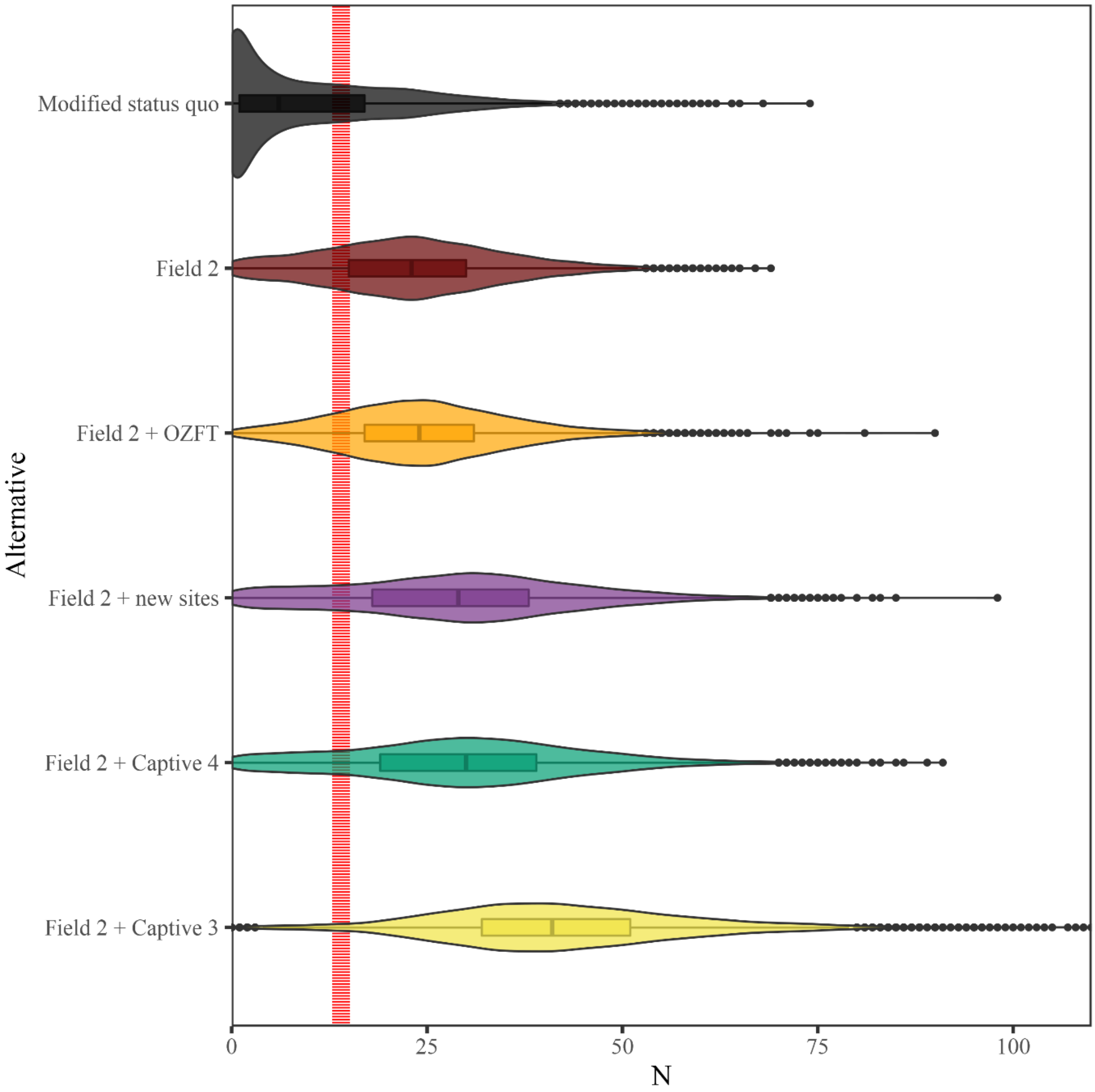

Figure S4. Violin plots showing frequency distributions of final adult population size N at *t* = 50. Red dashed line indicates adult tara iti population size in 2017. This figure assisted the recovery team in understanding uncertainty around final population sizes.

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