Exploring Complexity, Uncertainty, and Risk in Avian Reintroduction Decisions Through Structured Decision Making

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Abstract

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Avian species reintroductions are a vital tool for mitigating threats and avoiding extinction. However, the decisions required to reintroduce species are overwhelmingly complex. The complexity arises from many interacting factors, including challenges in identifying the specific decision to be made or the alternatives available to the decision maker, limited information about species biology or ecology, unknown effectiveness of management actions, risk of species loss, limited resources, multiple management objectives, multiple distinct management authorities, and lengthy time scales over which actions must be taken. Decision analysis, or structured decision making (SDM), offers a framework for gaining traction on

complex decision problems. An SDM process can provide decision makers with a clear view of the decision problem and a clearly defensible justification for the decision.

The research I present here explores the use of SDM for informing avian reintroduction decisions through the application of SDM to three case studies: vertebrate (primarily avian) restoration to the island of Guam, management of a reintroduced population of Hihi (Stitchbird; *Notiomystis cincta*) in New Zealand, and Streaked Horned Lark (SHLA; *Eremophila alpestris strigata*) reintroductions in Washington State. First, I facilitated a collaborative decision process to determine the problem structure of a complex decision problem regarding vertebrate species restoration in Guam (Chapter 2). Then, using the products of Chapter 2, I developed quantitative models to predict outcomes of alternative restoration strategies for two of Guam's vertebrate species (Chapter 3). Second, I employed SDM to address the challenge of a struggling reintroduced population of Hihi in New Zealand through the exploration of competing hypotheses about the cause of flat to slightly declining population trends (Chapter 4). Third, I used SDM to develop and evaluate reintroduction strategies for SHLA in Washington State (Chapter 5). Together, these case studies demonstrate how values-based decisions can be made by using all available scientific information to improve the likelihood of successful reintroduction outcomes. Further, my research demonstrates that uncertainty and complexity need not prohibit forward progress toward reaching management objectives.

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Chapter 1. Introduction

1.1 Background

Over the last few centuries there has been accelerated biodiversity loss worldwide (Ceballos et al., 2015) and a significant number of terrestrial vertebrates are predicted to disappear by 2050 (Ceballos et al., 2020). Island endemic and habitat specialist avian species have suffered particularly large losses, as they are disproportionately impacted by anthropogenic threats, such as habitat loss and invasive species (Clavero et al., 2009; Dueñas et al., 2021; Matthews et al., 2022; Pimm et al., 2006). Increasingly, conservation actions are being taken to mitigate these threats and avoid further avian species extinction (Hoffmann et al., 2010; Johnson et al., 2017), efforts which have had some effect in reducing extinction rates (Monroe et al., 2019). One such conservation action is conservation translocation, the intentional movement of species for conservation purposes. Reintroduction is a form of conservation translocation, wherein species extirpated from their historic range are restored through translocation (Seddon et al., 2012; Seddon and Armstrong, 2016). While reintroductions are a vital tool for combating extinction, the decisions required to reintroduce species are highly complex.

The complexity in species translocation decisions arises from many interacting factors including, but not limited to, challenges in identifying the specific decision to be made or the alternatives available to the decision maker, limited information about species biology or ecology, unknown effectiveness of management actions, risk of species loss, limited resources, multiple management objectives, multiple distinct management authorities, and lengthy time scales over which actions must be taken (Converse et al., 2013; Converse and Armstrong, 2016; Game et al., 2014; Keeney, 1982; Seddon and Armstrong, 2016). The inherent complexity and multifaceted nature of species or community restoration may hamper the identification of the particular decision(s) to be made (known as the decision problem) that are most relevant to decision makers at a given time and place (Converse and Grant, 2019; Paxton and Kraus, 2020). The alternative management strategies may themselves be composed of multiple component actions, complicating the generation of feasible and creative strategies for decision makers to consider (Aslan et al., 2014; Keeney, 1982; Samuelson and Zeckhauser, 1988). Uncertainty in natural resource problems can arise from the dynamic and stochastic nature of ecological systems, limited available information about species or communities, or unknown responses of a system to any of the numerous proposed management actions (Converse et al., 2013; Regan et al., 2002). If key uncertainties cannot be reduced, there is a meaningful amount of risk associated with implementing management actions (Canessa et al., 2020; Runge and Converse, 2020) and that risk must be traded off against limited resources available for either learning or acting. Common to many natural resource problems, there may be multiple decision makers and stakeholders involved, which can result in complex governance dynamics (Ascher, 2001; Estévez et al., 2015; McDaniels et al., 1999). Finally, for large-scale restoration efforts, management actions may need to take place over many years to decades, necessitating long-term institutional commitment and tolerance for a lack of short-term return on investment (Cortner et al., 1998; Du Toit, 2010; Wilson et al., 2016).

Decision analysis, also known as structured decision making (SDM), provides a framework for gaining traction on complex decision problems (Gregory et al., 2012; Hemming et al., 2022; Keeney, 1982; Runge et al., 2020). The structure of a decision-analytic process arises from the recognition that all decisions involve a consistent set of component parts, and that by breaking decisions into these component parts and engaging in a deliberative process, we can identify and address the impediments to decision-making, including complexity, uncertainty, and tradeoffs. The set of decision components common to all decisions includes a definition of the decision problem, objectives (what the decision maker wishes to achieve), action alternatives (the alternative strategies a decision maker may employ), predictive models (for predicting the outcome of alternatives in terms of the objectives), and some algorithm for identifying the best alternative for meeting the objectives (Figure 1.1). Decision analysts generally build decision frameworks by working through the components in this order, as a series of decision-analytic steps. Key to decision analysis is the inclusion of the subjective values of the decision makers and stakeholders, as is the inclusion of scientific information necessary to make predictions (Gregory et al., 2012; Hemming et al., 2022). Undergoing an SDM process will not eliminate uncertainty, but attention to the forms and characteristics of uncertainty is a principal component of the process. Identifying and accounting for uncertainties can, for example, help to identify action alternatives that are robust to uncertainty or indicate areas where collecting more information would be useful (Canessa et al., 2015; Converse et al., 2013; Runge et al., 2011). The SDM process can provide the decision maker with a clear view of the decision problem and a transparent and clearly defensible justification for the decision (Runge et al., 2020). Undergoing SDM framework development in a workshop setting where decision makers, stakeholders, and scientific experts collaborate and share information creates the possibility for greater consilience and collaboration across disparate groups (Garrard et al., 2017; Gregory et al., 2012).

1.2 Research objectives

The overall goal of my research was to explore the use of SDM for informing avian reintroduction decisions through the application of SDM to three case studies. Through my research, I aimed to offer tools to decision makers and stakeholders that will improve their ability to make reintroduction decisions. The specific case studies I explored were vertebrate (primarily avian) restoration on the island of Guam, USA; management of a reintroduced population of Hihi (Stitchbird; *Notiomystis cincta*) in New Zealand; and Streaked Horned Lark (SHLA; *Eremophila alpestris strigata*) reintroductions in Washington State, USA. In each case, management was encumbered by common characteristics of difficult conservation decisions, including multiple management objectives, complex alternative structures, limited available information, and uncertainty in species responses to management actions. Hence, through taking a deliberative approach to each of these case studies, appropriate decision-analytic tools were applied to overcome complexity and offer decision makers frameworks for improving decision quality for difficult conservation decisions.

Two standard decision-analytic approaches were applied across the case studies: rapid prototyping and expert elicitation. Additional tools were applied to meet the particular needs of each case study. A rapid prototyping approach involves progressing through the SDM steps in quick succession, iteratively revisiting each step to refine or modify the components as needed. A rapid prototyping approach is highly effective for identifying the structure of the decision problem and it produces a greater understanding and consensus about the predictive tasks that must be subsequently addressed (Garrard et al., 2017). In each of the case studies, I employed this approach in a collaborative process to develop the first three components in the SDM process, the definition of the decision problem, objectives, and alternatives (Figure 1.1) Second, I used expert elicitation to deal with limited data, a situation which is common in reintroduction decision problems because empirical information about species responses to novel management actions will not be available until the action is implemented. Expert elicitation is a formalized approach for obtaining expert judgements in cases where direct empirical data are unavailable and is recognized as critical in conservation decision-making (Martin et al., 2012; McBride et al., 2012). Elicited judgments provided key information to parameterize predictive models in each case study.

Each of the case studies presented here provides examples of how decision-analytic methods can be used to help inform management and improve decision quality for complex species reintroduction decisions. In Chapter 2, I facilitated a collaborative decision process to determine the structure of a complex decision problem regarding vertebrate species restoration in Guam. In Chapter 3 I expand on Chapter 2, with the development of quantitative predictive models based on the structure of conceptual models constructed in Chapter 2. Due to limited available information about species restoration and interactions with threatening forces, I parameterized predictive models with elicited expert judgements. Initial predictions will be vital for fueling discussion about next steps for Guam's conservation community, including identifying next steps for restoration, and pointing towards research that would reduce management-relevant uncertainty. Together, Chapters 2 and 3 will aid future decisions in Guam by providing a framework to improve communication among multiple cooperating agencies, synthesizing current information, and providing models that can be updated as new information becomes available.

In Chapter 4, I employed SDM to explore uncertainty about the cause of a struggling reintroduced population of Hihi, where the influences of threatening factors were not fully understood. A major obstacle for decision makers of this Hihi population was identifying which research to conduct that would allow them to better manage the population. Numerous competing hypotheses about the cause of the current and projected population trends were considered in a value of information analysis. The outcomes from the analysis provide decision makers with a list of the highest priority research areas that are likely to improve decision quality if uncertainty is reduced.

Lastly, in Chapter 5, I used SDM to develop and evaluate reintroduction strategies for SHLA in Washington State. I assessed alternative management strategies for release to a handful of potential release sites with models that integrate both empirical data and expert judgments. Key to this work was the consideration of post-release effects using expert judgements. The framework and tools developed in this chapter will be useful for making decisions about reintroduction efforts in the future. Additionally, this work provides a transparent cognitive framework for the managers and stakeholders involved in SHLA reintroduction.

1.3 Broader impacts

Together, these case studies demonstrate how values-based decisions can be made by using all available scientific information to improve the likelihood of successful outcomes. My work is directly applicable to management of the species considered and each case study's framework was formulated based on collaborative input from managers, stakeholders, and scientific experts. The transparent and intentional decision frameworks I developed also document the current values and current scientific understanding relevant to each case study. As more information is available or if values change, these frameworks can be revised. Because reintroduction is inherently uncertain, in each case study I handle uncertainty explicitly, offering insight into how to reduce uncertainty or make decisions under uncertainty while accounting for risk.

More broadly, through this research, I aim to provide examples, methods, and general insights that will contribute to the approaches and growing knowledge base focused on overcoming common decision-making impediments that result from the complexity and uncertainty inherent in conservation decisions. One of the major contributions of my research is demonstrating that uncertainty and complexity need not prohibit forward progress toward management objectives. Here, I demonstrate that developing a decision framework is crucial for momentum not only during the initial planning stage but also for established management programs. Another broad contribution of my research is illustrating that uncertainty is not a roadblock to understanding how alternative management actions impact objective outcomes. Specifically, I show that predictions about outcomes can be obtained when empirical information is limited or unavailable, using the judgments of experts. The predictive models in each of the case studies focuses on the explicit incorporation of uncertainty, providing predictions across the full range of possible outcomes and allowing the risk of poor outcomes to be assessed. Additionally, I demonstrate how quantitative predictive models based on qualitative models developed in a collaborative group setting can increase group understanding and buy-in regarding model outcomes. Lastly, my work demonstrates the improvements in decision outcomes when research and monitoring are placed within a decision-analytic context. Given that resources available for conducting research are often limited, research activities in a management context can most productively be centered on reducing critical uncertainties that improve decision quality.

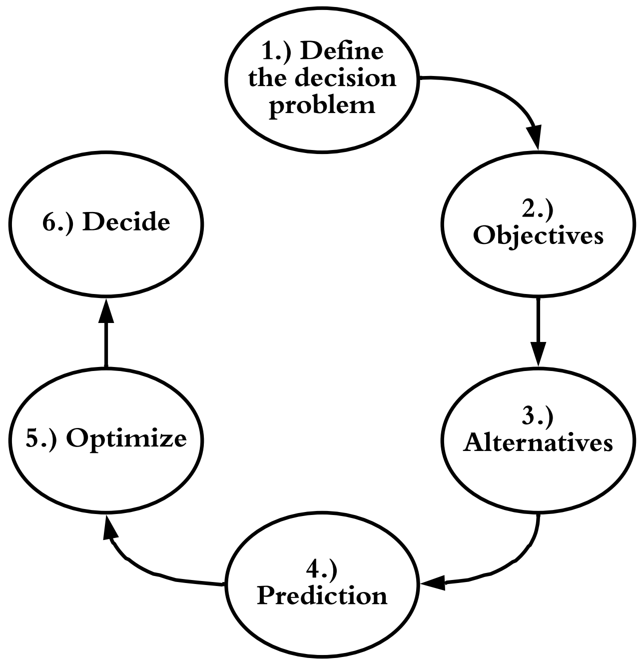

Figure 1.1 Diagram of the structured decision making (SDM) process (adapted from Converse et al. 2013). The basic elements in SDM include defining the decision problem to focus on, articulating the management objectives of the decision makers and stakeholders, generating feasible alternative management actions that may achieve objectives, predicting the influence of management alternatives on the objectives, and optimizing based on all other components to identify the preferred alternative.

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Chapter 2. PARADISE SOUGHT: THE COLLABORATIVE PATH TO RESTORING GUAM'S VERTEBRATES

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2.1 ABSTRACT

The decisions required to effectively implement broad-scale community restoration can be overwhelmingly complex. Decision-analytic techniques applied within collaborative working groups can guide development of practical frameworks to assist decision makers in effectively confronting complexity. We undertook a decision-analytic process relevant to vertebrate restoration efforts in Guam. Our goal was to develop a framework to guide future technical developments, actions, and monitoring by working collaboratively with management agencies, stakeholders, and scientific experts. We identified two key linked decision problems: where in Guam to focus restoration efforts and how to select the most appropriate restoration actions at any particular site. We identified sites that appeared promising for initial restoration efforts when focused on the 'where' problem. When focused on the 'how' problem, we identified a suite of management actions and developed conceptual models linking actions to species outcomes. Conceptual models were generated for Såli (Micronesian Starling, *Aplonis opaca*) and Ko'ko' (Guam Rail, *Gallirallus owstoni*) to guide future modeling efforts. This collaborative work provides a structure through which the agencies with responsibility for species management can evaluate feasible restoration actions, identify critical uncertainties, and develop the monitoring

plans needed to address those key uncertainties either before or in conjunction with active species restoration efforts.

2.2 Introduction

Increasingly, conservation managers are attempting to restore species and even entire communities impacted by large-scale threats (Hoffmann et al., 2010; Tittensor et al., 2014; Xu et al., 2021). Island species and communities are disproportionately vulnerable to threats, especially non-native species invasions, due to high endemism and relatively small land area (Kier et al., 2009; Tershy et al., 2015). As such, the restoration of island species or communities impacted by invasive species is a priority for conservation managers worldwide (Jones et al., 2016). The success of restoration efforts depends on the actions of managers with the legal and financial power to carry out large-scale management (Failing et al., 2013; Sapkota et al., 2018). Employing smart decision-making is key to the efficient use of resources for undertaking restoration (Converse et al., 2013b; Gregory and Keeney, 2002). Restoration in the face of invasive species threats is an immensely complex task (Prior et al., 2018; Zavaleta et al., 2001), even more so on inhabited islands (Glen et al., 2013; Oppel et al., 2011; Santo et al., 2015), and where invasive species cannot be fully eradicated (Armstrong et al., 2006; Hayward and Kerley, 2009; Lettink et al., 2010; Moseby et al., 2019, 2011; Parlato and Armstrong, 2018).

The restoration of native vertebrates to the island of Guam (Guåhan in the native Chamorro language) is emblematic of a complex conservation challenge. Guam is a United States territory in the Marianas Islands Archipelago of Micronesia (Figure 2.1). The brown treesnake (BTS, *Boiga irregularis*) was accidentally introduced to Guam post-World War II as a passive stowaway on military equipment (Rodda and Savidge, 2007). BTS predation caused the extirpation of 10 of Guam's 12 native forest bird species, and the extinction of two endemic species and one endemic subspecies (Savidge, 1987; Wiles et al., 2003). Three native bat species were historically present on Guam, but only the Fanihi (Mariana fruit bat; *Pteropus mariannus mariannus*) is known to occur in small populations on Guam today (U.S. Department of Navy, 2019). Native herpetofauna have also been reduced or extirpated by BTS (Campbell et al., 2012; Rodda and Fritts, 1992). Today, the food web in Guam forests is composed primarily of non-native and invasive species (Fritts and Rodda, 1998; Rogers et al., 2012), which has contributed to dramatic changes in forest structure (Caves et al., 2013; Mortensen et al., 2008; Rogers et al., 2017).

For several decades, research has focused on the ecology and management of BTS (Clark et al., 2012; Rodda et al., 1992; Savarie and Clark, 2006; Savidge, 1991, 1988). Eradication of BTS has remained elusive, but advances in research and management have resulted in the development of targeted BTS suppression techniques (Clark et al., 2017; Engeman et al., 2018). Fences have been developed to prevent snake incursions (Rodda et al., 1999) and toxicant bait drops have been developed to control BTS (Dorr et al., 2016; Siers et al., 2019). These two methods, perhaps in combination with targeted hand capture and removal (Christy et al., 2010) and trapping with various lures (Nafus et al., 2021; Yackel Adams et al., 2019), appear promising for reducing or locally eradicating BTS populations in exclosures to support species restoration (Nafus et al., 2022) while avoiding harvest-driven trait changes (Závorka et al., 2018) in BTS populations.

The complexity characteristic of species restoration problems is clearly evident in Guam. These complexities include substantial uncertainty, including uncertainty in the expected responses of species to management (Converse et al., 2013a, 2013b; Gregory and Long, 2009) and uncertainty about native species interactions (Palmer et al., 1997). Given these uncertainties, potential restoration actions entail meaningful risk, as the source populations are small and in some cases found only in captivity (MAC Working Group, 2013; U.S. Fish and Wildlife Service, 2008a, 2009). While restoration of species threatened by invasive species is most successful when the invasive species is eliminated (Seddon, 1999), eradication of BTS from large portions of Guam is not likely to happen in the near future due to the resource- and time-intensive management required to attain eradication (Nafus et al., 2022; Vice and Pitzler, 2000). Beyond BTS, other invasive species have significant potential to negatively influence reintroduced species (e.g., rodents, feral cats, feral dogs, and ungulates), but their interactions with extirpated native species are poorly understood (Fritts and Rodda, 1998). Lastly, numerous management agencies are critical to restoration, each of which has different regulatory mandates and variable ability to invest in restoration.

Despite the challenges and complexities, management agencies are interested in developing planning frameworks to support native vertebrate restoration in Guam. To maximize the effectiveness and efficiency of restoration, there is a need to develop planning tools to guide restoration management both within and across agencies. Management activities require time to devise, assess, implement, and evaluate, so starting these activities in advance of any desired restoration effort will reduce future delays. Failing to start the planning process as soon as possible may delay the time when necessary resources can be obtained and restoration actions can feasibly be implemented (Martin et al., 2012), potentially resulting in reduced availability of land for conservation, increased extinction risk for species subject to captive selection (Trask et al., 2021), further habitat degradation caused by the loss of seed dispersers (Rogers et al., 2017), or reduced public engagement caused by further erosion of the cultural connection to native species (Wald et al., 2019). It is sometimes thought that a lack of information makes it impossible or unproductive to plan for management, and consequently there is often a focus on "waiting until we know more" before restoration planning is undertaken. However, it has been noted elsewhere that a lack of information makes planning not less important, but more (e.g., Converse and Grant, 2019).

The decisions required to effectively implement restoration of a species or community affected by invasive species or other large-scale threats are overwhelmingly complex. The complexity arises from many interacting factors including, but not limited to, uncertainty in governance (i.e., who holds what decision-making authority), uncertainty about the specific decisions that must be made, multiple potentially conflicting management objectives, complex management alternatives, uncertainty and resulting risk, and lengthy time scales over which actions must be taken to achieve desired outcomes (Game et al., 2014; Keeney, 1982). Decision analysis, also known as structured decision making (SDM), provides a framework for gaining traction on complex decision problems (Gregory et al., 2012; Keeney, 1982; Runge et al., 2020). This process can provide the decision maker with a clear view of the decision problem and a transparent and defensible justification for the decision (Runge et al., 2020). Undergoing SDM framework development in a workshop setting where decision makers, stakeholders, and scientific experts collaborate and share information creates the possibility for greater consilience and collaboration across disparate groups (Garrard et al., 2017; Gregory et al., 2012).

SDM is valuable for contending with complicated decision problems (Runge et al., 2020), but success is highly dependent on getting the first steps in the process right: identifying a useful and relevant decision problem, and understanding the management objectives and alternatives. Undertaking a full decision-analytic process with a poor understanding of these decision components can lead to suboptimal outcomes, frustration amongst stakeholders, and lost time. A rapid-prototyping approach can help to guard against investing large amounts of time in solving the wrong problem (Garrard et al., 2017; Gregory et al., 2012). Rapid prototyping is an iterative approach that promotes a rapid progression through all the SDM steps, without concentrating undue attention on any one step, in order to develop an understanding of and consensus on the basic structure of the decision problem. Initial prototypes shed excess detail in favor of a broad and holistic view of the problem, allowing participants in a decision process to successively hone an adequately useful and realistic framework. In each successive prototype, the SDM steps are revisited and revised, and more detail is added to the components as needed to address the complexities of the decision. This method prevents stagnation in moving through the decision components or spending too much time working on a problem that may have been incorrectly defined, while allowing the decision makers to identify the obstacles to decision making. Rapid prototyping is particularly useful for collaborative decision processes, where input from multiple participants is critical but time is limited.

Our goal in this work was to develop a prototype decision framework for guiding the restoration of native vertebrates to Guam. We partnered with the U.S. Department of Navy (DoN) and other land managers, stakeholders, and scientific experts in a collaborative process to develop the decision framework. This framework is intended to guide future technical development, actions, and monitoring, and to encourage collaboration amongst management agencies, stakeholders, and scientific experts. We envision this framework as providing DoN, as well as other decision makers and stakeholders, with a transparent framework that can help to overcome the challenges that complexity imposes and facilitate a shared understanding of the concrete steps to be taken to move toward restoration. Our prototype conceptual framework includes a framing of the decision problem, management objectives, management alternatives, and conceptual models for understanding the effects of management on species. Here, we report on the collaborative process and its outcomes, and point to next steps.

2.3 Framework development

2.3.1 Participants and workshop format

Restoring ecological function and extirpated vertebrates to the island of Guam is a goal of conservation agencies in Guam, including the Guam Department of Agriculture, Division of Aquatic and Wildlife Resources (DAWR) and the U.S. Fish and Wildlife Service (USFWS). As a significant landowner on the island (Letman, 2016), the DoN is a major contributor to BTS research and management, and has articulated restoration goals and commitments (U.S. Department of Navy, 2019). We identified workshop participants through conversations with agency representatives and scientific experts, selecting participants to achieve a makeup of management agency decision makers, stakeholders, and subject-matter experts. Joint Region Marianas (JRM) provides executive-level management support to the U.S. Department of Defense (DoD), including DoN installations in Guam, and will be a key decision maker for restoration efforts in Guam. The USFWS and DAWR have regulatory authority over wildlife on Guam, and they along with the Commonwealth of the Northern Mariana Islands Department of Fish and Wildlife (CNMI DFW) partner with JRM to conserve natural resources on DoN lands in the Mariana Islands through research and management projects. The working group was composed of representatives from JRM (n = 4), USFWS (n = 3), DAWR (n = 1), CNMI DFW (n = 1), and scientific experts from outside these management agencies (n = 5). Participant roles, names, and affiliations are provided in Table A2.

The Covid-19 pandemic prohibited the working group from conducting in-person workshops as originally planned. We adapted by holding remote meetings, where the group could work through components of the SDM process using a rapid prototyping approach. The complete working group met 7 times between June 8, 2020, and August 17, 2020. Four additional meetings were held with smaller groups to work through specific components. A final meeting was held with group members in April 2021 to review project outcomes and identify future directions.

2.3.2 Problem definition

Developing a clear decision framing is a critical step in decision analysis, with all other components dependent on the framing (Gregory et al., 2012). In this case, no specific restoration decision by any one decision maker was conceptualized. Instead, the focus was on anticipating the general types of decisions that would be faced by a variety of decision makers working towards restoration in the future.

Through structured discussions, the group identified two general, and linked, decision problems. Linked decisions are decisions in which one decision influences the context and analysis of the other. In this case, we describe these two linked decision problems as, 1) the *where* problem, and 2) the *how* problem. The *where* problem considers the selection of a portfolio of restoration sites to meet a variety of restoration management objectives across Guam. The *how* problem conditions on selection of a particular site and focuses on the selection of specific restoration actions at that site. These decisions are linked because the *where* problem – i.e., selection of sites – will influence the context of the *how* problem, i.e., the characteristics and conditions at a particular site are likely to dictate the feasible actions to be considered. For example, if a species is reliant on a habitat feature not present or possible to create in a selected site, the species is not a viable option for the site and would not be favored in analysis of the *how* problem. Similarly, the predicted outcomes of the *how* problem, e.g., likelihood of successful restoration at a particular site, influences the analysis of the *where* problem, i.e., a decision about whether a particular site should be selected. The working group began by approaching the *where* problem and later turned their focus to the *how* problem once the *where* problem was better understood.

2.3.3 Analysis of the Where *problem*

A smaller group worked through a rapid prototype of the *where* problem over the course of a few weeks, bringing the findings back to the larger group for review and discussion. The decision problem was identified as: what are the initial areas where managers can establish populations, learn, and create partnerships for long term conservation? The smaller working group identified four objectives that the thought likely to be relevant to decision makers considering this decision problem: 1) maximize native vertebrate restoration, 2) maximize native forest restoration, 3) maximize public, monitoring, and research access, and 4) minimize cost. Likely objectives were identified while recognizing that objective identification in a specific decision context is within the purview of the decision maker(s). The alternative sites identified by the smaller working group were divided into two groups. The first group was composed of conservation areas where management of invasive species (e.g., fencing, toxicant drops, etc.) or habitat was either ongoing or planned for the near future; we refer to these as “first-tier” restoration sites. The second group was, more broadly, federal or local government lands allocated for conservation, but without ongoing or specifically planned invasive species or habitat management; we refer to these as “second-tier” restoration sites. Available habitat, management, and landowner information for each of the identified areas were summarized for first-tier sites (details below). Maps were created to share with the working group for discussion (Figure 2.2). We were limited to participant knowledge and publicly available information about land parcels, and as such we may not have had comprehensive information about planned conservation or management activities on all sites.

In discussion of the *where* problem with the full working group, there was general agreement to focus discussions around first-tier sites. Given the very limited experience with restoration in Guam, and the expense and technical challenges of establishing management infrastructure (e.g., anti-predator fencing), the working group judged that, at this point in time, extensive vertebrate restoration projects at second-tier sites were unlikely to gain support without first demonstrating success at first-tier sites. Thus, we focused subsequent discussions on first-tier sites where initial restoration efforts could lead to experience, learning, and early victories. Further, the habitat and size of at least some of the first-tier sites offered promising conditions for a number of the vertebrate species of interest (Thierry and Rogers, 2020).

The first-tier sites identified in the *where* problem analysis included the Habitat Management Unit, the North Finegayan Site, the Ritidian Unit, and the Anao Conservation Area (Figure 2.2b). The Habitat Management Unit is a 55-ha parcel located on Andersen Airforce Base and administered by DoN. The area is comprised of disturbed native limestone forest, has an existing BTS one-way exclusion fence (i.e., snakes can leave but not enter), has been treated with aerial toxicant drops and other methods of BTS control (Dorr et al., 2016; Siers et al., 2017), and is ungulate free (N.B., the fence surrounding the HMU was severely damaged during Typhoon Mawar in May 2023). The Habitat Management Unit falls within the Guam National Wildlife Refuge (GNWR) Air Force overlay unit, which constitutes DoN lands suitable for providing habitat for federally listed species, in cooperation with USFWS (U.S. Department of Navy, 2019; U.S. Fish and Wildlife Service, 2010).

The North Finegayan Site, a 976-ha parcel also on DoN land, contains native and degraded limestone forest. It is adjacent to JRM's Hapatu Ecological Restoration Area and also lies within the Air Force overlay unit of GNWR. Land within North Finegayan has, in recent years, undergone forest enhancement and ungulate fencing and removal, with multi-species exclusion barriers planned for conservation areas within the site (U.S. Department of Navy, 2019).

The 385-acre Ritidian Unit on GNWR is managed by the USFWS, contains native and degraded limestone forest habitat, and is designated critical habitat for Fanihi, Sihek (Guam Kingfisher, *Todiramphua cinnamominus cinnamominus*), and Åga (Mariana crow; *Corvus kubaryi*; U.S. Fish and Wildlife Service, 2004). Managers of the Ritidian Unit have proposed constructing a multi-species barrier and conducting invasive species control within the barrier, to establish an area for vertebrate and native plant restoration. The Ritidian Unit also contains limestone cave habitat for cave-dependent species, e.g., Yåyaguak (Mariana Swiftlet; *Aerodramus bartschi*; U.S. Fish and Wildlife Service, 2010).

Lastly, Anao Conservation Area was designated as one of three areas for conservation owned by the Government of Guam. Anao is approximately 764 acres, with native and degraded limestone forest habitat. The site allows limited public access (Guam Division of Aquatic and Wildlife Resources, 2006). Anao has been open to ungulate hunting in the past (Guam Division of Aquatic and Wildlife Resources, 2006), and while there is no ongoing predator control there, of the conservation lands owned by the Government of Guam, Anao has been identified as having the greatest potential for conservation management due to the habitat characteristics and location (L Duenas, DAWR, pers comm; Thierry and Rogers, 2020). As such, it was included in our first-tier sites.

Our prototype analysis of the *where* problem was intended to provide a broad, not exhaustive, understanding of the challenges associated with selection of spatial units for conservation, and to provide the group with enough information to determine how to focus further analysis. Development of a shared understanding of the availability of conservation areas led to the group conceiving of conservation opportunities on a small number of first-tier sites throughout the rest of the process. Consideration of the *where* problem also led to the recognition that landownership would be influential in many aspects of restoration management, and this recognition shaped discussion of objectives for the *how* problem.

2.3.4 Analysis of the How *problem*

With the first-tier sites from the *where* problem analysis in mind, the working group shifted focus to the problem of how to select the most appropriate restoration actions at a site. We began by identifying a broad set of management objectives that working group members thought likely to be important to decision makers when selecting a restoration strategy for a site. These objectives included: 1) maximize native vertebrate restoration, 2) maximize native forest restoration, 3) maximize ecosystem function, 4) maximize public access, 5) maximize learning, and 6) minimize cost and the use of other limited resources (e.g., staff time). The group recognized that these six objectives are likely to have differing weights (i.e., differing relative importance) depending on landowner and agency mandates at any given site. Thus, while the identified objectives represent a broad set, they are not intended to indicate a one-size-fits-all approach to decision making across landowners and sites.

Restoration approaches at any given site are likely to be quite complex, with multiple components, including the target species, the type of predator control, the type of habitat management, the source of individuals for translocations, and more. Complex alternatives are characteristic of restoration actions, particularly reintroductions, and alternatives in these cases are known as strategy-based alternatives (Converse and Grant, 2019). Strategy-based alternatives are built by choosing one or more elements under each of multiple components. These components and elements can be usefully organized into a strategy table (see Table 2.1; Gregory et al. 2012). The number of alternative management strategies is large – it is composed of all feasible combinations of elements under each component in Table 2.1 – and therefore the strategy table is critical for making this complexity manageable. The strategy table components for the *how* problem included: target species (including species of birds, mammals, and reptiles), population management (e.g., translocation type), fencing, BTS control method, non-BTS invasive species control method, native predator control method, habitat restoration actions, and education and people management. Certain elements may be inappropriate for inclusion in management alternatives at a specific site due to the mandates of the landowner or because the site is not appropriate for a particular species or action. Similarly, certain sets of elements would be unlikely to make sense in the same alternative (e.g., we would never choose "artificial nest boxes" if we were going to include only reptiles as focal species at a site), so building alternative strategies from the strategy table requires more attention than simply creating all possible combinations.

2.3.5 Conceptual models – influence diagrams

A key component of decision analysis is the use of scientific information to predict how alternatives will differentially affect the realization of management objectives. The qualitative ways in which alternatives influence objectives can be organized and visualized using influence diagrams (Howard and Matheson, 1983). This conceptual modeling tool links alternative actions to objectives through mechanistic pathways in an intuitive way, representing the consequences of alternative actions in terms of the objectives the decision maker cares about (Gregory et al., 2012). Influence diagrams are useful in collaborative settings to help organize objectives, alternatives, external factors, and uncertainty (Bernard et al., 2020). Influence diagrams serve as a precursor to quantitative predictive models and can be refined through the rapid prototyping process.

Given the primacy of species persistence as a management objective in this decision context, the working group focused on building tools for making predictions about the effects of restoration management on species establishment and persistence on Guam. Two species, Ko'ko' (Guam Rail, *Gallirallus owstoni*) and Såli (Micronesian Starling, *Aplonis opaca*), were selected by the group as case studies, and the group collaboratively built influence diagrams linking management actions to the objective of species persistence for these two species.

The first species selected was the Ko'ko' (Guam Rail, *Gallirallus owstoni*). The Ko'ko' is endemic to Guam, listed as endangered under the Endangered Species Act, and is found in captivity and in two locations in the wild: one introduced on Cocos Island (Islan Dåno in the native Chamorro language), just off the coast of Guam (U.S. Fish and Wildlife Service, 2008b), and another conservation introduction on Rota, which is designated as experimental, non-essential by the USFWS (U.S. Fish and Wildlife Service, 1989). The Ko'ko' has not been reestablished in Guam itself. The Ko'ko' is a flightless, ground-nesting, generalist omnivore that was historically found across the island of Guam. Due to successful captive breeding and translocation of the Ko'ko', the information available for understanding the impacts of management activities is more robust than for several other Guam endemic birds (e.g., Sihek). The Ko'ko' is a culturally important species and the species' recovery plan provides regulatory support for reintroduction to the island of Guam.

The second species selected for modeling was the Såli, a species which is present on Andersen Air Force Base on Guam, with recent increased observations in urban environments in Northern and Central Guam, and a small population on Cocos Island (Pollock et al., 2021). Populations are also found elsewhere in the Mariana Islands. Såli could be reintroduced to a new site or could be encouraged to naturally recolonize through habitat and predator management. Såli are generalists, with a high reproductive capacity and the ability to promote native forest restoration or perpetuate invasive plant species through seed dispersal (Caves et al., 2013). Given their continued presence on Guam and elsewhere in the Marianas, there is relatively robust information available from studies of Såli interactions with non-native predators, effectiveness of anti-predator nest boxes (Pollock et al., 2019), and dispersal of seeds (Pollock et al., 2020). Såli on the other Mariana Islands are plentiful, resulting in low risk of global extinction for this species. The species is not listed under the Endangered Species Act, which would minimize the regulatory hurdles associated with translocating Såli (MAC Working Group, 2013).

The Ko'ko' and Såli were used as starting points to understand the consequences of management actions on the native vertebrate restoration objective. Smaller working groups met twice to construct influence diagrams for each species. Consistent with the rapid prototyping approach, the smaller working groups focused on the most important dynamics dictating how populations would respond to the management activities presented in Table 2.1. The web-based software Lucidchart (www.lucidchart.com) was used to facilitate group collaboration during these meetings. After the group work on the influence diagrams, the diagrams were refined to ensure consistency and internal cohesion by the process facilitators.

2.4 Process insights

Here we report on the structure of the two species-specific influence diagrams. Management actions that were considered not relevant for a species, based on participant input, were not included in influence diagrams. For both species, abundance of the species was included as the objective, recognizing that in a dynamic population model, modeling abundance of the species over time allows for the projection of extinction risk (e.g., Morris and Doak, 2002).

Demographic processes, including breeding success (number of chicks per adult), and stage-specific survival (fledgling, juvenile, and adult survival) were included as the highest-level demographic processes in the influence diagrams. We assumed that there would be a carrying capacity for each species at a given site, influenced by the specific habitat components at the site. Immigration was assumed to depend on translocations or facilitated movement of individuals into the site, whereas emigration was assumed to function, for practical purposes, as a form of mortality (i.e., loss of individuals from the site). For each species, relevant management actions were modeled as either directly or indirectly influencing one of these five processes (breeding, survival, carrying capacity, immigration, or emigration). For instance, the management action of planting native trees indirectly influences survival through its effects on food quality and quantity (Figures 2.3 and 2.4).

Certain ecological and management processes were identified by group members as having differential effects on the two species. For example, the group thought that the ecological processes governing response to release and population management would be species-specific (Figure 2.5a). Also, Ko'ko' survival and breeding success were thought to be sensitive to anthropogenic mortality (e.g., illegal hunting) and direct disturbance, whereas Såli were not thought to be highly vulnerable to anthropogenic mortality or disturbance (Figures 2.4 and 2.5c). However, a larger suite of management actions were thought by participants to have somewhat similar influences on both species, as follows:

- Habitat restoration activities, such as planting native trees and removing non-native plants, were identified by group members as having an influence on the habitat quality and food availability for each species. These management actions were thought to impact survival and breeding success (Figure 2.5a).
- BTS fencing, multi-species barrier fencing, and control of a variety of native and non-native predators (including BTS, cats, rats, and other species) were thought to influence predation, and thereby survival and breeding success, across life stages for both species (Figures 2.5b).
- Group members thought that management of ungulates via ungulate or multi-species fencing, or via suppression or eradication, would influence breeding success and survival of both species. Ungulates were thought to impact food quality and availability, hiding cover or roosting sites, carrying capacity, and nesting through their effects on vegetation communities (Figures 2.3 and 2.4).

The influence diagrams reveal actions that were thought likely by group members to have important influences across species: those that exclude or control key shared predators and those that restore habitat. One single management action that appears likely to have multiple benefits is a multi-species barrier fence, through exclusion of predation by non-native predators and exclusion of ungulate impacts on habitat (Figures 2.3 and 2.4).

2.5 Discussion

Conservation problems can be daunting in their complexity (Game et al., 2014), but through decision-analysis, cognitively tractable decision frameworks can be developed to guide restoration (Garrard et al., 2017). A structured decision making framework is beneficial in the early stages of conservation planning to help teams develop a shared understanding of the key elements of a conservation challenge (Guerrero et al., 2017; Martin et al., 2018). We developed a prototype decision framework to provide DoN and its partners with a structured way to think about species restoration in Guam. While this framework encompasses two of the key decisions that must be made to get underway with restoration: *where* to do restoration and *how* to do restoration, we fully expect that the decision framework will evolve over time as more information is gained through research and practice. Yet, this effort lays the groundwork for future development and collaboration, providing a common framework to spur progress toward restoration goals (Gregory and Keeney, 2002).

A rapid prototyping approach to framing the problem allowed the working group to determine the most useful elements on which to focus discussion (Garrard et al., 2017; Gregory et al., 2012). In the framing of the *where* problem, we recognized that, after the initial pass at identifying potential restoration sites in Guam, our efforts were better spent on understanding the *how* problem. Once we identified first-tier sites to consider for restoration, additional work on site identification and evaluation would not have provided more clarity, because only the few first-tier sites were thought to be practical alternatives in the near term. While this was the shared opinion of participants, we recognize the risk of too readily adopting perceived constraints

(Gregory et al., 2012), such that there may be value in engaging more local stakeholders in the identification of potential restoration sites.

The group decided to focus mainly on the native vertebrate restoration objective in analysis of the *how* problem, as achieving this objective was the motivation for our process and is hindered by substantial uncertainty. We developed a strategy table to support the identification of alternative management actions for the *how* problem. Due to the large number and complexity of available alternatives, using a strategy table provided the group with a cognitively tractable way to think about alternatives (Converse and Grant, 2019). Consistent with rapid prototyping, the strategy table included broad rather than detailed actions (e.g., general approaches to translocation of animals rather than detailed capture, transport, and release plans), though detailed plans will need to be developed in the future ahead of any specific translocation project.

Conceptual models allowed the group to visualize the main processes by which actions may influence demographic rates and therefore species abundance. The collaborative approach to model development resulted in an atmosphere conducive to sharing ideas about threats and effects of management for each species and allowed the substantial collective knowledge of group members to shape the models. The resulting influence diagrams provide a clear visual illustration of the complexity of vertebrate restoration in Guam. For instance, while BTS predation is a chief concern for vertebrate restoration, the conceptual models demonstrate that it is only one aspect of the challenge. If left unaddressed, threats other than BTS may well obstruct successful restoration. Moreover, many threats will interact, e.g., the dynamics between rodents and BTS will complicate how community structure shifts in response to eradication or control efforts (Zavaleta et al., 2001). As such, a better understanding of relationships between species demography and factors such as habitat quality and non-BTS predation would vastly improve our ability to make predictions about effective restoration approaches. Although we constrained the models to be simple, there are still many interacting processes, and the significance of each process is uncertain.

In spite of the complexity and uncertainty, there appear to be common management actions that could promote successful restoration of multiple species. Perhaps the most obvious of these is the promotion of predator-free areas with multi-species barrier fences (Figures 2.3 and 2.4). The development of multi-species barrier fences on Guam, which would effectively exclude BTS and other non-native predators, is an ongoing technical challenge (Hileman et al., 2021). Ko'ko' and Såli are both vulnerable to BTS predation, which affects both survival and breeding success in these species. The recognized threat of BTS predation on native species implies the need to target control activities at reducing this threat for restoration to be feasible. Control of rats on Cocos Island (Pitt et al., 2012) and feral cats on Rota (U.S. Fish and Wildlife Service, 2009) appear to have contributed to Ko'ko' restoration in those locations, and predation of post-fledging Såli by feral cats recorded on Andersen Airforce Base (Pollock et al., 2019) suggests that non-native predators other than BTS are having effects on Såli. There is no existing evidence of predation on Ko'ko' or Såli by monitor lizards (Hilitai; *Varanus indicus*), although a potential link is identified in the influence diagrams. Historically, monitor lizards were believed to have depredated eggs and adults, but not at rates significant enough to negatively alter populations (Fritts and Rodda, 1998). Native coconut crabs (*Birgus latro*) are also capable of depredating large vertebrates (Laidre, 2017), but there is little evidence that they do. Habitat restoration actions, such as planting native trees and removing non-native plants, also appear to have multiple likely benefits, as do removal and exclusion of ungulates. Nafus et al. (2018) found a significant increase in tree abundance, even without seed dispersers present, in areas where ungulates were excluded. Exclusion of ungulates in concert with non-native plant removal and planting of native species could restore native forest and enhance restoration success for multiple native vertebrates on Guam.

We expect that models for other native vertebrates will share at least some similarities with the Såli and Koko models, e.g., predator and habitat interactions. However, to form a comprehensive community-based restoration strategy, additional species models could be constructed to better capture the effects of key threats not represented by the Ko'ko' and Såli models. Grouping species by life history similarities and developing models representative of groups may provide an efficient way to detect shared themes. Bird and bat species may be conceivably grouped by their dispersal distance, territory size, sensitivity to habitat quality, and vulnerability to non-native predators. The same method could be used for the three native reptile species in Table 2.1, where a single model could be built that captures major patterns for the group. Using this tactic would provide additional understanding of the common actions that are likely to benefit multiple species, while recognizing that individual species will have specific management needs.

Because particular management actions could promote restoration of several species, implementing these management actions as part of a comprehensive community restoration strategy may result in decreased total cost and resource use. Harnessing actions that simultaneously benefit multiple species has the potential to decrease the costs associated with establishing restoration sites for multiple species. Focusing on a strategic approach for restoring native species interactions and ecosystem function, e.g., through the temporal sequencing of reintroductions, can, theoretically, lead to long term success and more efficient use of management resources in the future. An effective community-based restoration plan can be realized through identifying shared influential management actions across multiple native species (U.S. Department of Navy and U.S. Fish and Wildlife Service, 2015).

While Guam's restoration problem presents unique challenges and dynamics, insights gained from other island restoration efforts are valuable. Take, for example, restoration of multiple native species in New Zealand, with its long list of species threatened by non-native predators and habitat modification (Bellingham et al., 2010). Populations of many native species were first established on predator-free offshore islands and these populations in turn provided a source for translocations to mainland reserves with predator fencing or intensive predator control (Burns et al., 2012; Saunders and Norton, 2001; Towns and Ballantine, 1993). The increasing number of mainland reserves have since restored native animals and vegetation communities (Simberloff, 2019). Habitat restoration is typically required, as is non-native species control in and around fenced areas to prevent reinvasion (Innes et al., 2019; Saunders and Norton, 2001). Although there have been numerous advances and conservation wins in New Zealand, ecosystem restoration goals have required, and will continue to require, a long-term institutional commitment to address challenges as they arise (Hare et al., 2019; Norton et al., 2016; Russell et al., 2015). The lessons learned through decades of restoration work in New Zealand, as well as other islands such as Mauritius (Florens, 2013; Jones and Merton, 2012) and California's Channel Islands (Morrison et al., 2014; Parkes et al., 2010) may be useful in informing Guam's restoration planning.

Developing a fully formed framework to inform restoration in Guam will require additional effort. Quantitative predictions of the probability of species restoration success will be needed to support the policy processes required to undertake translocations (i.e., the intentional movement of species for the purpose of restoration), and our existing conceptual models provide a jumping-off point for quantitative modeling. Deeper consideration of other objectives, particularly in the context of specific restoration efforts, will be necessary to address landowner objectives and mandates. Finer scale descriptions of management alternatives will also need to be developed for quantitative modeling, for instance defining release methods (e.g., soft or hard release), the number of individual animals to release, and the level of effort for BTS and other predator control. Identifying key uncertainties remaining after all available information is brought to bear provides the opportunity for adaptive management of reintroduction efforts (Armstrong et al., 2007; McCarthy et al., 2012).

Difficult conservation problems are characterized by complexity, which can hinder forward progress (Game et al., 2014). Structuring the Guam restoration problem in a collaborative setting with time for discussion and deliberation initiated the development of a shared understanding of the steps needed for progress toward restoration in Guam. The process facilitated communication between multiple agencies and stakeholders and laid the groundwork for future coordination. The focus of the Guam conservation community has, for multiple decades, been BTS research and management. Given the knowledge and technologies that now exist, the community's attention is likely to focus increasingly on native vertebrate restoration. In light of this shifting focus, a framework for conceptualizing restoration efforts will be necessary for success in the years ahead.

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2.8 Tables and Figures

Table 2.1 Strategy table outlining components of alternatives in the ‘how’ problem for vertebrate restoration in Guam. Potential actions are separated into categories, where one or more elements from each category can be selected to form alternative strategies. Both Chamorro and English names are given for species unless the Chamorro name is unknown.

Category	Bird Species		Mammal Species	Reptile Species	Release and Population Management	Fencing	BTS Control	Other Species Control (Invasive, (I), and Native, (N))	Habitat Restoration	Education and People Management
Actions	Såli (Micronesian Starling)	Nosa’ (Bridled White-Eye)	Fanihi (Mariana Fruit Bat)	Guali'ek halom tano' (Slevin's Skink)	Release from captivity	No fence/ landscape control	Trapping	None	None	None
	Sihek (Guam Micronesian Kingfisher)	Égigi (Micronesian Honey Eater)	Pacific Sheath-tailed Bat	Guali’ek (Micronesian Gecko)	Translocations from wild (Guam)	Multi-species barrier	Ground-based toxicant baiting	Ungulate control or eradication (I)	Plant native trees	Land management
	Ko’ko’ (Guam Rail)	Totot (Mariana Fruit Dove)		Guali'ek halom tano' (Oceanic Snake-eyed Skink)	Translocations from wild (CNMI)	Snake only barrier	Toxicant Baiting	Rodent control or eradication (I)	Non-native plant removal	Area closures
	Chuchurika (Rufous Fantail)	Yåyaguak (Mariana Swiftlet)			Facilitated dispersal	Ungulate fence	Aerial bait drops	Katu (Cat) control or eradication (I)	Artificial nest boxes	Public education about disturbance
	Åga (Mariana Crow)	Gå’ga’ Karisu (Nightingale Reed-Warbler)			Pre-release predator aversion training		Perimeter trapping	Ga'lågu (Dog) control or eradication (I)	Artificial nest substrate	Public education about illegal hunting
	Paluman Å’paka’/Fache’ (White Throated Ground Dove)	Sasangat (Micronesian Megapode)					Hand captures			Public education about conservation

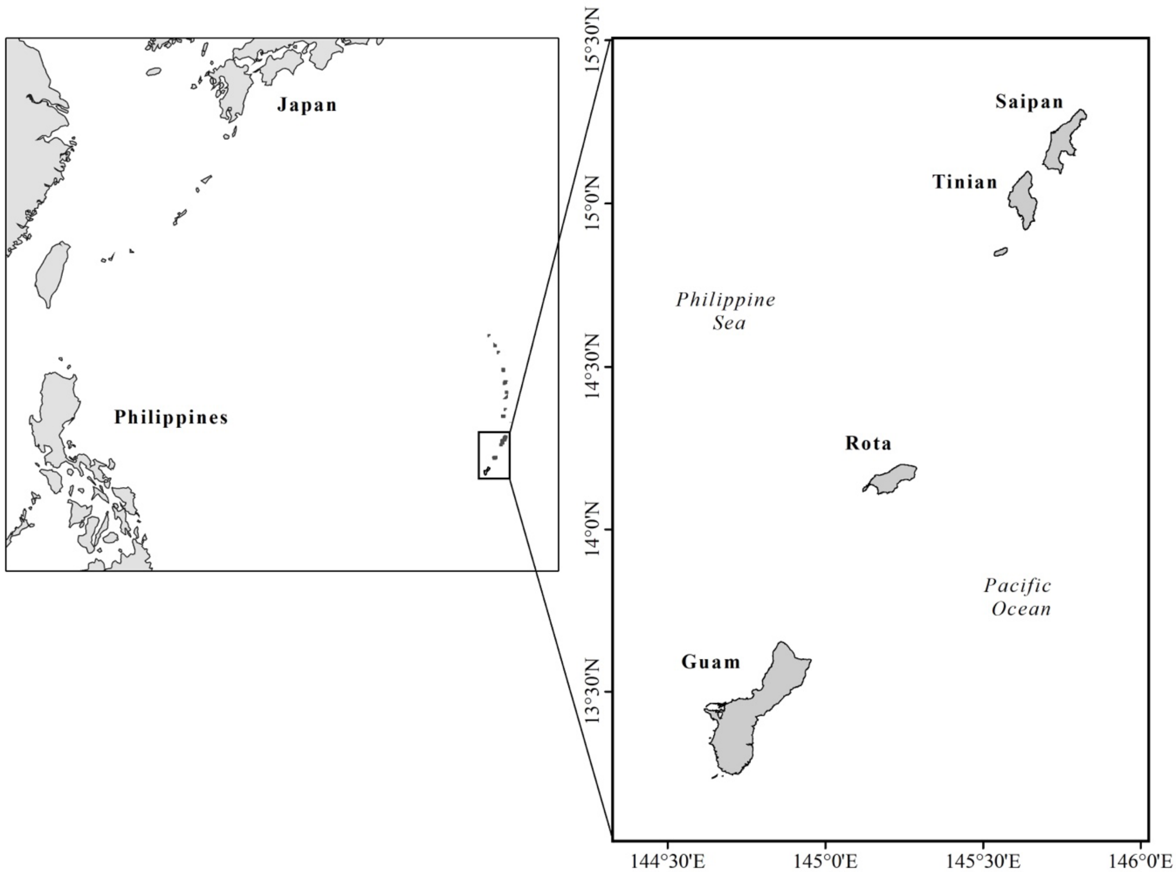

Figure 2.1 Map of the Mariana Islands. On the left, the Mariana Islands are shown east of the Philippines in the Pacific Ocean. The right shows a detailed view of the Mariana Islands, where Guam is shown in relation to Rota, Tinian, and Saipan.

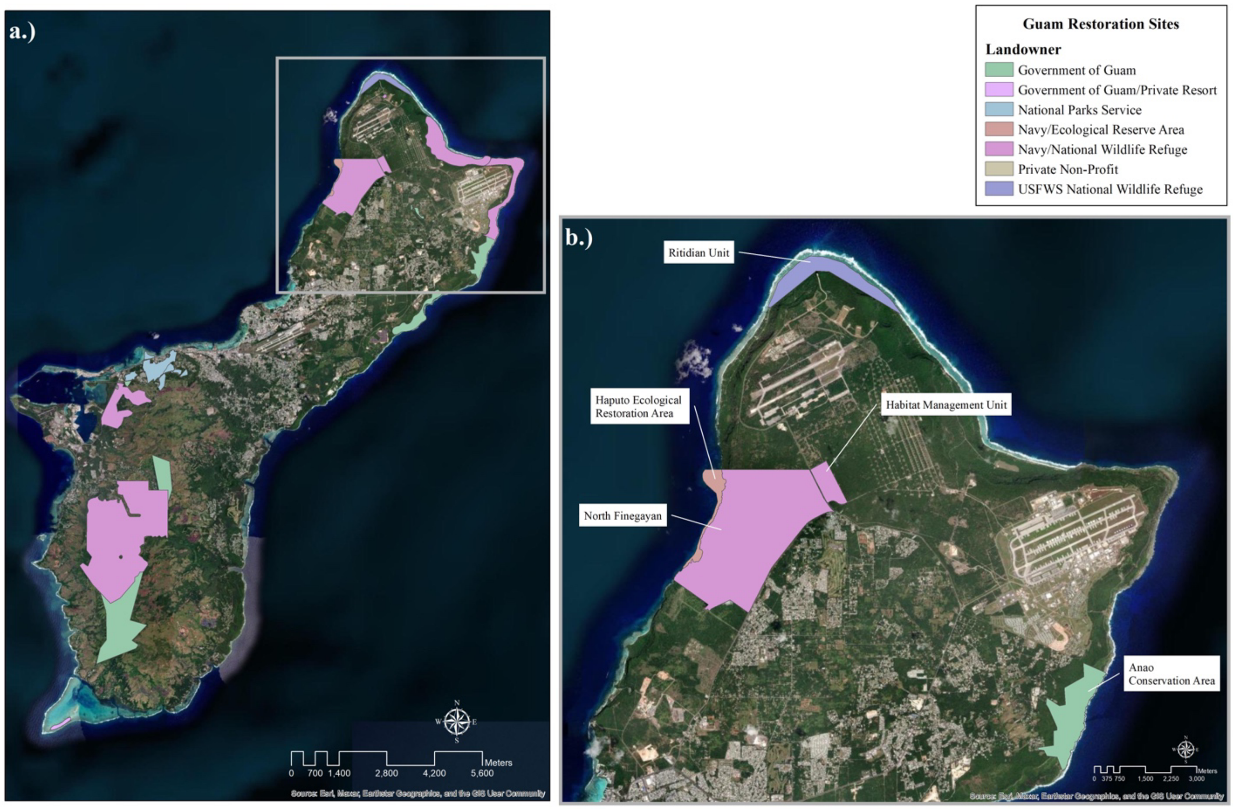

Figure 2.2 a.) Map of potential restoration sites in Guam. Sites constitute the alternatives in the analysis of the 'where' problem, i.e., where might restoration actions take place on Guam. Parcels are shown with colors corresponding to ownership. Sites include "first-tier" sites (conservation areas where management of invasive species and/or habitat was, at the time of this effort, either ongoing or planned for the near future, see specific first-tier sites identified in panel b) and "second-tier" sites (federal or local government lands allocated for conservation, but without ongoing or specifically planned invasive species or habitat management). b.) Map of potential first-tier sites for vertebrate restoration in Guam. First-tier sites were conservation areas where management of invasive species and/or habitat was either ongoing or planned for the near future at the time of this effort.

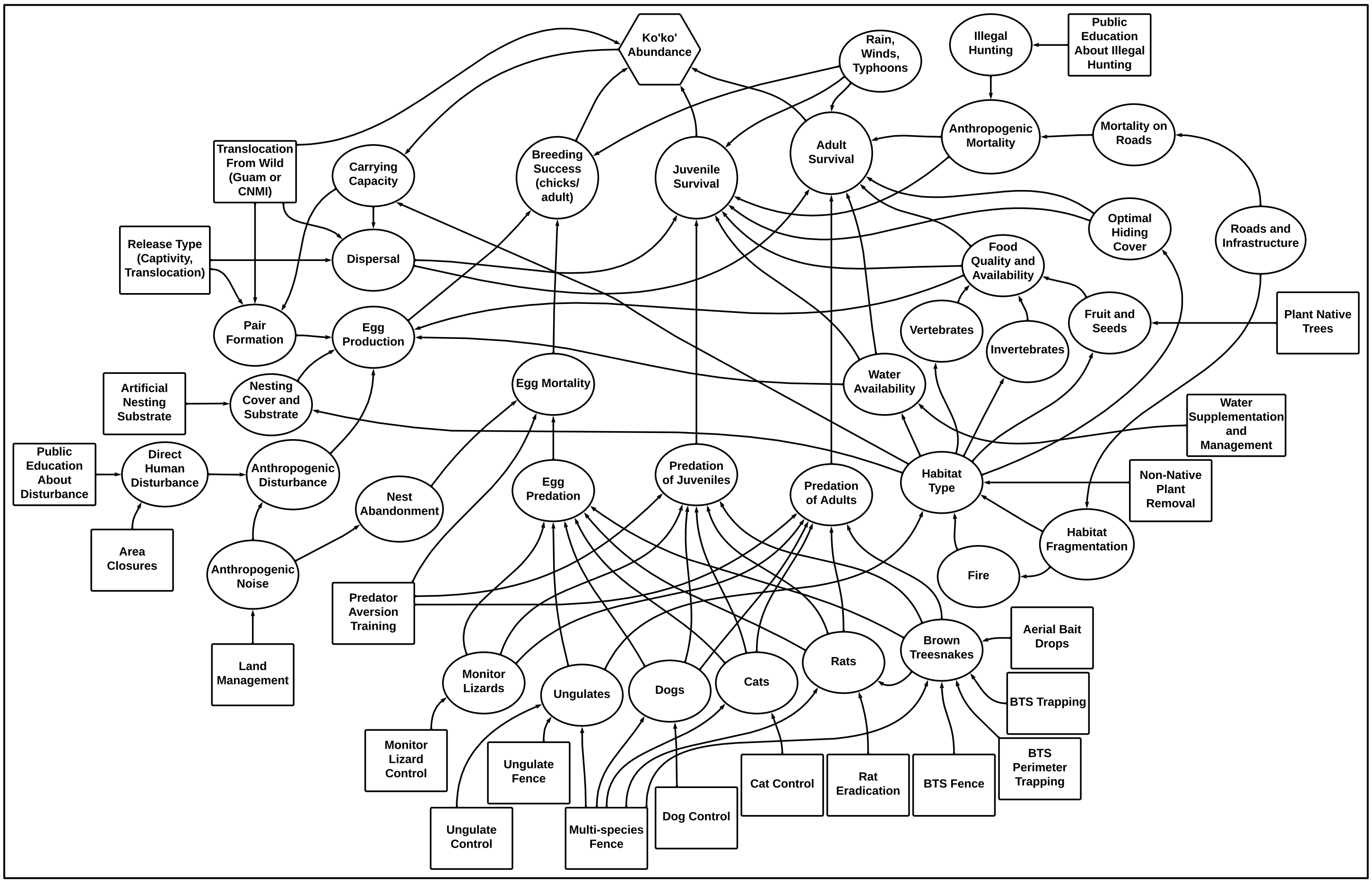

Figure 2.3 Full Ko'ko' influence diagram incorporating the species-specific management actions (boxes) from Table 2.1 that may be important to the objective of maximizing Ko'ko' abundance (hexagon, top center) via demographic processes (circles). Management actions indirectly influence the objective through various hypothesized processes and dynamics (ovals).

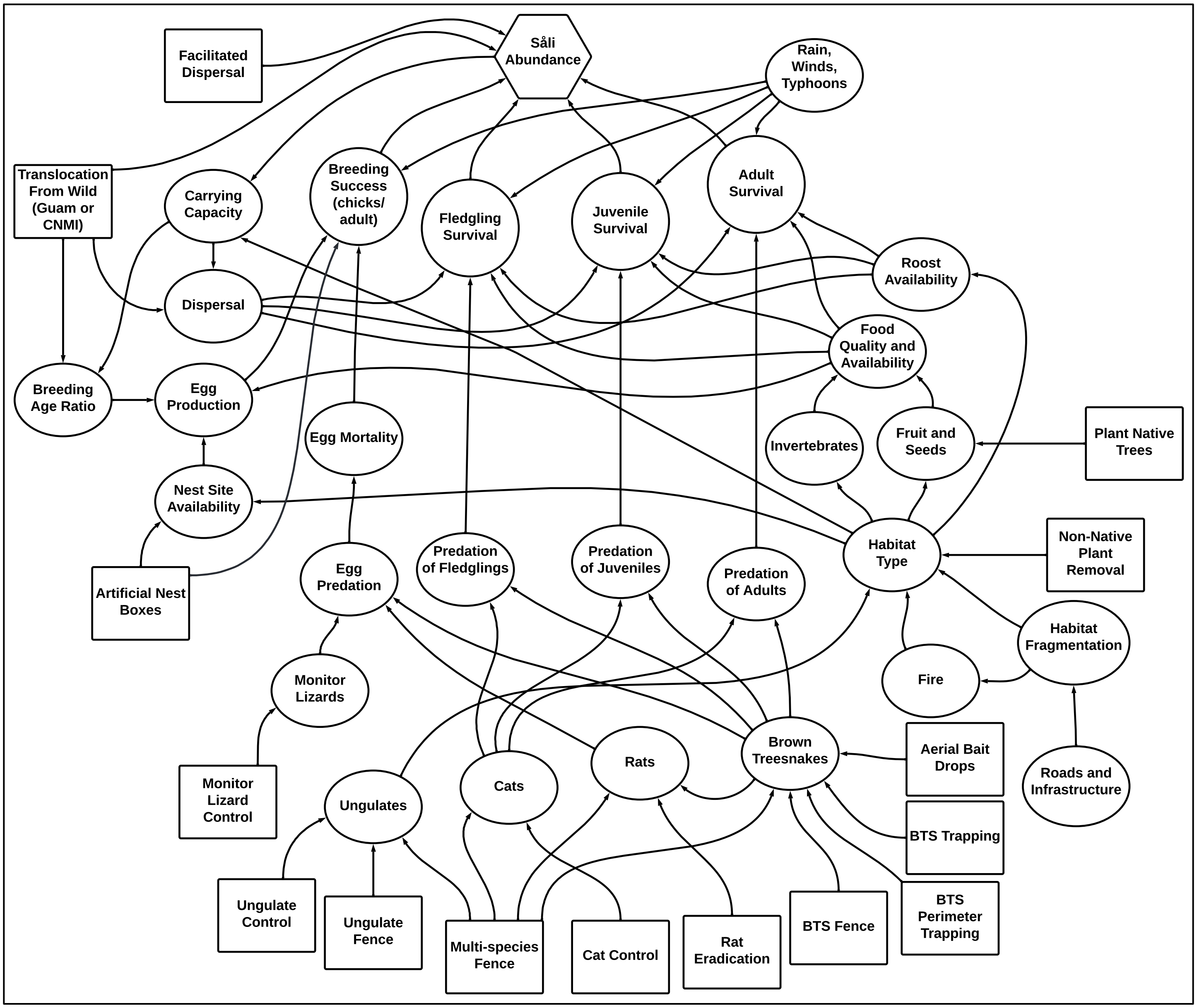

Figure 2.4 Full Såli influence diagram incorporating species-specific management actions (boxes) from Table 2.1 that may be important to the objective of maximizing Såli abundance (hexagon, top center) via demographic processes (circles). Management actions indirectly influence the objective through various hypothesized processes and dynamics (ovals).

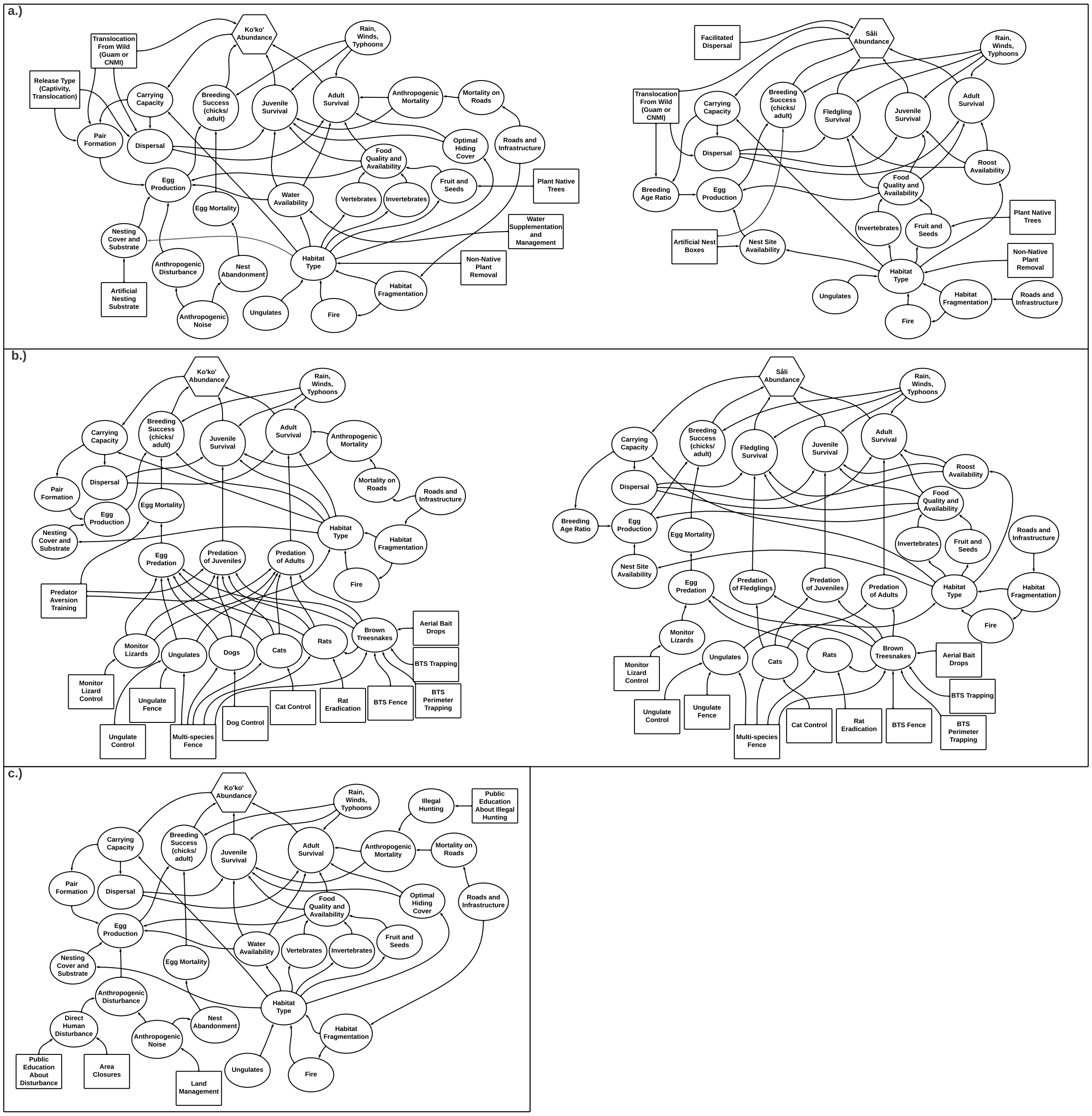

Figure 2.5 Ko'ko' and Såli influence diagram divided by categories. Each panel shows actions present in Table 2.1 by category for both species, including a) habitat restoration actions and release and population management actions, b) predator control actions, and c) education and human management actions (for Ko'ko' only). Boxes indicate management actions, ovals are the processes and dynamics impacting the objective of abundance (hexagon), and circles are demographic processes.

2.9 APPENDIX 2

Table A2 Roles, participants, and participant affiliations in the Guam vertebrate restoration working group. We have included individuals who are affiliated with organizations that have management authority over land or wildlife in Guam and the Northern Mariana Islands as "Decision Makers and Stakeholders" though many of these individuals are also subject matter experts.

Role	Participants	Affiliation
Decision Makers and Stakeholders	Dana Lujan	Marianas Conservation and Planning, NAVFAC Marianas, JRM[1]
	Marc Hall	BTS Program Manager, NAVFAC Marianas, JRM[1]
	Coralie Cobb	Senior Natural Resource Specialist, NAVFAC Pacific[1]
	Jennifer Horeg	Conservation Resources Program Manager, NAVFAC, JRM[1]
	Megan Laut	Wildlife Biologist, USFWS[2]
	Toni Mizurek	Fish and Wildlife Biologist, USFWS[2]
	Tammy Summers	Refuge Manager, Guam National Wildlife Refuge, USFWS[2]
	Laura Duenas	Wildlife Biologist, DAWR[3]
	Steve Mullin	Wildlife Section Supervisor, CNMI DWF[4]
	William Pitt	Deputy Director, SCBI[5]
Subject Matter Experts	Robert Reed	Branch Chief at the USGS Fort Collins Science Center, USGS[6]
	Haldre Rogers	Assistant Professor, Department of Ecology, Evolution, and Organismal Biology, Iowa State University[7]
	Doug Armstrong	Professor in Conservation Biology, School of Agriculture and Environment, Massey University[7]
	Nick Holmes	Lead Scientist, Island Conservation, TNC CA[7]
Decision Analysts and Facilitators	Sarah Converse	Unit Leader, Washington Cooperative Fish and Wildlife Research Unit, USGS[6] and Associate Professor, University of Washington
	Amy Yackel Adams	Research Ecologist, Fort Collins Science Center, USGS[6]
	Eben Paxton	Research Ecologist, Pacific Island Ecosystems Research Center, USGS[6]
	Hannah Sipe	Graduate Student, University of Washington

1) Joint Region Marianas (JRM) is the command that supports management functions for U.S. Department of Defense installations on Guam and military leased lands in the Northern Mariana Islands. They partner with USFWS[2], DAWR[3], and CNMI DFW[4] to conserve natural resources in the Mariana Islands through management projects on DoN administered lands (U.S. Department of Navy, 2019). JRM representatives offered knowledge of military operations and mandates on their administered lands. Representatives from JRM included representatives of Naval Facilities Engineering Systems Command (NAVFAC).
2) The U.S. Fish and Wildlife Service (USFWS) has management authority over species on Guam that are protected under the Endangered Species Act and the Migratory Bird Treaty Act. The USFWS also manages the National Wildlife Refuge on Guam.
3) The Guam Department of Aquatic and Wildlife Resources (DAWR) has management authority over wildlife on Guam and holds knowledge of native species ecology while managing several conservation breeding programs for native species.

4) The Commonwealth of the Northern Mariana Islands Department of Fish and Wildlife (CNMI DFW) holds key information about species ecology on the Northern Marianas Islands and serves as a potential source for reintroduction of native species to Guam.
5) The Smithsonian Conservation Biology Institute (SCBI) conducts and collaborates on conservation breeding of native Guam species.
6) The U.S. Geological Survey (USGS) serves as a major contributor to invasive and native species research occurring on Guam. Prior to completion of this report, Robert Reed became Supervisory Ecologist and Deputy Center Director of the USGS Pacific Island Ecosystems Research Center.
7) The subject matter experts were selected based on their expertise on conservation or management in Guam or in similar systems. Scientists from Iowa State University, Massey University in New Zealand, and The Nature Conservancy California (TNC CA) were included in the group. The authors of this report served as the decision analysts and facilitators for the working group (SJC, AYA, EHP, and HAS).

Chapter 3. Developing a Framework to Guide Vertebrate Restoration in Guam

Publication history: This study was co-authored with Amy A. Yackel Adams, Eben H. Paxton, and Sarah J. Converse. At the time this dissertation was published, this chapter was not in review with a journal.

3.1 Abstract

The decisions required to effectively implement broad-scale ecological community restoration can be overwhelmingly complex, especially when threats remain on the landscape. Decision-analytic techniques applied within collaborative working groups can guide development of practical frameworks to assist decision makers in effectively confronting this complexity. We undertook a decision-analytic process to guide long-term vertebrate restoration efforts in Guam, where much of the biodiversity has been lost in the wake of the introduction of brown treesnakes (*Boiga irregularis*). Our goal was to develop an initial prototype framework to assess reintroduction feasibility, inform future actions, and identify uncertainties by working collaboratively with management agencies, stakeholders, and scientific experts. Building off a problem framing process in which conceptual models for two of Guam's native birds, the Såli (*Aplonis opaca*) and the Ko'ko' (*Hypotaenidia owstoni*) were developed, we extended these conceptual models to build predictive population models. Due to limited available information about species response to management actions, we elicited expert judgements to parameterize predictive models and capture uncertainty in the projection of population trajectories. Predicted population persistence for both Såli and Ko'ko' was highest when invasive predator control was applied and habitat was restored. Uncertainty in predicted persistence was large for Såli under all strategies at the end of the timeframe, whereas uncertainty was lower for Ko'ko' under the best

performing alternative management strategy. Outcomes from this modeling effort will provide decision makers with an initial understanding of which types of management actions may be most effective for Såli and Ko'ko' restoration and can provide a framework for ongoing analysis of restoration efforts in Guam.

3.2 Introduction

Globally, habitat destruction and invasive species are major threats to species persistence (Clavero et al., 2009). For island endemic avian species, invasive species are one of the main drivers of extinction (Dueñas et al., 2021). One tool conservation managers can employ to combat species extinction is reintroduction. Reintroduction is a form of conservation translocation, wherein species extirpated from their historic range are restored through intentional movement and release of individuals (Seddon et al., 2012; Seddon and Armstrong, 2016). Although reintroduction best practices call for threatening forces to be removed prior to reintroduction (IUCN/SSC, 2013; Seddon and Armstrong, 2016), invasive species threats may not be feasibly eliminated in the near term (Armstrong et al., 2006; Hayward and Kerley, 2009; Moseby et al., 2019; Parlato and Armstrong, 2018) and delaying restoration actions may result in further species loss (Martin et al., 2012b). Hence, managers contemplating whether they should attempt reintroduction with invasive species threats on the landscape face difficult and complex decisions.

Decision analysis, or structured decision making (SDM), is a framework for grappling with complexity and improving overall decision quality (Gregory et al., 2012; Hemming et al., 2022; Keeney, 1982; Runge et al., 2020). All decisions consist of a common set of component parts and, through SDM, decisions are broken into these component parts through a deliberative process. The set of component parts are (1) a definition of the decision problem, (2) objectives, or what the decision maker wants to achieve, (3) alternatives, the alternative management actions that could be employed, (4) predictive models, to forecast the outcome of alternatives in terms of objectives, and (5) a method of identifying the best alternative to reach objectives (Gregory et al., 2012; Runge et al., 2020). SDM is especially useful for identifying and addressing barriers to decision making, aiding in the characterization of uncertainty, and providing decision makers with a transparent justification for the decision at hand (Gregory et al., 2012; Runge et al., 2020). Because reintroductions are inherently complex and uncertain, framing reintroductions using SDM is particularly beneficial (Converse et al., 2013). Further, SDM can be used to assess the feasibility of a reintroduction program (Keating et al., 2023) and due to the resource-intensive nature of avian translocations, offering a feasibility assessment may improve buy in of decision makers and stakeholders.

The predictive step in SDM is essential for informing reintroduction decisions. Quantitative demographic models offer a method for capturing how alternative management actions impact specific demographic processes (Converse et al., 2013; Converse and Armstrong, 2016). In addition to projecting how alternatives will perform, models are key for providing a framework for documenting the current understanding of the decision problem and a summary of the available information (Addison et al., 2013). However, predictions about reintroduction outcomes are fundamentally difficult due to uncertainty. The uncertainty results from limited information about species ecology, response to novel management actions, and the stochastic nature of ecological systems (Converse et al., 2013; Regan et al., 2002). Additionally, for species that do not currently coexist with threats that are present at potential release sites, there is uncertainty regarding the tolerance of the species to those threats (Parlato and Armstrong, 2018).

Expert elicitation can be used to overcome limited knowledge in reintroduction decision problems. Expert elicitation is a formalized approach for obtaining expert judgements in cases where direct empirical data are unavailable. Expert elicitation can provide critical missing information in many conservation decision contexts (Martin et al., 2012a; McBride et al., 2012). Through expert elicitation, expert knowledge about systems or species is obtained by asking experts to provide values based on their knowledge about the target system or a proxy system. Elicitation methods are designed to characterize uncertainty around the elicited information (MacMillan and Marshall, 2006; Speirs-Bridge et al., 2010). Expert elicitation offers a structured way to gain information that can be used to parameterize demographic models when data are unavailable.

The island of Guam (Guåhan in the native Chamorro language) has experienced dramatic biodiversity loss as a result of the accidental introduction of the brown treesnake (BTS, *Boiga irregularis*) post-World War II (Rodda and Savidge, 2007). Due to the introduction of this novel arboreal predator, most of the endemic avian species have been extirpated or driven to extinction (Savidge, 1987; Wiles et al., 2003). In addition to BTS, Guam has a suite of non-native mammalian predators (Fritts and Rodda, 1998). Eradication of BTS across moderately scaled landscapes, let alone island-wide eradication, has yet to be achieved, but advances in suppression methods show potential for local control or eradication in fenced areas (Clark et al., 2017; Engeman et al., 2018) offering opportunities for local restoration efforts. However, there is substantial uncertainty about whether reintroduction is feasible on Guam due to limited information about native species biology, uncertainty about native species response to management actions, and uncertainty about native species interactions with non-native predators. Consequently, any potential reintroduction actions involve substantial risk.

A selection of management agencies in Guam are interested in developing planning frameworks, assessing whether reintroduction is feasible, and understanding what management actions are necessary for supporting restoration goals. Through a collaborative process that included decision makers, stakeholders, and scientific experts, the initial decision components for this problem (i.e., definition of the decision problem, objectives, alternatives, and construction of conceptual models) have been developed (Chapter 2). However, the development of quantitative predictive models is necessary for informing the feasibility of reintroduction actions, identifying promising restoration actions, and identifying critical uncertainties. Predictive models will also be key for building agency and stakeholder support for Guam restoration efforts, given limited conservation dollars and risks associated with releasing threatened species into a BTS-occupied landscape.

Here, we build off the previously developed problem structure for Guam vertebrate restoration by developing predictive models linking alternative restoration strategies to species-level outcomes. We developed quantitative models based on an existing conceptual model structure (Chapter 2) for two Guam species, Ko'ko' (Guam Rail, *Gallirallus owstoni*) and Såli (Micronesian Starling, *Aplonis opaca*). Given that empirical data were unavailable to parameterize predictive models, we elicited expert judgements about demographic rates under various alternative management strategies at potential release sites. We developed stage-structured population models to project the probability of species persistence and investigated the importance of risk attitude to the decision. The results of this work provide the Guam conservation community with a framework for evaluating restoration actions at a site-specific level. Additionally, the models we developed serve as an initial framework that can be customized for specific decision makers and contexts and updated as more information becomes available.

3.3 Methods

3.3.1 Problem structure

A working group of decision makers, stakeholders, and experts previously defined a decision problem, identified objectives, and generated the elements in a strategy table that can be used to create alternative strategies for vertebrate restoration in Guam (Chapter 2). The group defined two linked decision problems: where in Guam to focus restoration efforts (the *where* problem) and how to select the most appropriate restoration actions at a specific location (the *how* problem). Through a rapid analysis of the *where* problem, the working group identified four sites that appeared to be promising for initial restoration efforts. They then shifted focus to the *how* problem, and identified objectives for the *how* problem, namely 1) maximize native vertebrate restoration, 2) maximize native forest restoration, 3) maximize ecosystem function, 4) maximize public access, 5) maximize learning, and 6) minimize cost and the use of other limited resources (e.g., staff time). In their analysis of the *how* problem, the working group organized and summarized how alternative management actions may influence species-level outcomes through the construction of conceptual models.

Here, we build on those conceptual models to create predictive population models. We focus on the objective of maximizing native vertebrate restoration, as we expect this to be the primary motivating objective for reintroductions, but we recognize that in future iterations of this work, additional objectives (i.e., cost or ecosystem function) could be explored further as the framework is customized for particular decision contexts. We used the structure of the conceptual models developed for Ko'ko' and Såli (see Figures 2.3 and 2.4) to develop quantitative models that predict the probability of species persistence under various alternative management strategies at the four sites identified in the previous analysis of the *how* problem.

3.3.2 Alternative strategy development

We focused on two native species, the Ko'ko' and the Såli, that have been impacted to varying degrees by BTS introduction. The Ko'ko' is a flightless, ground-nesting, generalist omnivore that was historically found across the island of Guam. Ko'ko' are endemic to Guam and are listed as endangered under the Endangered Species Act (U.S. Fish and Wildlife Service, 2020). Ko'ko' are found in captivity and in two wild populations (Cocos Island and Rota; U.S. Fish and Wildlife Service 1989, 2008), but have yet to be reestablished in Guam. Såli are generalists with a high reproductive capacity, with populations found throughout the Marianas Islands, including in human-dominated areas in Guam (Pollock et al., 2021).

Alternatives in this problem are known as strategy-based alternatives, which are alternatives that are composed of a selection of elements associated with multiple decision components (Converse and Grant, 2019). For example, a given strategy may have components associated with where to release species, what species to release, how to manage habitat, etc. A collection of actions associated with each of these components constitutes one alternative (Gregory et al., 2012). The strategy table developed in Chapter 2 (Table 2.1) was used to guide development of alternative management strategies for Såli and Ko'ko' by selecting elements from a subset of alternative components.

In our evaluation of strategies, we assumed that both Ko'ko' and Såli were already present at the release site, i.e., that each species survived the initial translocation phase and were no longer experiencing post-release effects (e.g., Panfylova et al. 2016, Armstrong et al. 2017).

This simplification allowed us to focus on long-term feasibility rather than the intricacies of particular translocation methods. However, there is ample evidence that animals experience a decrease in demographic rates immediately after translocation. Choosing translocation methods to minimize post-release effects will be important for achieving success in the early stages of translocation (Armstrong and Seddon, 2008).

We limited the scope of the alternative strategies to a selection of components related to threats that participants thought to be most limiting to restoration of avian species in Guam. For each strategy, we assumed there was a multi-species barrier in place (capable of excluding BTS, ungulates, and dogs, but not cats or rodents), dogs and ungulates were eradicated from the site, nest boxes or nesting substrate were provided if appropriate for the species in question, and the site was closed to public access. The components that we altered between strategies were the level of BTS control within the fence, the control of rats and cats, and habitat restoration. The set of strategies developed were intended to investigate which management actions are critical to species persistence.

In total we developed seven alternative strategies, each of which included some form of BTS control. The first three strategies were generated to assess species response to different levels of BTS control when mammals were controlled, and habitat was restored (‘BTS + Mammal + Habitat’ Strategies 1a – 1c). The next three strategies were designed to assess how species would respond to various levels of BTS control without mammal control and with habitat restoration (‘BTS + Habitat’ Strategies 2a – 2c). The last strategy controlled for all predators but did not restore habitat (‘BTS + Mammal’ Strategy 3). We generated the set of alternatives to provide an assessment of species response to predator control and habitat restoration in combination with one another, as well as independently. For example, the alternative strategies where BTS were eradicated evaluated how only mammal control or habitat restoration impacted species persistence. See Table 3.1 for a description of all strategies and information about predator control actions.

We evaluated the alternative strategies as applied to the four sites previously identified as the most promising initial release sites for restoration in Guam (Chapter 2). These sites included the Habitat Management Unit (HMU), the Guam National Wildlife Refuge land at Ritidian Point (Refuge), Anao, and North Finegayan. Sites vary in size and landownership, but each site contains native limestone habitat that is expected to be appropriate for a suite of avian species. The HMU is the most managed of the four sites, with a BTS-fence, multiple BTS control tools applied, and eradication of ungulates and dogs (Dorr et al., 2016; Siers et al., 2017). The Refuge does not currently have a BTS-fence, but managers have proposed construction of an invasive species barrier and subsequent invasive species control within the barrier (U.S. Fish and Wildlife Service, 2010). Anao was designated as an area for conservation by the Government of Guam and was identified as a promising restoration site due to its habitat characteristics, even though there is currently no ongoing predator control at the site (Guam Division of Aquatic and Wildlife Resources, 2006). Land within the North Finegayan site has undergone forest enhancement, ungulate removal, and has an ungulate barrier with plans for a multispecies barrier in the future (U.S. Department of Navy, 2019). See Chapter 2 for site descriptions and assessment.

3.3.3 Expert elicitation and analysis of expert judgements

Information about the impact of alternative management strategies is limited because of local extirpation of Ko'ko' and Såli from the four potential restoration sites and because many management actions have not previously been implemented. As such, to parameterize our predictive models, we elicited key demographic parameters under each alternative strategy at each site. We asked experts to provide stage-specific survival probabilities and fecundity parameters under each alternative management strategy at each site for Såli and Ko'ko'. Table 3.2 provides a description of each parameter that we elicited.

We used the 4-point elicitation method, which asks experts to provide a best judgment, minimum, and maximum value for a given parameter, along with their confidence level that the true value lay between the minimum and maximum values (Speirs-Bridge et al., 2010). The 4-point elicitation method allowed us to explicitly incorporate both within- and between-expert uncertainty in the final combined values.

Scientific experts were invited to participate in our elicitation process based on their subject matter expertise on Guam or Northern Marianas vertebrates, Guam habitat, and Guam invasive species (see Table A3 for list of experts and affiliation). We held an initial meeting to onboard the group of experts to the elicitation process, clarify and discuss assumptions about alternative management strategies, and work through a practice question. We provided experts with a document outlining instructions and information about the elicitation process, as well as relevant scientific literature and a summary document for both Såli and Ko'ko'. A Shiny application (Winston Chang et al., 2021) was built in R (R Core Team, 2022) to aid experts in visualizing their values in terms of a statistical distribution.

We used a modified Delphi process for elicitation (MacMillan and Marshall, 2006). Experts completed their first round of elicitation independently and offline using a fillable spreadsheet. After experts completed their first round of scores, we held a second remote meeting to discuss the results. During the discussion meeting of the first round of results, we found that the experts believed parameter values at the Refuge, Anao, and North Finegayan to be approximately the same. As such, the group decided that for the second round of elicitation, and subsequent predictive analysis, only the HMU and Refuge would be assessed. Following the discussion meeting, experts continued the discussion online by commenting on a shared document containing the compiled results from the first round of elicitation. Experts were then offered the chance to revise their first-round responses and the second-round values were taken as the final results. Group facilitation materials, e.g., instructions, practice question, and RShiny app code can be found on GitHub (linked below).

We used the 4-point expert judgements to estimate either beta or gamma distributions, as appropriate, by minimizing the sum of square errors at the expert-provided quantiles and modes. We chose the beta distribution to model the distribution of probabilities (survival and nest success probabilities) and the gamma distribution to model fecundity rates (number of nests per pair and number of young per nest). For each elicited quantity, we estimated shape parameters for a beta distribution or shape and rate parameters for a gamma distribution on the expert-provided confidence interval. We then converted these distributions to a 99% confidence interval, which resulted in a comparable distribution for each parameter and expert. We combined the individual distributions to create a mixture distribution for each parameter. The general form of the mixture distribution for either the gamma or beta distributions for N experts was defined as:

$$P(x|\theta) = \sum_{i}^{N} w_i * p(x_i|\theta_i) \tag{1}$$

where $p(x|\theta_i)$ is the density function (either beta or gamma), θ_i are the elicited parameters for expert i (α and β for a beta or gamma distribution), and w_i is the weight assigned to expert i. Note that we assigned equal weight across experts: $\sum_i^N w_i = 1$ and $w_i = \frac{1}{N}$, but this need not be the case in other applications (Hemming et al., 2020).

By combining expert distributions in this manner, we retained each expert's individual judgements and were able to reflect the combined judgements in simulations, capturing the full range of parametric uncertainty within and across experts. The nest success, nests per pair, and fledgling or hatchling per nest parameter values were multiplied together to give fecundity, f, which is the number of fledglings (for Såli – an altricial species) or hatchlings (for Ko'ko' – a precocial species) produced per pair in a year. We then used the mixture distributions to parameterize Såli and Ko'ko' demographic predictive models.

3.3.4 Predictive models

We developed predictive demographic models for Såli and Ko'ko' to evaluate the impact of alternative management strategies on probability of species persistence. For these demographic models, we assumed that immigration does not occur, and we assumed emigration was equivalent to mortality. A life history diagram for each species is provided in Figure 3.1, the basis for which was the conceptual models developed in Chapter 2 (Figures 2.3 and 2.4). For both species we include demographic stochasticity, as this form of stochasticity is particularly important for small populations (Lande, 1988). Parametric uncertainty was incorporated following the guidelines outlined in McGowan et al. (2011).

The basic stage-structured projection model for Såli included two life stages: juveniles (aged 1 year), and adults (aged 2 years or older; Figure 3.1). We modeled the stage-specific abundance of Såli adults in year t as:

$$N_{ad,t} \sim Binomial\left(N_{ad,t-1} + N_{j,t-1}, \phi_{ad}\right) \tag{2}$$

where the number of adult Såli in the current year depends on the number of adult and juvenile Såli, $N_{ad,t-1} + N_{j,t-1}$, that survived the previous year with adult survival probability, ϕ_{ad}. The number of Såli juveniles produced in year t was modeled as:

$$N_{j,t} \sim Poisson\left(N_{ad,t-1} * f * \phi_{fl} * \phi_j\right) \tag{3}$$

where the number of juveniles is a function of the number of fledglings produced by adults, i.e., $N_{ad,t-1}$* f, that survive their first month with probability ϕ_{fl} and subsequently survive from one month to one year of age with probability ϕ_j.

For Ko'ko', the basic stage-structured projection model also included two life stages: juveniles and adults (Figure 3.1). Adult Ko'ko' were modeled similarly to adult Såli (Eq 2) and we modeled the abundance of juvenile Ko'ko' in year t as:

$$N_{j,t} \sim Poisson\left(N_{ad,t-1} * f * \phi_j\right) \tag{5}$$

where the number of juveniles, $N_{j,t}$, is a function of the number of hatchlings produced in year t, $N_{ad,t} * f$, that survive to 1-year of age with probability ϕ_j.

We included density dependence in each species projection model by reducing f to 0 if the number of adults in the population was greater than or equal to the carrying capacity for the site being modeled. We approximated carrying capacity using reported information on abundance and territory size for both species. Pollock et al. (2021) estimated an abundance of 200 Såli on Cocos Island, a small island south of Guam. We used this abundance to calculate the number of Såli per ha by dividing the Såli abundance on Cocos Island by its area (38.63 ha), resulting in a density of 5.177/ha and an estimated carrying capacity of 285 at the HMU and 811 at the Refuge. There are reports of Ko'ko' territory size on Rota and from a release attempt on Guam of approximately 2 ha (Beauprez and Brock, 1999). We extrapolated these territory sizes to the HMU and Refuge, resulting in a carrying capacity for Ko'ko' of 110 for the HMU and 312 for the Refuge. We recognize finer precision in carrying capacity estimates may be required in the future, either by eliciting values or monitoring populations to gain estimates of carrying capacity.

3.3.5 Simulations

For each iteration in our simulations, we generated samples from the elicited mixture distributions by first drawing from a categorical distribution with a probability vector containing the weights for each expert, i.e., $Categorial([w_1, \dots, w_N])$, with each $w_i = \frac{1}{N}$, where $i \in 1, \dots, N$, for N total experts. The outcome from the categorical draw determined which expert's beta or gamma distribution was used for a given demographic parameter in each iteration of the predictive model. That is, when the random categorical draw for Såli adult survival, ϕ_{ad}, was for expert i, we used expert i's beta parameters to draw a random probability from a beta distribution, i.e., $\phi_{ad} \sim Beta(\alpha_i, \beta_i)$. In each iteration, a collection of parameters, describing sampling uncertainty both within and across experts, was used to project the population forward.

We ran 400,000 simulations, thereby capturing parametric uncertainty in the elicited mixture distributions. Using a collection of elicited parameters values, we projected the population forward for 21 years in each simulation iteration and drew 50 samples for each set of parameter values per year, thereby accounting for demographic stochasticity. We simulated the seven alternative management strategies for Såli and Ko'ko' at the HMU and Refuge. Further, we repeated the simulations at four initial adult abundances (10, 20, 30, or 40, with equal number of males and females for each). We ran the simulations both with density dependence and without density dependence (i.e., carrying capacity). Models were built and simulations were run in R (R Core Team, 2022). Simulation code and code to build expert- and parameter-specific distributions can be found on GitHub (https://github.com/sipeha/Dissertation_Chapter_3).

3.3.6 Assessment of projections and risk analysis

The model outputs were analyzed for each alternative, species, site, and initial starting abundance by visualizing the mean probability of persistence and the 95% quantile around the mean over 21 years, i.e., giving the cumulative probability of persistence over time. The probability of persistence was calculated by taking the number of samples where the abundance of adults was greater than 0 in a year and dividing that by the total number of samples. We also made density plots of the probability of persistence in the last year for each simulation.

We tested the sensitivity of the probability of persistence to carrying capacity by running simulations with and without carrying capacity included. Sensitivity to the inclusion of carrying capacity was compared by taking the difference between equivalent alternative strategy, site, species, and initial starting abundance simulations with and without density dependence.

The probability of persistence predictions in the last year, i.e., year 21, were compared across strategies and sites for both species by plotting the cumulative density distribution and assessing stochastic dominance. Stochastic dominance provides information about how important risk attitudes are to the decision and can be used to rank alternative strategies through the comparison of cumulative probability distributions. Stochastic dominance offers information about risk attitudes but does not require explicit elicitation of utility, thereby offering an initial analysis of risk (Canessa et al., 2016; McCarthy, 2014).

First-order stochastic dominance was assessed by comparing the cumulative distribution functions (CDF) of the probability of persistence under each alternative strategy in the last year of the simulation. The CDF for probability of persistence represents the probability that the cumulative distribution of persistence is less than or equal to any value x. In mathematical terms, for each value x on the interval from 0 to 1, the CDF is defined as:

$$F(x) = P(X \leq x) \tag{6}$$

where $F(x)$ represents the CDF evaluated at any point x and $P(X \leq x)$ is the probability that the probability of persistence is less than or equal to x. An alternative strategy is first-order stochastic dominant when it has a smaller or equal CDF for any value x than all other alternatives, i.e., the CDF for the first-order stochastic dominant strategy is always less than or equal to all other CDFs, provided the CDFs do not intersect one another (Canessa et al., 2016; McCarthy, 2014). The CDF that is less than all others indicates that the given density at each point x is smaller and the probability of persistence is greater than x. We used numerical integration to derive the probability of persistence CDFs for each alternative strategy (code found on GitHub).

3.4 RESULTS

The probability of persistence was not found to be sensitive to the inclusion of carrying capacity. There was minimal difference in probability of persistence between simulations with and without carrying capacity (the maximum difference was less than 0.04) and the ordering of alternative strategies remained the same regardless of carrying capacity. Here, we report only on the simulations that included density dependence.

The probability of persistence for both species was higher under strategies where predation threats were minimized. The strategy that resulted in the highest mean probability of persistence across the 21-year timeframe for both species was the BTS + Mammal + Habitat

Strategy 1a. The BTS + Mammal Strategy 3 was the second best in terms of mean probability of persistence for Såli and Ko'ko'. The strategy representing the least amount of predator control, BTS + Habitat Strategy 2b, resulted in predictions with the lowest overall probability of persistence for both species. On average, strategies with complete predator control had 40% greater probability of persistence in the final year than strategies with minimal predator control. Strategies at the Refuge generally resulted in higher persistence than at the HMU and higher initial adult abundance produced higher probability of persistence (Figure 3.2).

The uncertainty around the mean probability of persistence for Såli across all years in the timeframe was large, with 95% quantiles reaching from 0 to 1 by year 21 (Figure 3.3). The uncertainty around the mean probability of persistence for Ko'ko' was also large for poorer performing strategies (i.e., BTS + Habitat Strategy 2b), but for the two top alternative strategies (BTS + Mammal + Habitat Strategy 1a and BTS + Mammal Strategy 3), 95% quantiles did not include 0 (Figure 3.4).

The BTS + Mammal + Habitat Strategy 1a resulted in a mean probability of 0.84 for Såli and 0.96 for Ko'ko' in year 21. For many of the alternatives, probability of persistence distributions were bimodal in year 21, where distributions showed large densities for values near 0 or 1, with limited density in the middle (Figures 3.5 and 3.6). For Såli, the BTS + Mammal + Habitat Strategy 1b and BTS + Habitat Strategies 2b and 2c resulted in a probability of persistence density near 0 in year 21. All other strategies for Såli resulted in bimodal distributions (Figure 3.5). For Ko'ko', BTS + Mammal + Habitat Strategy 1a and BTS + Mammal Strategy 3 had a probability of persistence near 1 in year 21. Except for BTS + Mammal + Habitat Strategy 1a and BTS + Mammal Strategy 3, persistence distributions for

Ko'ko' were also bimodal and the distributions became less bimodal, for some alternatives, with increasing initial adult abundance (Figure 3.6).

None of the CDFs intersected each other, so we were able to identify strategies that were first order stochastic dominant in every case. The probability of persistence CDF for BTS + Mammal + Habitat Strategy 1a was less than all other strategies, across initial adult abundances and at both sites. Therefore, the BTS + Mammal + Habitat Strategy 1a was first-order stochastic dominant to all other alternatives (Figure 3.7).

3.5 Discussion

Here, we present a predictive modeling framework for assessing species restoration efforts under various alternative management strategies at restoration sites in Guam. The predictive models we developed were parameterized with expert judgements, serving as a synthesis of available information and initial model structure that can be updated as new data become available, i.e., through Bayesian updating (Ellison, 2004). Moreover, the initial projections offer the opportunity to assess whether reintroduction attempts are feasible and under what management conditions the probability of success is greatest. Given limited available resources and potential risk of poor outcomes, predictive models will be key to building support for restoration efforts in the future. Further, this initial assessment offers insight into what additional work is required to inform decisions about restoration.

The top performing alternative resulted in a mean persistence probability of 0.84 for Såli and 0.96 for Ko'ko' in year 21 (Figures 3.5 and 3.6). Species persistence for Såli and Ko'ko' was predicted to be highest under the same alternative strategy, i.e., BTS + Mammal + Habitat Strategy 1a. Såli persistence was higher at the Refuge than HMU and higher when habitat restoration actions were applied. Ko'ko' were less sensitive to the choice of site and habitat restoration actions. Although the top alternative was similar for both species, the ranking of intermediate strategies was not (Figure 3.2). Såli persistence was more sensitive to BTS presence, as strategies that ranked higher included more BTS control efforts regardless of mammal control. Low post-fledgling survival of Såli in Guam on Anderson Airforce Base has been shown to be caused by BTS predation and feral cats have been observed depredating post-fledgling Såli (Pollock et al., 2019). However, less is known about how adult Såli are impacted by non-native predators. Ko'ko' have benefitted from control of rats and feral cats in their two wild populations of Ko'ko' (Pitt et al., 2012; U.S. Fish and Wildlife Service, 2009), but less is known about BTS threats. Yet, Ko'ko' were predicted to have a higher tolerance to BTS presence when mammalian predators are controlled for, which may be related to their large body size and territorial nature.

Unsurprisingly, the alternative strategy that performed the best for both species was also the alternative that is most cost and resource intensive (i.e., BTS + Mammal + Habitat Strategy 1a at the Refuge with an initial adult abundance of 40 adults). In Guam, achieving local eradication of BTS in moderately sized fenced areas has not yet been accomplished, although it is likely possible with years of intensive control efforts (Nafus et al., 2022). Control of mammalian predators and habitat restoration are less technically challenging but would, nonetheless, require a large investment of resources (Nogales et al., 2004; Russell and Holmes, 2015). Our results provide some encouraging insight, as Ko'ko' were predicted to have tolerance to BTS when mammals are controlled for (Figure 3.6). However, including finer-scale detail about the level of snake presence under various BTS control measures would help identify strategies that maximize persistence using fewer resources. Additionally, analyzing costs alongside the species persistence objective would provide decision makers with insight about the feasibility of restoration efforts and allow for explicit consideration of tradeoffs between persistence and cost (Converse, 2020; Edwards et al., 2022; Ferrière et al., 2020; Regan et al., 2023).

We formally elicited expert knowledge using a structured and transparent process that allows for elicited information to be confronted with empirical information when available (Martin et al., 2012a). In doing so, expert elicitation provided an opportunity to develop initial predictions about restoration actions in Guam. Since we kept the alternative strategies broad, we spent time with experts to discuss assumptions and any misunderstanding prior to eliciting values (Hanea et al., 2017; Hemming et al., 2018). We also supplied experts with the relevant literature about demographic rates for both species, although only a small number of documented reports were available. We used the 4-point elicitation method where experts supplied their own confidence level that a parameter's true value was between their given minimum and maximum (Speirs-Bridge et al., 2010), allowing for an expert's values to be fit to a statistical distribution. We supplied experts with an application to visualize their distributions. Offering this tool to experts helped minimize the cognitive task by providing a translation between the values and the distribution. Further, our use of the mixture distribution allowed us to include the full range of uncertainty elicited by experts in our projections.

To further confront uncertainty in this decision, we assessed the alternative strategies and associated uncertainty using stochastic dominance to identify if risk attitudes could impact the decision choice (Canessa et al., 2016; McCarthy, 2014; Yemshanov et al., 2013). The BTS + Mammal + Habitat Strategy 1a was first order stochastically dominant to all other strategies for both species, implying that, at this stage and with these results, including decision maker's risk attitude would not change the preferred alternative. Stochastic dominance is a transparent approach to considering alternative outcomes as it uses the whole predicted distribution instead of a summary statistic (Canessa et al., 2016). For the purpose of this initial framework, stochastic dominance offered a way to identify which alternative is the most promising given uncertainty.

According to our analysis, restoration of Såli and Ko'ko' to fenced sites appears possible under certain management conditions. The alternative strategy that controlled for predators and restored habitat led to probability distributions for both species with minimal density near 0 in year 21 (Figures 3.5 and 3.6). In other words, there is a low probability that a population would go extinct within 20 years under the top performing alternative. The predicted outcomes from the other alternatives resulted in probability distributions with high density around 0 or bimodal probability distributions, where there was density around very high and very low probabilities of persistence but minimal density between those extremes. In both these cases, there is a reasonably high probability that a population would go extinct within 20 years. A key question for managers to consider next is whether alternatives resulting in relatively low probability of extinction are achievable in Guam.

The framework we developed here can be seen as an initial prototype useful for supplying information to fuel discussion of next steps regarding reintroducing vertebrates to Guam. Because of this, we kept the components relatively simple, following the rapid prototyping approach (Garrard et al., 2017). One potentially important consideration for the next iteration of this work would be to consider various release strategies for translocating species and including post-release effects in the predictive model. Post-release effects have been shown to be critical to include when planning for a reintroduction, since the act of translocation can result in lower survival and fecundity in the short term (Armstrong and Reynolds, 2012; Armstrong and

Seddon, 2008). Incorporating release strategies and post-release management into predictive models offers a chance to identify which actions will result in persistence through the initial translocation stage (Armstrong et al., 2017). Therefore, one of the next steps for species restoration efforts in Guam would be to either elicit expert judgements about species responses to release strategies or to conduct trial releases and use that information directly to model outcomes.

Additional species could be integrated into future iterations of this framework. Including additional species would help to further identify management actions that are beneficial to multiple species. Explicitly accounting for interspecific interactions between species would then be necessary to determine what timing and sequence of reintroductions is likely to promote overall restoration success (Geary et al., 2020; Plein et al., 2016). Here, we focused on modeling individual persistence for two species without incorporating any ecological interactions between them. While there is no evidence to suggest that Såli and Ko'ko' have interspecific interactions with each other that would reduce persistence of either species, this assumption could be confronted in future work or when additional species are considered. For instance, Peterson et al. (2021) developed a decision-analytic framework for assessing multispecies reintroductions in Australia, using expert elicitation to inform the structure of species interactions. They evaluated potential reintroduction strategies that involved the sequence of species releases, the time between releases, and the specific release location in the assessment (Peterson et al., 2021). Should decision makers choose to move forward with vertebrate restoration in Guam, undergoing a similar assessment could provide valuable insight.

This work demonstrates what can be accomplished prior to making a decision about whether to reintroduce species or prior to carrying out any conservation actions for a problem that lacks data and has considerable uncertainty. Throughout this process, we showed the progress that can be made to inform future decisions by developing quantitative models even when little empirical information is available to parameterize models. The predictions from our models present an opportunity to understand general themes about multi-species restoration in a landscape where threatening forces are present and to identify management actions that may benefit multiple species. Further, the predictions show which management actions relate to higher species persistence and with this information, managers could take steps toward achieving eradication of BTS and mammal control, if they decide to move forward with reintroduction. Overall, this work provides a customizable framework that can be adapted to fit the needs of specific decision makers in determining if reintroduction should take place and if so, an initial framework for assessing strategies for conducting reintroduction of multiple species. Although Guam is an extremely complex example of species restoration, the methods applied here are relevant to many other conservation problems.

3.6 Acknowledgements

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3.8 TABLES AND FIGURES

Table 3.1. Alternative management strategies for Ko'ko' and Såli reintroduction to restoration sites in Guam. Alternative strategies were generated from an alternative strategy table (Table 2.1, Chapter 2). For these strategies, we assumed that Såli or Ko'ko' were already present at the site, i.e., species were successfully established. For each strategy, we assumed that (1) there was a snake and ungulate barrier maintained at the site, (2) dogs and ungulates were eradicated from the site, (3) nest boxes for Såli or nesting substrate for Ko'ko' were provided, and (4) the site was closed to public access. Experts were asked to provide parameter values for each species at all the potential release sites (HMU, Refuge, North Finegayan, Anao) under each of the strategies provided in the table.

	Strategy	BTS control	Rodent control	Cat control	Habitat restoration
BTS + Mammal + Habitat	Strategy 1a	Eradication[1]	Ongoing rodent control[4]	Ongoing cat control[5]	Remove non-natives and plant natives
	Strategy 1b	Ongoing aerial bait drops[2]	Ongoing rodent control[4]	Ongoing cat control[5]	Remove non-natives and plant natives
	Strategy 1c	Ongoing aerial bait drops and supplemental captures[3]	Ongoing rodent control[4]	Ongoing cat control[5]	Remove non-natives and plant natives
BTS + Habitat	Strategy 2a	Eradication[1]	No rodent control	Targeted cat removal[6]	Remove non-natives and plant natives
	Strategy 2b	Ongoing aerial bait drops[2]	No rodent control	Targeted cat removal[6]	Remove non-natives and plant natives
	Strategy 2c	Ongoing aerial bait drops and supplemental captures[3]	No rodent control	Targeted cat removal[6]	Remove non-natives and plant natives
BTS + Mammal	Strategy 3	Eradication[1]	Ongoing rodent control	Ongoing cat control	None

1.) Eradication – removal of all BTS from a fenced site.
2.) Ongoing aerial bait drops – aerial bait drops involve distributing dead mice laced with acetaminophen, a toxicant to BTS, into the tree canopy by way of helicopter (Dorr et al., 2016). Aerial baits are efficient for treatment of large landscapes (Siers et al., 2020), but may not remove all sizes of BTS (Nafus et al., 2022).
3.) Supplemental captures – visual search and hand captures or passive trapping. Captures are labor intensive, but can result in removal of all size classes of BTS (Christy et al., 2010).
4.) Ongoing rodent control – aerial delivery of rodenticide, supplemental ground baits.
5.) Ongoing cat control – deployment of cat traps throughout site.
6.) Targeted cat removal – targeted cat trapping when a cat predation is detected.

Table 3.2. Table of elicited parameters for Såli and Ko'ko' with definitions of each parameter. Note that fecundity, f, is the product of nest success, fledglings or hatchlings per nest, and the number of nest attempts.

Species	Parameter	Parameter definition
Såli	Adult survival, ϕ_{ad}	Annual survival probability for birds that are 1 year or older
	Juvenile survival, ϕ_j	Survival probability of birds from 1 month post-fledge to 1 year of age, i.e., over an 11-month period)
	Fledgling survival, ϕ_{fl}	Survival probability of birds that are fledglings, i.e., just left the nest, to 1 month post-fledge
	Nest success	Probability that a nest produces one or more fledglings
	Fledglings per nest	The number of fledglings produced per successful nest
	Nest attempts	The number of nesting attempts per adult per year
Ko'ko'	Adult survival, ϕ_{ad}	Annual survival probability for birds that are 1 year or older
	Juvenile survival, ϕ_j	Survival probability of birds from hatching to 1 year of age, i.e., over a 12-month period
	Nest success	Probability that a nest produces one or more hatchlings
	Hatchlings per nest	The number of hatchlings produced per successful nest
	Nest attempts	The number of nesting attempts per adult per year

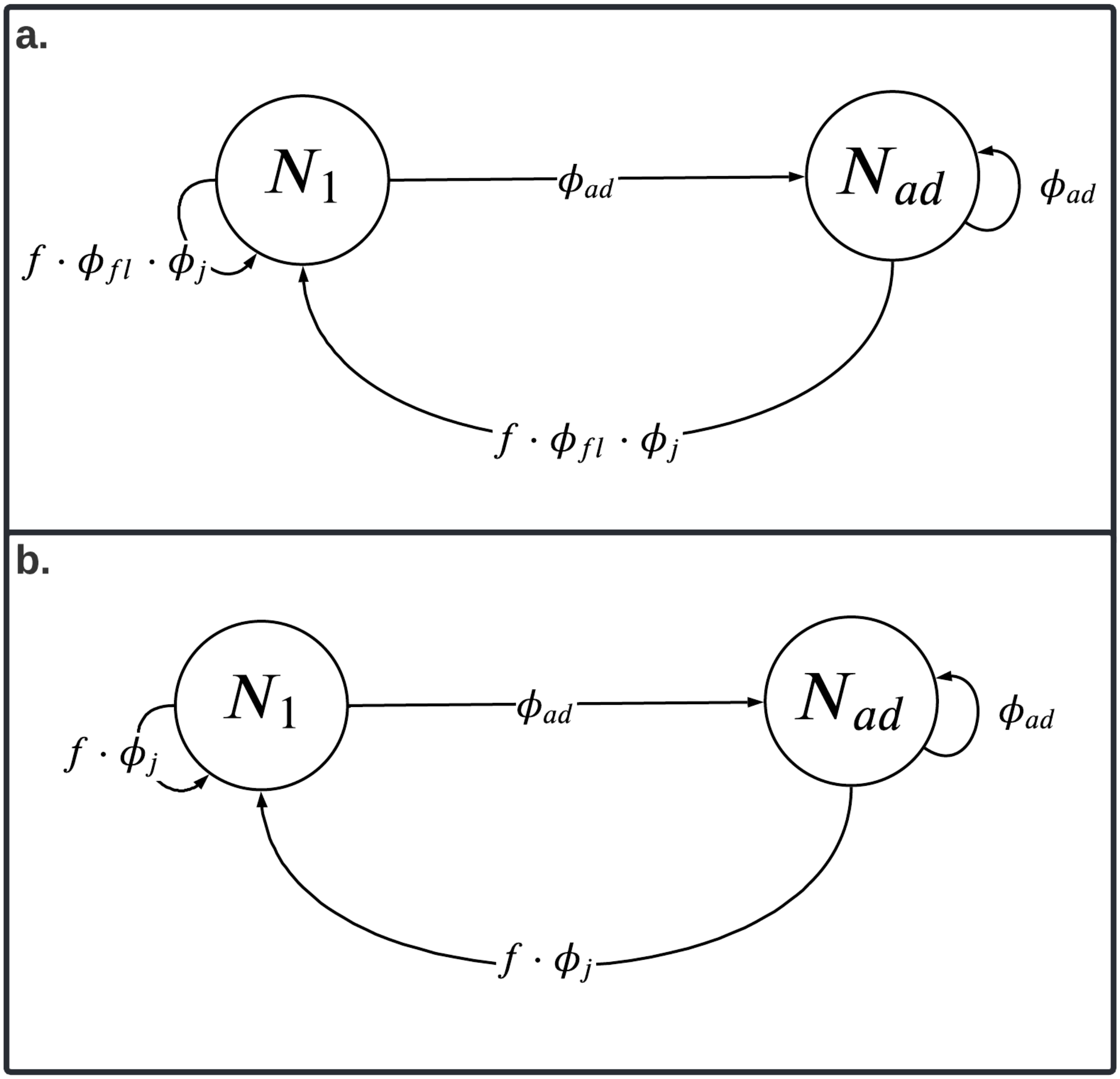

Figure 3.1. Basic life-cycle diagram for Såli and Ko'ko'. Age classes shown for both species are one-year-olds, N_1, and adults, N_{ad}. (a) Both Såli age classes produce offspring with fecundity rate f and offspring survive the fledgling stage with probability ϕ_{fl} and the juvenile stage with probability ϕ_j in order to become one-year-olds. Both age classes survive with probability ϕ_{ad}. (b) Ko'ko' are modeled similarly to Såli, but without a fledgling stage.

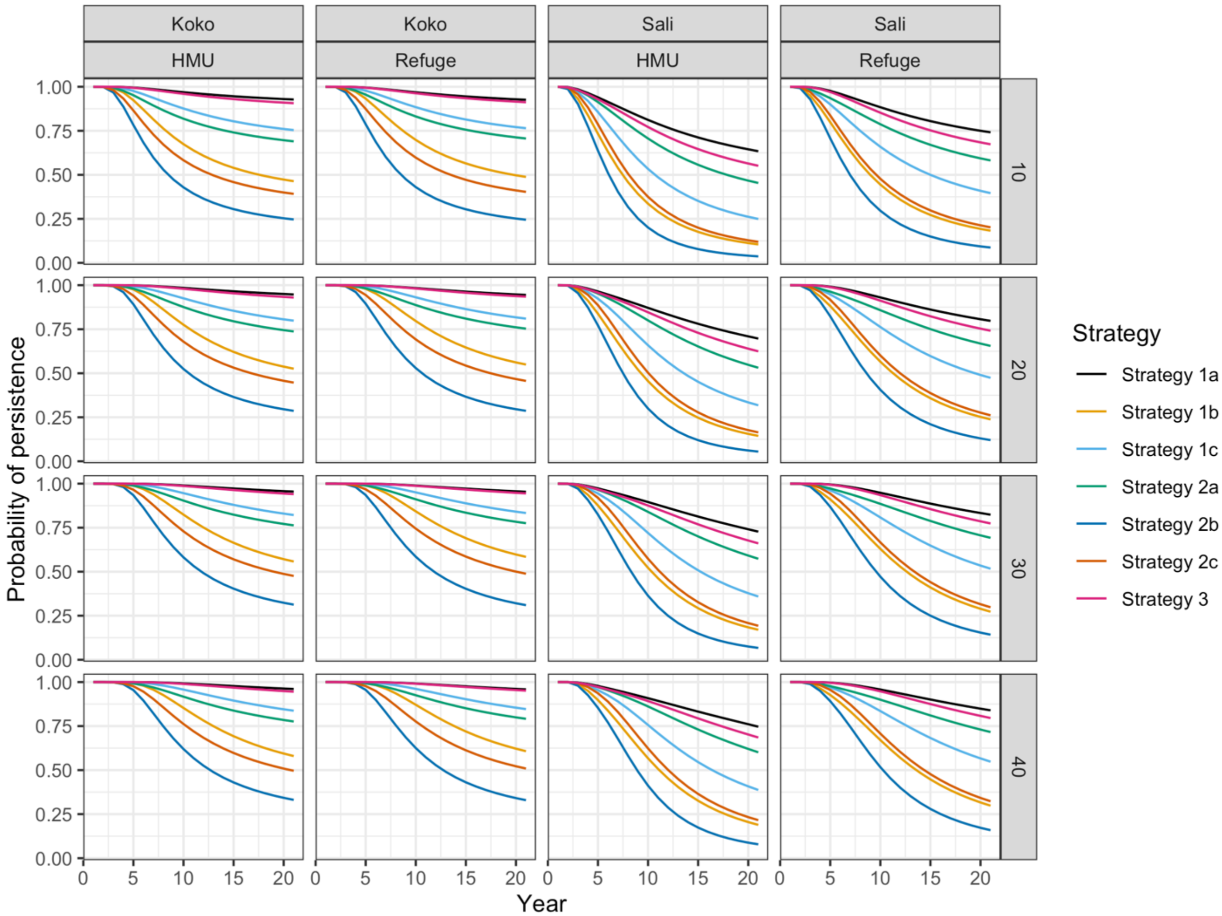

Figure 3.2. Mean cumulative probability of persistence over time of Ko'ko' and Såli at the HMU and Refuge under each alternative management strategy. Each row shows the distributions for initial adult abundances 10, 20, 30, or 40 adults. Strategies 1a, 1b, and 1c are the 'BTS + Mammal + Habitat' strategies, strategies 2a, 2b, and 2c are the 'BTS + Habitat' strategies, and strategy 3 is the 'BTS + Mammal' strategy. For a full description of alternative management strategies, see Table 3.1.

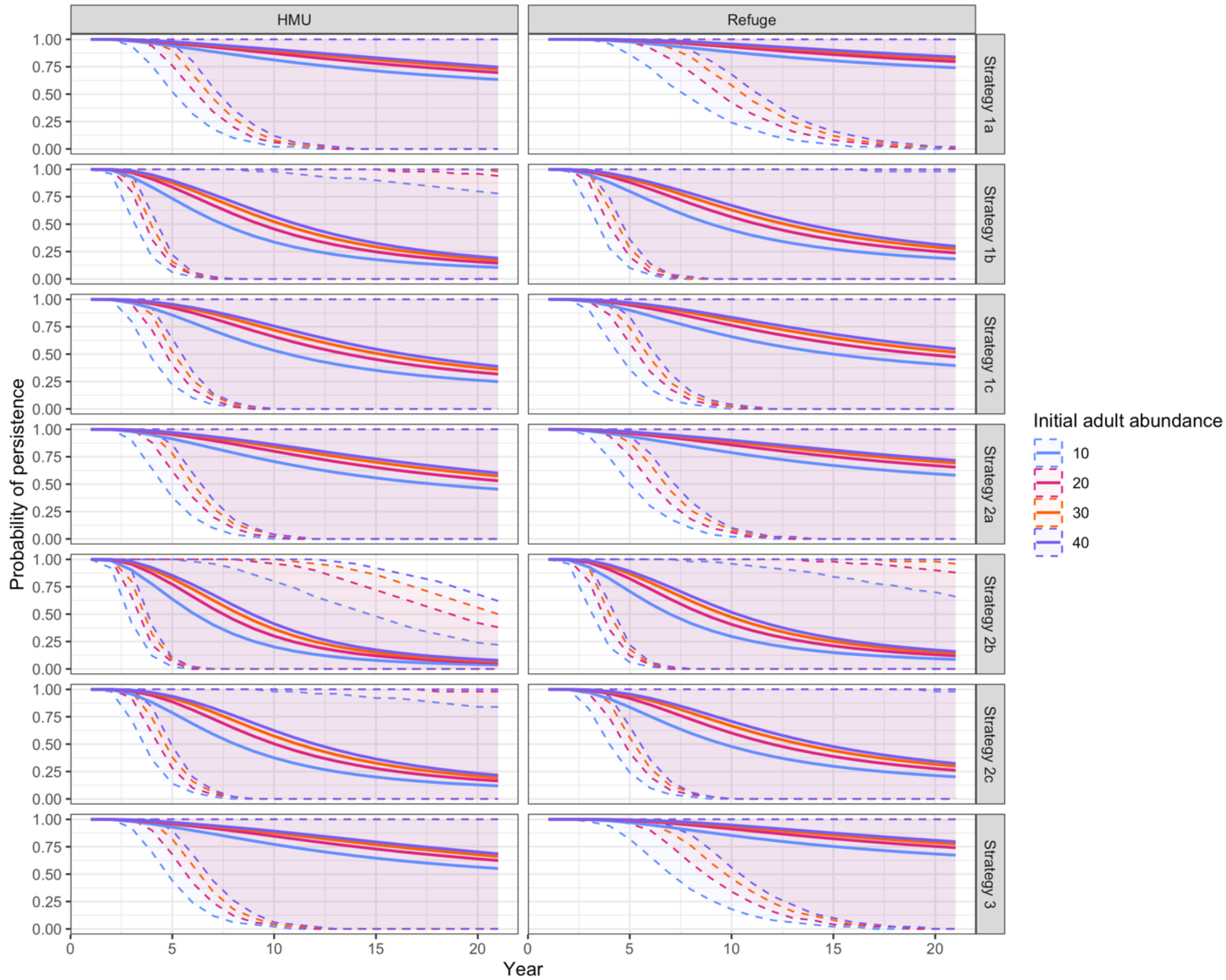

Figure 3.3. Mean and 95% credible intervals for probability of persistence of Såli at the HMU and Refuge over 21 years under each alternative management strategy and for four different initial adult abundances. Strategies 1a, 1b, and 1c are the ‘BTS + Mammal + Habitat’ strategies, strategies 2a, 2b, and 2c are the ‘BTS + Habitat’ strategies, and strategy 3 is the ‘BTS + Mammal’ strategy. For a full description of alternative management strategies, see Table 3.1.

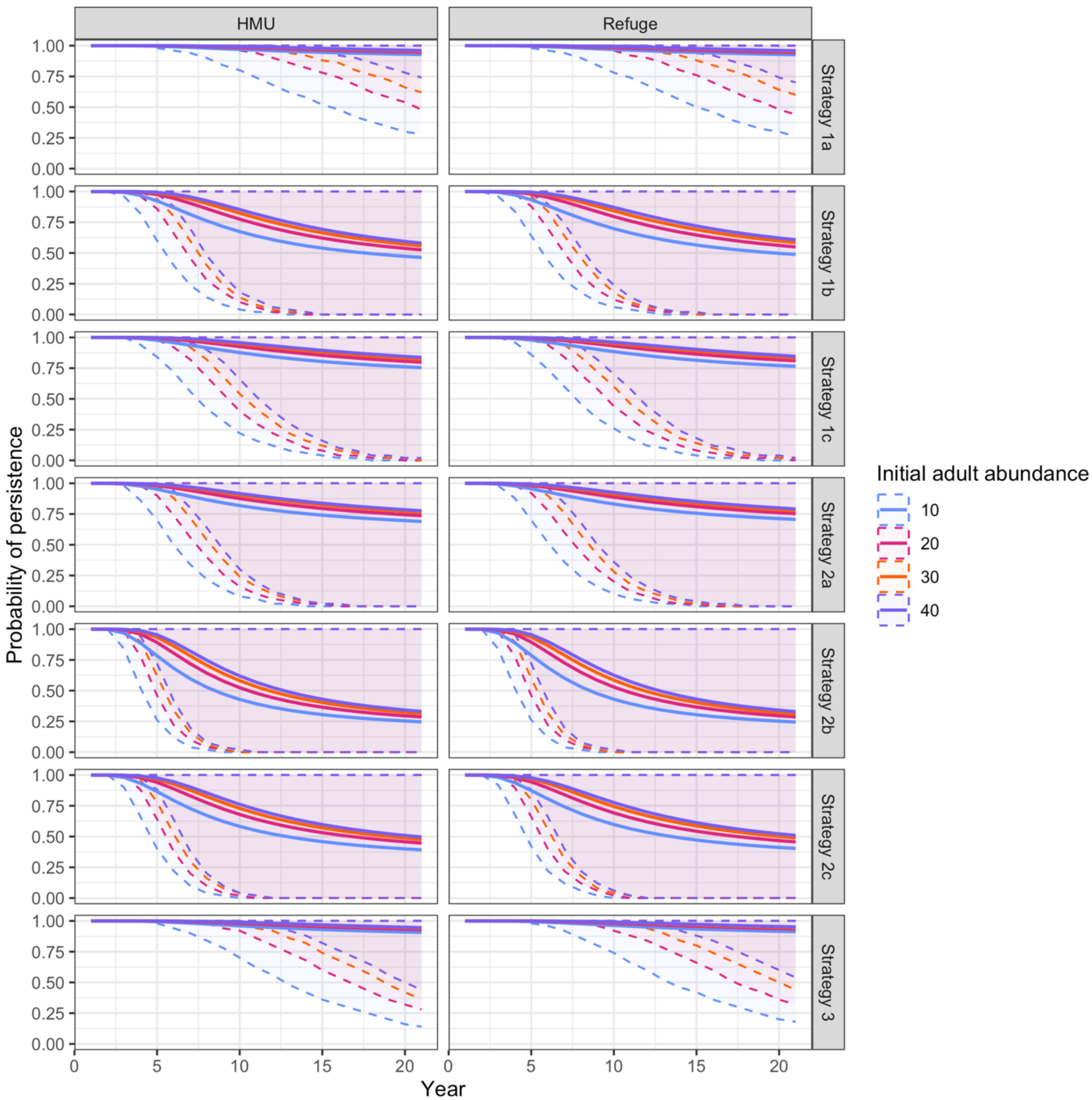

Figure 3.4. Mean and 95% credible intervals for probability of persistence of Ko’ko’ at the HMU and Refuge over 21 years under each alternative management strategy and for four different initial adult abundances. Strategies 1a, 1b, and 1c are the ‘BTS + Mammal + Habitat’ strategies, strategies 2a, 2b, and 2c are the ‘BTS + Habitat’ strategies, and strategy 3 is the ‘BTS + Mammal’ strategy. For a full description of alternative management strategies, see Table 3.1.

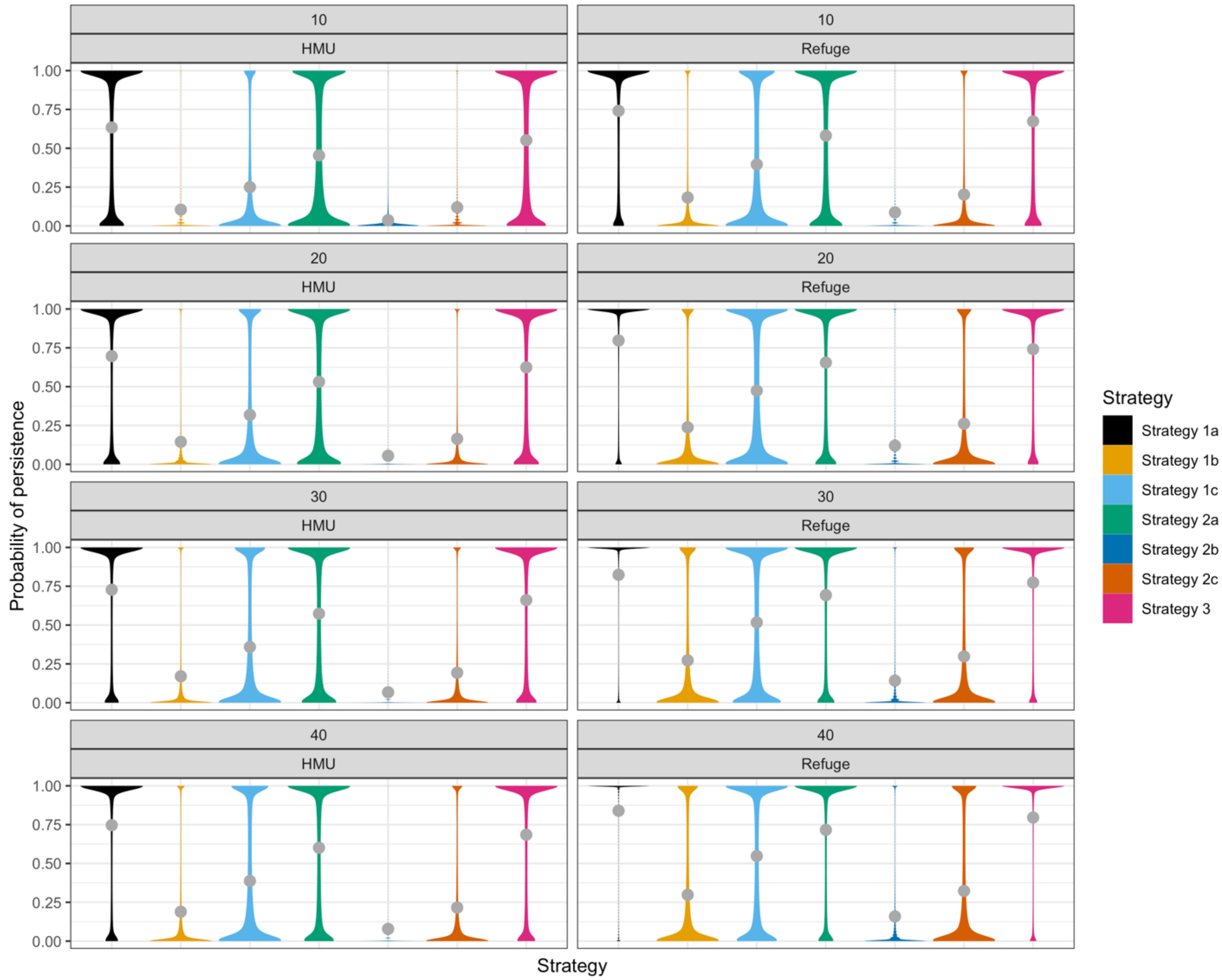

Figure 3.5. Density plots showing the distribution of probability of persistence for Såli at the HMU and the Refuge in Guam at year 21 under each alternative management strategy. Gray points are the mean values for each violin plot. Each row shows the distributions for initial adult abundances of 10, 20, 30, or 40 adults. Strategies 1a, 1b, and 1c are the 'BTS + Mammal + Habitat' strategies, strategies 2a, 2b, and 2c are the 'BTS + Habitat' strategies, and strategy 3 is the 'BTS + Mammal' strategy. For a full description of alternative management strategies, see Table 3.1.

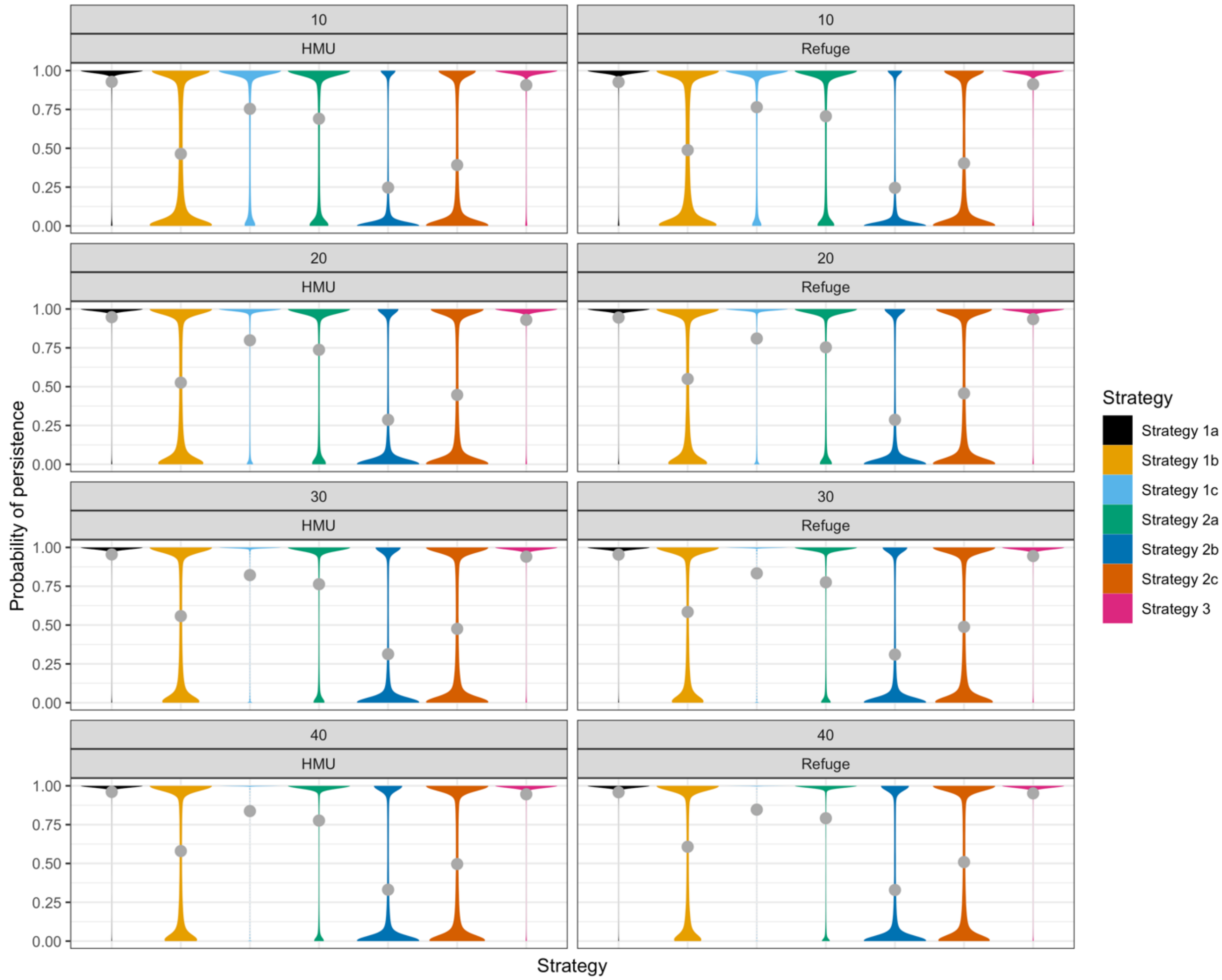

Figure 3.6. Density plots showing the distribution of probability of persistence for Ko'ko' to HMU and the Refuge in Guam at year 21 under each alternative management strategy. Gray points are the mean values for each violin plot. Each row shows the distributions for initial adult abundances of 10, 20, 30, or 40 adults. Strategies 1a, 1b, and 1c are the 'BTS + Mammal + Habitat' strategies, strategies 2a, 2b, and 2c are the 'BTS + Habitat' strategies, and strategy 3 is the 'BTS + Mammal' strategy. For a full description of alternative management strategies, see Table 3.1.

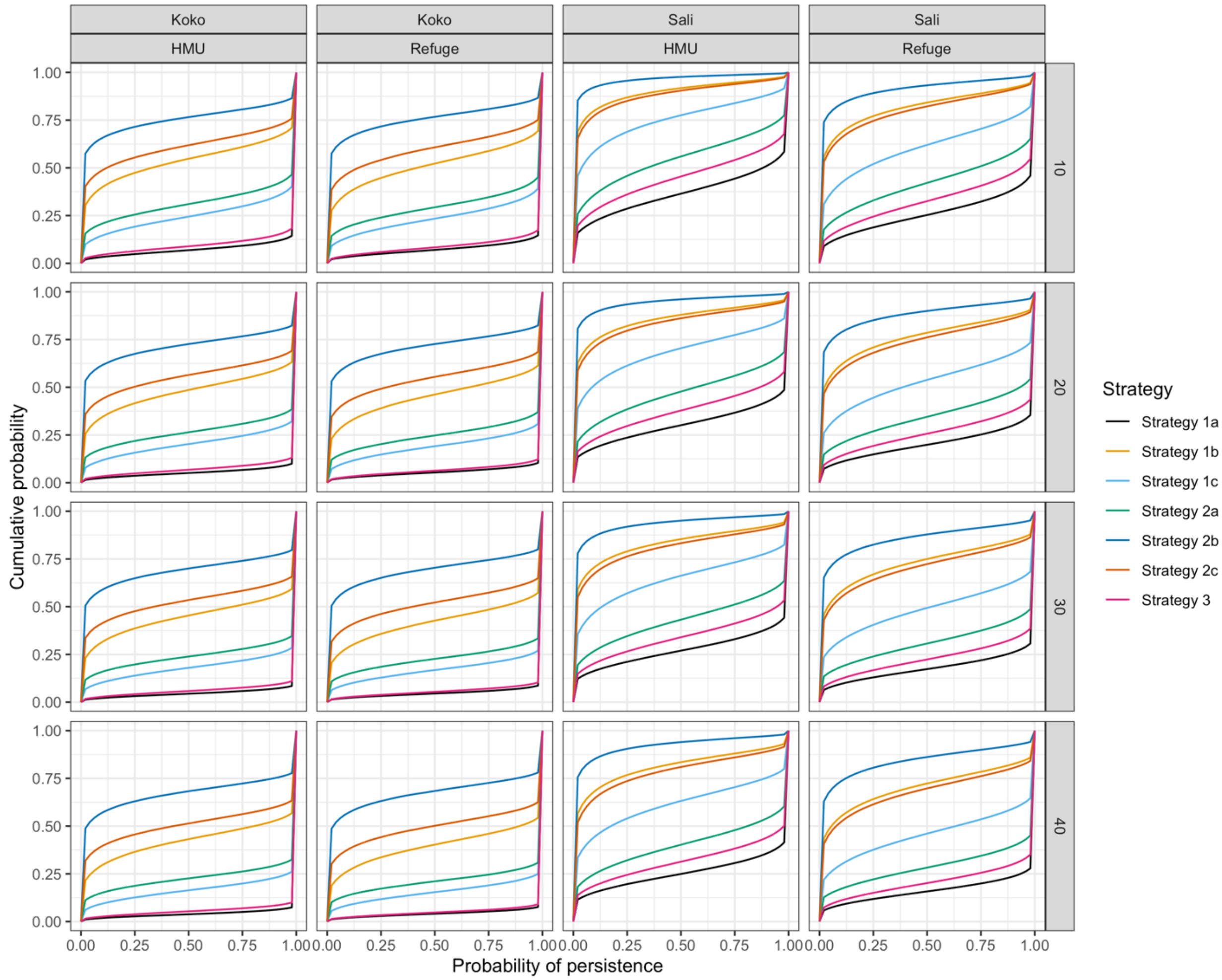

Figure 3.7. Cumulative probability distributions for probability of persistence of Ko'ko' and Såli at the HMU and the Refuge in Guam at year 21 under each alternative management strategy. The rows are the initial adult abundances (10, 20, 30, or 40 adults with equal number of males and females). Strategies 1a, 1b, and 1c are the 'BTS + Mammal + Habitat' strategies, strategies 2a, 2b, and 2c are the 'BTS + Habitat' strategies, and strategy 3 is the 'BTS + Mammal' strategy. For a full description of alternative management strategies, see Table 3.1.

3.9 APPENDIX 3

Table A3. Scientific experts and expert affiliations for expert elicitation process.

Scientific Expert	Affiliation
Amy Yackel Adams	Research ecologist, Fort Collins Science Center, U.S. Geological Survey
Eben Paxton	Research ecologist, Pacific Island Ecosystems Research Center, U.S. Geological Survey
Haldre Rogers	Associate professor, Department of Fish and Wildlife Conservation, Virginia Polytechnic Institute
Martin Kastner	PhD student, Department of Fish and Wildlife Conservation, Virginia Polytechnic Institute
William Pitt	Managing Director, Smithsonian Conservation Biology Institute

Chapter 4. USING CONSTRUCTED VALUE OF INFORMATION TO IDENTIFY UNCERTAINTIES IN THREATENED SPECIES MANAGEMENT PROGRAMS

Publication history: This study was co-authored with John G. Ewen, Stefano Canessa, Lynn Adams, Danielle F. Shanahan, Kari C. Beaven, Rachel Selwyn, and Sarah J. Converse. At the time this dissertation was published, this chapter was not in review with a journal.

4.1 ABSTRACT

Managers tasked with species conservation must make difficult decisions in the face of uncertainty. Sometimes, but not always, these uncertainties limit the achievement of management objectives by obscuring identification of the best management strategy. Approaching management through a decision-analytic lens provides access to a suite of tools for identifying management-relevant uncertainties, i.e., those uncertainties that limit management performance. These tools fit in the general category of value of information (VOI) analysis, and most involve quantification of VOI in units of the objective function. However, quantifying VOI may not be appropriate or feasible during the initial exploration of uncertainties due to the time required to conduct the analysis or because the number of uncertainties is exceedingly large. Constructed value of information is a recently described VOI tool that relies on elicitation of expert judgments on a constructed scale, offering a less taxing approach for identifying management-relevant uncertainties. Here, we illustrate the utility of the constructed value of information (CVOI) for identifying key uncertainties about stagnant growth in a population of reintroduced Hihi (Stitchbird, *Notiomystis cincta*) in New Zealand. CVOI, as applied to date, relies on expert judgment to estimate (1) *magnitude of uncertainty*, (2) *relevance*, and (3) *reducibility* based on a scoring rubric. Here, we developed a rubric that allows for decomposition of one of these elements, *relevance*, to assist experts in scoring through a more mechanistic

presentation of the concept of management relevance. The CVOI analysis involved (1) initial decision structuring, i.e., defining the problem, objectives, alternative management actions, and alternative hypotheses in collaboration with decision makers and scientific experts; (2) further refinement of hypotheses with subject-matter experts; (3) expert elicitation to score the CVOI components; and (4) a novel sensitivity analysis to test the robustness of results. Uncertainties about the impact of inbreeding and a male skewed sex ratio were identified as the highest research priorities for the Hihi population. Our results demonstrate both the utility and the customizability of CVOI for identifying uncertainties that are most limiting management outcomes.

4.2 Introduction

Managers tasked with threatened species conservation often face difficult decisions about how best to manage populations (Converse, 2020; Converse et al., 2013; Game et al., 2014; Gregory et al., 2012; Keeney, 1982; Seddon and Armstrong, 2016). One factor that makes decisions difficult is uncertainty, e.g., uncertainty about how a species will respond to management actions or other factors. These uncertainties may be a function of limited data or the inherent variability of ecological systems (Converse et al., 2013; Regan et al., 2002). There can be meaningful risk when decisions are made in the face of uncertainty (Canessa et al., 2020; Runge et al., 2020) and that risk may be reducible. For uncertainties that can be reduced, decision makers may choose to undertake monitoring or experimentation prior to taking action (Smith, 2020) or might choose to adopt an adaptive management framework, wherein uncertainty is reduced iteratively by monitoring the outcomes of management (Canessa et al., 2016; Walters, 1986; Williams, 2011). However, not all uncertainties are equally important, and reducing irrelevant uncertainties will waste scarce resources, including time that could be spent in achieving conservation objectives. Therefore, methods designed to identify uncertainties that restrict management performance, i.e., management-relevant uncertainties, offer tremendous value to managers (Canessa et al., 2015; Runge et al., 2011; Williams and Johnson, 2015).

Decision analysis, also known as structured decision making (SDM), is valuable for grappling with decisions that are complex and uncertain (Gregory et al., 2012; Hemming et al., 2022; Keeney, 1982; Runge et al., 2020). SDM involves decomposing problems into a standard set of component parts: a definition of the decision problem, objectives (what the decision maker wishes to achieve), action alternatives (the alternative strategies a decision maker may employ), predictive models (for predicting the outcomes of alternatives in terms of the objectives), and some method for identifying the best alternative for meeting the objectives (Gregory et al., 2012; Runge et al., 2020). Attention to the forms and characteristics of uncertainty is a principal characteristic of SDM. Identifying and accounting for uncertainties can, for example, help to identify action alternatives that are robust to uncertainty or indicate areas where collecting more information would be useful (Canessa et al., 2015; Converse et al., 2013; Runge et al., 2011). Under the umbrella of SDM is adaptive management, wherein uncertainty is reduced by applying management and then monitoring the outcomes to reduce uncertainty in an iterative process (Canessa et al., 2016; Walters, 1986; Williams, 2011).

One form of epistemic uncertainty, or uncertainty associated with limited knowledge of a system, is variously known as structural or model uncertainty (Regan et al., 2002). Structural uncertainty arises when there are multiple competing hypotheses, or models, describing how a system operates. Structural uncertainty can be reduced through the collection of basic monitoring data, experimental data, or data arising from adaptive management. In each case, data analysis allows us to update our belief regarding the veracity of hypotheses. However, determining whether and what data to collect to reduce structural uncertainties and improve future decision quality can be challenging. Data collection will not always result in the ability to address management-relevant uncertainties, and so there is a risk of wasting resources by collecting data with little management relevance.

A useful approach for identifying management-relevant uncertainty is value of information (VOI) analysis. VOI analysis is achieved through a set of tools for estimating the expected improvement in management outcomes should uncertainty be resolved fully (i.e., expected value of perfect information, EVPI; Runge et al. 2011, Canessa et al. 2015), should subsets of uncertainty be resolved (i.e., expected value of perfect partial information, EVPXI), or should uncertainty be resolved partially (i.e., expected value of sample information, EVSI). Focusing learning on uncertainties with high VOI has been identified as a best practice for both one-time decisions and decisions that are made iteratively through adaptive management (Runge et al., 2011; Williams et al., 2011). VOI analysis traditionally involves quantification of VOI in units of the objective function, i.e., the task is to estimate how much the performance on objectives would improve with a reduction of uncertainty.

In conservation applications, data for conducting VOI analysis are often either imperfect or entirely unavailable, and instead the elicitation of expert judgment is used to obtain the necessary information (Johnson et al., 2017; MacMillan and Marshall, 2006; Martin et al., 2012; Runge et al., 2011). Although traditional VOI analysis is valuable, eliciting the required information from experts may be cognitively infeasible or unreasonably time consuming when the number of uncertainties is large. Constructed value of information (CVOI) has recently been presented as an alternative to traditional VOI analysis. CVOI is a method that is used to approximate EVPI through elicitation of values on a constructed scale and is less cognitively taxing for experts than traditional VOI analysis (Runge et al., 2023). CVOI is based on the recognition that VOI can be conceptually decomposed into two elements, the degree of uncertainty (*magnitude of uncertainty*; Runge et al. 2023) and the relevance of the uncertainty to the decision (*relevance*). Scoring rubrics can be used to elicit information on each component from experts. CVOI is calculated by taking the product of *magnitude of uncertainty* and *relevance* scores for a given hypothesis. A third component, *reducibility*, scores the hypothesis in terms of the ability to collect information that can be used to investigate it. Taken together, when CVOI is assessed against *reducibility*, it provides a prioritization of hypotheses, with the highest priority given to those hypotheses that have a large amount of uncertainty and relevance to the decision, i.e., high CVOI, and those that are relatively tractable to address, i.e., high *reducibility* (Runge et al., 2023; Rushing et al., 2020).

To date, CVOI has been applied to only a handful of case studies. Rushing et al. (2020) used CVOI to prioritize uncertainties about land acquisition for migratory bird conservation under the threat of climate change. Lawson et al. (2022) used CVOI to identify uncertainties relevant to Eastern Black Rail (*Laterallus jamaicensis jamaicensis*) conservation in an adaptive management framework. Stantial et al. (2023) employed CVOI to prioritize uncertainties about avian marsh species' response to prescribed fire management alternatives. In each of these case studies, the authors refer to CVOI as qualitative value of information (QVOI) but are referring to the same method as described in Runge et al. (2023). Each of these case studies show the value of CVOI for identifying key uncertainties. Runge et al. (2023) note that there is flexibility in defining the ratio scales and narrative definitions of the scoring rubric used to elicit CVOI components, so long as the scores are on a ratio scale, suggesting that VOI theory could be advanced by demonstrating the customization of CVOI elicitation for different applications.

Here, we demonstrate a customized application of CVOI for prioritizing uncertainties about management of threatened populations. We apply CVOI to explore uncertainty about the reason for stagnant growth in a reintroduced population of Hihi (Stitchbird; *Notiomystis cincta*) found at Zealandia Wildlife Sanctuary in Wellington, New Zealand. Working with decision makers and experts in a collaborative process, we first determined the problem structure (i.e., definition of problem, objectives, and alternatives). Then, we worked with experts to generate explicit alternative hypotheses that may explain stagnant population growth. We then developed a customized CVOI rubric that subject-matter experts applied in a multiple-step elicitation process, and we used the results to calculate CVOI and *reducibility*. We used a bootstrapping procedure to characterize the uncertainty in the elicited scores and undertook a sensitivity analysis to evaluate the results for robustness. Our results demonstrate both the usefulness and the customizability of CVOI for prioritizing uncertainties relevant to conservation of threatened populations.

4.3 METHODS

4.3.1 STUDY SPECIES AND LOCATION

The Hihi is an endemic New Zealand passerine listed as Nationally Vulnerable, defined as facing a risk of extinction in the medium term, by New Zealand's Department of Conservation (Robertson et al., 2021). The species is listed as Vulnerable on the IUCN Red List (iucnredlist.org; accessed 9/19/2023). Prior to European arrival in New Zealand, Hihi were found throughout the North Island and on nearby offshore islands. European colonization brought threats, including introduced predators and habitat destruction, that led to the extinction of Hihi on all but the isolated island of Te Hauturu-o-Toi (Department of Conservation, 2005; Hihi Recovery Group, 2022). Initial efforts in the 1980s to establish additional Hihi populations on the island of Taranga using birds from Te Hauturu-o-Toi were unsuccessful (Bellingham et al., 2010; Castro et al., 1994), but efforts continued, leading to the first successful reintroduction on Kapiti Island in 1991 (Castro et al. 1994). Today, there are six reintroduced populations of Hihi, including two on offshore islands and four in fenced mainland reserves (Hihi Recovery Group, 2022). Hihi are locally managed through reducing the risk of predator incursions, controlling any predators that do bypass barriers, supplementary feeding of sugar water, maintenance of nest boxes, and control of nest parasites (Department of Conservation, 2005; Hihi Recovery Group, 2022). Through ongoing management of reintroduced populations, Hihi numbers have increased, but predation, disease, small population genetic effects, and climate change continue to threaten Hihi populations (Ewen et al., 2013).

One of the reintroduced Hihi populations is found at Zealandia Wildlife Sanctuary, a 225-ha predator-free fenced reserve, termed a ‘mainland island’, within the city of Wellington. The Zealandia population was established through translocations in 2005, with subsequent supplemental releases in several years following the initial reintroduction (Brekke et al., 2011; Miskelly and Powlesland, 2013). The current predicted trajectory of the Hihi population at Zealandia is uncertain, with the predicted growth rate poised between growth and decline (λ = 1.02, 95% CRI = 0.84 – 1.21). The low projected population growth at Zealandia appears to be driven largely by lower adult survival in this population compared to others (Parlato et al., 2021). Currently, the influences of threatening factors at Zealandia are not fully understood.

4.3.2 Workshop format and participants

The participants identified and invited to collaboratively develop the problem structure were Zealandia decision makers, national decision makers, and scientific experts. The problem structure, e.g., definition of the decision problem, objectives, and alternatives, was developed through four remote working group meetings (‘problem structuring meetings’). Once the problem structure was established, a group of Hihi experts were identified and invited to take part in the CVOI process, ensuring that there was a wide range of subject matter expertise present in the expert group. Three remote meetings were held with the expert group to undertake the CVOI process (‘CVOI meetings’; see ‘CVOI analysis’ section, below, for more details). A list of meetings and attendants is provided in Table 4.1.

4.3.3 Defining the decision problem

Decision makers charged with the management of Zealandia’s Hihi population were motivated to develop a decision framework for informing future Hihi management decisions given the existing uncertainty. In particular, decision makers were contemplating decisions about whether to continue to support the population with active management and supplementation. Because of the unknown factors contributing to uncertain population projections, managers were hesitant to use supplementation without understanding threats and the prospects for improved population growth. Furthermore, managers were considering ceasing supplementation and supportive management or translocating birds to another site in the face of a low probability of population persistence at Zealandia. Hence, the initial step in developing a framework to inform future Hihi management was to identify and prioritize uncertainties.

4.3.4 Objectives

Management decisions for Hihi occur at both national and local scales. New Zealand’s Hihi Recovery Group (hihiconservation.com) has developed national-scale management objectives, including (1) increasing the number of Hihi nationwide, (2) increasing the natural ecological setting of the Hihi, (3) reducing the cost of managing Hihi, and (4) increasing awareness and appreciation of Hihi. Based on the first three national objectives, continued investments at a site with poor prospects for success may be suboptimal. Conversely, the Zealandia Hihi population fulfills an important role as a site where the public can access and appreciate this species, which is relevant to the fourth national objective. Zealandia is located in the capitol city of Wellington, and in 2018-2019, the last report year prior to the COVID-19 pandemic, nearly 140,000 visitors were recorded at Zealandia (Zealandia Team, 2019). At a local scale, Zealandia has a range of management objectives that are inspired by the organization's vision of restoring local forest and freshwater ecosystems to their pre-human state (Burns et al., 2012; Marques et al., 2019). Zealandia is also focused on education and outreach through local programs and tourism (www.visitzealandia.com). The specific objectives articulated for Hihi at Zealandia were the same as the four fundamental objectives established for the national Hihi recovery program, namely: (1) maximize the total number of Hihi, (2) maximize the natural ecological setting of the Hihi population (e.g., minimize supplementary feeding or nest boxes), (3) minimize the cost of Hihi management, and (4) maximize public appreciation and awareness of Hihi. In addition, Zealandia managers also articulated that they would like to maximize learning about the threats to Hihi populations, with the hope that what is learned at Zealandia would benefit other reintroduced Hihi populations.

4.3.5 Alternative hypotheses and alternative management actions

During the problem structuring meetings, we asked participants to develop a list of hypotheses explaining the stagnant population growth of Hihi at Zealandia, along with alternative management actions that would be promising to implement if a given hypothesis was true. Participants worked collaboratively to ensure that hypotheses specified the threat and the population process that the threat impacted. They also identified management actions that could potentially be taken to counter the threat (Lawson et al., 2022; Stantial et al., 2023). Hypotheses were further developed during the first CVOI meeting with experts participating in the CVOI process. The hypotheses ranged widely in the type of threat impacting the population (e.g., dispersal outside of the reserve, inbreeding depression, weather, disease, competition, and interacting combinations of threats), while the population processes hypothesized to be impacted by threats included female survival, adult survival, fledgling survival, or breeding success. See Table 4.2 for a full list of alternative hypotheses.

Potential management actions that were promising under various hypotheses were developed alongside the hypotheses (see Table 4.3 for alternative management actions). In addition to the management alternatives described for hypotheses specific to Zealandia Hihi, general alternatives were also generated to explore Hihi management at a national level. The broader, national-scale alternatives included both the continuation of management at Zealandia along with reducing the resources allocated to the population in various ways (e.g., stopping supplemental releases, discontinuing supplemental feeding, or translocating remaining Hihi to another site; see 'General alternatives' in Table 4.3).

4.3.6 CVOI Analysis

A standardized rubric was used to score the three CVOI components (*magnitude of uncertainty*, *relevance*, and *reducibility*) on a ratio scale (Runge et al., 2023). We deviated from the general form of the rubric used in Rushing et al. (2020), Lawson et al. (2022), and Stantial et al. (2023) in several significant ways: (1) we specified scoring scales from 0 to 4 rather than 0 to 3; (2) we separated *relevance* into two parts; and (3) we provided narrative definitions for scores that were specific to the Hihi decision problem. First, the decision to ask experts to score all components from 0 to 4 was largely driven by the recognition that it may be cognitively more tractable for experts to express judgements on a ratio scale using values from 0 to 4, because, e.g., a score of 1 is half as large as a score of 2, and a score of 2 is half as large as a score of 4. Potentially less intuitive ratios result from a 0 to 3 scale. Second, we decomposed the *relevance* component into two parts to explicitly capture the degree to which the hypothesized threat is impacting the population and the degree to which the hypothesized threat can be reduced through management actions. If a threat is having a very small impact on the population, value of information will not be high and if a threat cannot be mitigated through management, value of information will not be high. Third, we supplied narrative definitions associated with different scores that were specific to the Hihi decision context, and in doing so we created a rubric that can easily be modified to work in many threatened population management contexts. Throughout rubric development and elicitation, we focused on ensuring that a ratio scale was maintained, as that is imperative for approximating EVPI (Runge et al., 2023), and we also ensured that the narrative definitions followed best practices for constructed scales (Keeney and Gregory, 2005).

For *magnitude of uncertainty*, experts were asked to score their uncertainty regarding the "degree of impact a hypothesis was having on the population based on current knowledge, theory, personal experience, or any other rationale". A score of 0 for *magnitude of uncertainty* indicated an expert's belief that "there is little to no uncertainty regarding the degree of impact the hypothesis is having on the population" and a score of 4 indicated an expert's belief that "there is little to no information currently available to discern the degree of impact this hypothesis is having on the population." We divided *relevance* into two parts: *relevance (a)* was the degree to which a hypothesis is impacting the population, and *relevance (b)* was the degree to which available management actions could reduce the effect of the hypothesized threat on the population. For *relevance (a)*, a score of 0 indicated an expert's belief that, even if the hypothesis were true, "the hypothesis has no to very little impact on the population," whereas a score of 4 indicated an expert's belief that "if true, the hypothesis has a large impact on the population." In scoring *relevance (a)*, experts were asked to base their judgment on their best assessment of the impact, while capturing their uncertainty in that assessment in the *magnitude of uncertainty.* For *relevance (b)*, a score of 0 indicated an expert's belief that "management actions will have no to very little effect on the hypothesized effect" and a score of 4 indicated an expert's belief that "management actions will mostly or completely reduce the hypothesized effect." Lastly, for *reducibility*, a score of 0 indicated an expert's belief that "the uncertainty about the hypothesis is unlikely to be reduced within a relevant management timeline due to the expected cost, logistics, and ability to obtain necessary data," whereas a score of 4 indicated an expert's belief that "the uncertainty about the hypothesis could be mostly or completely reduced within a relevant management timeframe based on the expected cost, logistics, and ability to obtain necessary data." See definitions for intermediate *magnitude of uncertainty*, *relevance (a)*, *relevance (b)*, and *reducibility* scores in Table 4.4.

To reflect the wide range of hypotheses about the causes of Hihi population trends, subject-matter experts with a variety of expertise were invited to participate in CVOI elicitation. We identified experts based on conversations with Zealandia and New Zealand Hihi Recovery Group leadership. During our first CVOI meeting with experts, we shared the problem structure that had been developed, discussed the initial set of hypotheses, and asked the experts to share any additional hypotheses that they thought relevant to explaining the Zealandia Hihi population's stagnant growth (see Table 4.2 for the final list of hypotheses). We held a second CVOI meeting shortly after the first, where we discussed the CVOI process, had the experts work through a practice question, and provided instructions for the experts to complete their scores. We compiled an annotated bibliography of relevant literature and a document summarizing Hihi monitoring and research at Zealandia, and made those available to the experts prior to the elicitation. We used a modified Delphi process for elicitation (Hemming et al., 2018; MacMillan and Marshall, 2006). Experts completed their first round of CVOI scores independently and offline using a fillable scoring sheet. After experts returned their first round of scores, a third CVOI meeting was held to discuss the results from the first round with experts. Following the discussion meeting, experts continued the discussion online by commenting on a shared document containing the results from the first round of elicitation. Experts were then given the chance to revise their first round CVOI scores and the scores from the second round were taken as the final results.

After each round of elicitation, *relevance* was computed for each expert by multiplying *relevance (a)* and *relevance (b)* and rescaling the product to 0 – 4. The *relevance* score was then multiplied by *magnitude of uncertainty* to obtain CVOI for each expert and hypothesis. For each hypothesis, the mean of the CVOI scores was taken across experts and was plotted against the mean of the *reducibility* scores across experts (Lawson et al., 2022; Runge et al., 2023; Rushing et al., 2020; Stantial et al., 2023). The priority rankings of the hypotheses, i.e., highest priority, high priority, medium priority, and low priority, were determined by where scores fell in the quadrants determined by the median values of the CVOI and *reducibility* scores across all hypotheses.

4.3.7 Sensitivity analysis

To test the sensitivity of hypothesis ranking and to estimate the uncertainty around the mean scores for each hypothesis, we bootstrapped the elicited responses. For each hypothesis, the elicited component scores, i.e., *magnitude of uncertainty*, *relevance (a)*, *relevance (b)*, and *reducibility*, for the set of experts were each resampled with replacement 1000 times to create

1000 sets of scores for each component and hypothesis. CVOI and *reducibility* were then calculated for each of the 1000 sets of scores as described above. The bootstrapped samples were used to estimate the standard error of the mean for each CVOI component. Further, the bootstrapped samples were used to gauge the sensitivity of each priority ranking to expert-to-expert variation by calculating the percentage of bootstrapped samples that retained the mean priority ranking. Bootstrapping code and all group facilitation materials (e.g., instructions, practice question, scoring sheets, and background information) can be found on GitHub (https://github.com/sipeha/Dissertation_Chapter_4).

4.4 RESULTS

We elicited scores for *magnitude of uncertainty*, *relevance (a)*, *relevance (b)*, and *reducibility* from nine experts for 22 hypotheses (Table 4.5). The hypotheses with the highest mean *magnitude of uncertainty* across experts were the Weather + Disease Hypothesis (#9c, mean score of 3.00, SE = 0.15) and the Sex Ratio + Disease Hypothesis (#9d, mean score of 3.00, SE = 0.22), while the Fence Mortality Hypothesis (#10, mean score of 0.33, SE = 0.15) and the Mammal Control Hypothesis (#13, mean score of 0.33, SE = 0.16) were the hypotheses with the lowest *magnitude of uncertainty*. The Dispersal Predation Hypothesis (#2) had the highest *relevance (a)* with a mean score of 2.67 (SE = 0.23). The Fence Mortality Hypothesis (#10, mean score of 0.11, SE = 0.11) and the Mammal Control Hypothesis (#13, mean score of 0.11, SE = 0.10) both had the lowest *relevance (a)*. The Sex Ratio Hypothesis (#5) had the highest *relevance (b)*, with a mean score of 2.56 (SE = 0.32), whereas the Native Species Predation Hypothesis (#1) had the lowest (mean score of 0.11, SE = 0.11). The Sex Ratio Hypothesis (#5) had the highest overall *relevance* (the product of *relevance (a)* and *relevance (b)* rescaled to 0 –

4) with a mean score of 1.39 (SE = 0.28) and the Fence Hypothesis (#10) had the lowest (mean score of 0.00, SE = 0.01; Table 4.5).

The Sex Ratio + Disease Hypothesis (# 9d) was the hypothesis with the highest CVOI (*magnitude of uncertainty* * *relevance*) with a mean score of 2.67 (SE = 0.68). The Fence Hypothesis (#10, 0.00, SE = 0.00) and the Mammal Control Hypothesis (#13, 0.00, SE = 0.02) were the hypotheses with the lowest mean CVOI (Table 4.5, Figure 4.1). The hypothesis with the highest *reducibility* was the Inbreeding Hypothesis (#4) with a mean score of 2.67 (SE = 0.44), whereas the Weather + Disease Hypothesis (#9c) was the hypothesis with the lowest *reducibility*, with a mean score of 0.44 (SE = 0.17; Table 4.5, Figure 4.1).

The Dispersal Predation Hypothesis (#2), Inbreeding Hypothesis (#4), Sex Ratio Hypothesis (#5), and Disease Hypothesis (#7) fell into the 'highest priority' ranking, i.e., they were above the median score on both CVOI and *reducibility*. Seven hypotheses fell into the 'high priority' ranking (above the median on CVOI but below the median on *reducibility*), seven fell into the 'medium priority' ranking (below the median on CVOI but above the median on *reducibility*), and four fell into the 'low priority' ranking (below the median on both; Figure 4.1).

4.4.1 Sensitivity analysis

Here we report on the sensitivity of hypothesis ranking to expert-to-expert variation for those hypotheses that fell into the highest priority ranking based on their mean scores. We also highlight hypotheses that had overall robustness to expert-to-expert variation, defined here as hypotheses for which >90% of the bootstrapped samples retained the ranking of the mean. Hypotheses that were moderately sensitive to expert-to-expert variation were defined as hypotheses with >75% but less than 90% of the bootstrapped samples retaining the ranking of the mean. High sensitivity to expert-to-expert variation was defined here as hypotheses with

<75% of the bootstrapped samples retaining the ranking of the mean. For all other hypotheses, see Table 4.6, which provides the percent of bootstrapped samples that fell into each priority ranking for each hypothesis.

For the four hypotheses that had a highest-priority ranking, based on mean scores, one was robust to expert-to-expert variation, two were marginally sensitive, and one was highly sensitive. The Sex Ratio Hypothesis (#5) was robust to expert-to-expert variation, with 93.9% of the bootstrapped samples falling into the highest-priority quadrant. The Dispersal Predation Hypothesis (#2) and the Inbreeding Hypothesis (#4) were moderately sensitive, with 85.8% and 78.5% of bootstrapped samples falling into the highest-priority quadrant. The Disease Hypothesis (#7) was highly sensitive to expert-to-expert variation, with 53.1% of the bootstrapped samples falling into the highest-priority quadrant and 42.0% falling into the high-priority quadrant.

Two high priority hypotheses, the Inbreeding + Disease Hypothesis (#9b) and the Sex Ratio + Disease Hypothesis (#9d), were robust to expert-to-expert variation, with 97.8% and 92.2%, respectively, of the bootstrapped samples falling into the high-priority quadrant. The Native Species Predation Hypothesis (#1), the Fence Mortality Hypothesis (#10), the Wasp Ingestion Hypothesis (#11), and Mammal Control Hypothesis (#13), all of which fell into the medium-priority quadrant, were robust to expert-to-expert variation with 93.5%, 92.8%, 92.8%, and 98.9%, respectively, of the bootstrapped samples falling into the medium-priority quadrant. The Weather + Disease Hypothesis (#9c) was the only low priority hypothesis that was robust to expert-to-expert variation with 94.8% of bootstrapped samples falling into the low-priority quadrant (Table 4.6).

4.5 Discussion

Here, we demonstrated the use of CVOI to identify the highest priority management-relevant uncertainties for a threatened population. We developed a rubric with applicability to a wide variety of threatened population management contexts and applied it to evaluate a large number of hypotheses for stagnant population growth of Hihi at Zealandia Wildlife Sanctuary in Wellington, New Zealand. We also demonstrated an approach to sensitivity analysis, using bootstrap resampling to determine the sensitivity of priority rankings to between-expert variation.

The Dispersal Predation Hypothesis (#2), Inbreeding Hypothesis (#4), and the Sex Ratio Hypothesis (#5), and the Disease Hypothesis (#7) were identified as the highest priority hypotheses (Figure 4.1). The priority ranking of the Sex Ratio Hypothesis was robust to between-expert variation, whereas the Dispersal Predation Hypothesis and the Inbreeding Hypothesis showed slight sensitivity and the Disease Hypothesis showed substantial sensitivity to expert-to-expert variation (Table 4.6). Therefore, the Sex Ratio Hypothesis, Dispersal Predation Hypothesis, and Inbreeding Hypothesis are the hypotheses for which there was relatively consistent judgment across experts that reducing these uncertainties could lead to improved management outcomes.

The Dispersal Predation Hypothesis scored the highest in *relevance (a)* among all the hypotheses (Table 4.5), possibly owing to the observations of Hihi moving outside of Zealandia's fenced reserve and the expected high rate of predation by non-native predators outside of the fence. The Sex Ratio Hypothesis scored the highest in *relevance (b)* and overall *relevance*, likely because a skewed sex ratio could be relatively easily addressed by removing males from the population or otherwise excluding them from certain areas frequented by females. In 2022, the observed sex ratio at Zealandia was approximately 4 males for every 1 female, which is an increase from the observed sex ratio of approximately 2 males to 1 female in 2020, and male Hihi exhibit aggressive behavior towards females during the breeding season (Brekke et al., 2013; Ewen et al., 2011, 2004). The Inbreeding Hypothesis scored the highest in *reducibility* (Table 4.5). This is perhaps due to the fact that reducing uncertainty about inbreeding could be accomplished by re-analyzing existing pedigree data that have been analyzed previously (Brekke et al., 2011; Rutschmann et al., 2020).

If evidence to support the Dispersal Predation Hypothesis, the Sex Ratio Hypothesis, or the Inbreeding Hypothesis was found through data analysis or experimentation, there are conceivable management actions to counter each threat. Management actions that could be taken to counter the impact of the Dispersal Predation Hypothesis include reduction of bird feeding outside of the reserve and increasing non-native predator control in the area surrounding the reserve. For the Inbreeding Hypothesis, potential management actions could include translocating Hihi to Zealandia from another population or exchanging Hihi between Zealandia and another population. Potential management actions that could be taken should there be evidence supporting the Sex Ratio Hypothesis include translocating excess males to another reintroduction site, excluding males from feeder locations, or translocating additional females to Zealandia. Reducing uncertainty about the importance of these hypotheses has potential to improve future management for the Zealandia population as well as other reintroduced populations.

Four hypotheses were ranked as low priority through the CVOI analysis, including: the Weather + Sex Ratio Hypothesis (#9a), the Weather + Disease Hypothesis (#9c), the Inter/intraspecific Competition Hypothesis (#16), and the Phenology Hypothesis (#17; Figure

4.1). All of the low priority ranking hypotheses had high *magnitude of uncertainty* scores. However, they also had low to moderate *relevance (a)*, low *relevance (b)* and low overall *relevance* scores, such that the result was overall low CVOI scores. This, in conjunction with limited ability to collect information to reduce uncertainty (low *reducibility*; Table 4.5), resulted in these hypotheses having a low priority ranking. Thus, while experts were uncertain about the degree of impact these hypotheses are having on the population, their best assessment was that it was low to moderate (*relevance (a)*) and that management actions would be ineffective at reducing the impact (*relevance (b)*). Take for example, the Weather + Disease Hypothesis, which posits that cold weather events during the early breeding season cause stress to females that makes them more susceptible to disease. This hypothesis had the highest score on *magnitude of uncertainty*, but a low score on *relevance (b)* and on *reducibility*. The low score on *relevance (b)* suggests the difficult nature of managing for both weather and disease. In addition, experts judged that uncertainty about the Weather + Disease Hypothesis was unlikely to be reduced in a relevant timeframe, likely given the difficulty of collecting information about the interacting effects of changing weather and disease.

For the full set of hypotheses, there was a slightly negative relationship between *magnitude of uncertainty* and *reducibility*, i.e., the higher the *magnitude of uncertainty*, the lower the *reducibility* (Figure 4.2). Since Hihi are a well-studied species, this pattern likely reflects that hypotheses that are relatively easy to address are more likely to have been addressed already. Many of the hypotheses that scored high in *magnitude of uncertainty* and low in *reducibility* were found in the high priority quadrant, e.g., Inbreeding + Disease Hypothesis (#9b), Sex Ratio + Dispersal Hypothesis (#9d), and Habitat + Disease Hypothesis (#9f), indicating that they do represent opportunities for improving management decisions. However, the cost of investing resources to reduce such complex hypotheses will need to be weighed against the potential gain by decision makers (Rushing et al., 2020).

In selecting hypotheses on which to focus, in addition to looking at priority ranking and the robustness of that ranking, managers may also consider dominance. A non-dominated hypothesis is one in which an increase on either CVOI or *reducibility* cannot be achieved without a sacrifice in the other (Marler and Arora, 2004; Williams and Kendall, 2017). The highest priority hypotheses that are non-dominated are the Dispersal Predation Hypothesis (#2), the Inbreeding Hypothesis (#4), and the Sex Ratio Hypothesis (#5) (Figure 4.1). While the Disease Hypothesis (#7) is a highest priority hypothesis, it is dominated by the Sex Ratio Hypothesis (#5) (Figure 4.1). Selecting one hypothesis on which to focus from amongst the non-dominated highest priorities would require a value judgement made by the decision maker, essentially trading off the expected management benefit against the cost of learning. However, other issues, such as availability of the necessary scientific expertise and the availability of non-fungible funding sources (e.g., a request for proposals focused on topics in genetics), are also likely to inform research priorities. Furthermore, if multiple research priorities can be addressed simultaneously, dominance is less critical, as in that case it is the set of hypotheses, rather than individual hypotheses, that comprise the alternatives.

CVOI is a relatively new method and the scoring rubric used was customized specifically for the Zealandia Hihi decision context, so we took steps to ensure experts understood the process and had ample resources to consult during elicitation (Hanea et al., 2017; Hemming et al., 2018). An annotated bibliography with pertinent literature for each hypothesis was provided to the experts to aid in scoring *magnitude of uncertainty* (as recommended in Stantial et al. 2023), which ensured that experts from all backgrounds had the same base information to reference. Additionally, a document with relevant observational data on Zealandia Hihi was shared to aid experts in scoring *magnitude of uncertainty*, but also likely influenced other component scores, e.g., *reducibility* as they could see what data were available for analyses. The onboarding CVOI meeting and the discussion CVOI meeting with the expert group after the first round of elicitation were particularly useful for resolving linguistic uncertainty about our rubric (Regan et al., 2002). However, due to time limitations, the group was not able to discuss all the first-round results during the remote discussion meeting. An online document was created for experts to continue the discussion, and while only a few experts contributed to the discussion, many of them seemed to gain insight from it, as all experts revised their scores during the second round of elicitation.

By combining an extensive list of hypotheses generated by a group of experts from varied backgrounds with a process for evaluating those hypotheses using CVOI has resulted in a prioritization of uncertainties for managers to consider. We safeguarded against including hypotheses that were not well conceived by requiring hypotheses to include a threat and the demographic process that the threat impacts. Participants also generated related management actions that may counter the hypothesized threat (Lawson et al., 2022; Stantial et al., 2023). Even with limitations, the number of hypotheses generated was large and traditional VOI analysis would have been time intensive and taxing for experts. The results from this work could be further refined through quantitative VOI analysis on a subset of hypotheses (Canessa et al., 2015; Runge et al., 2011). Additionally, the smaller subset of highest-priority uncertainties could be used to frame an adaptive management program in which uncertainties could be reduced over time by applying management, monitoring management outcomes, and updating beliefs in an iterative process (Nichols et al., 2007; Runge et al., 2011; Smith et al., 2013; Williams et al., 2011).

The results presented here will help guide research activities in the short and medium terms. Furthermore, the decision framework we developed will serve as a starting point for future decision making. The learning outcomes from this effort will also have application to other reintroduced populations of Hihi by providing a better understanding of how potential management actions may influence threats. Many of the reintroduced populations of Hihi have benefitted from the resolution of uncertainty from Hihi in a single reintroduced population. For example, an adaptive management effort clarified the importance of supplemental feeding in supporting reintroduced Hihi populations (Armstrong et al., 2007) and supplemental feeding is now used widely and is seen as a key component of supportive management for this species. Through a focus on learning and the application of structured decision processes, the national population of Hihi continues to grow (Hihi Recovery Group, 2022).

Our case study offers an example of how substantial structural uncertainty about a threatened species can be managed by approaching the problem through a decision-analytic lens. Through our threatened species application, we demonstrated how CVOI can be customized to the decision context, which we hope will provide a helpful example for others who are seeking to employ this method. The results of our work show the utility of CVOI for identifying and prioritizing uncertainties that impede management outcomes. CVOI presents a tractable, effective, and adaptable approach for prioritizing the critical uncertainties present in a problem and, through research and monitoring of prioritized uncertainties, management outcomes are likely to be improved.

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4.8 Tables and Figures

Table 4.1 Remote group meetings, meeting activities, participants, and participant roles in the process.

Problem structuring meetings	
Meeting 1	Definition of the decision problem, objectives
Meeting 2	Generation of alternative hypotheses and alternative actions
Meeting 3	Generation of alternative hypotheses and alternative actions
CVOI meetings	
Meeting 1	Project background, problem structure, hypothesis generation, and hypothesis discussion
Meeting 2	CVOI onboarding, practice question, instructions for CVOI
Meeting 3	Discussion of first round of elicitation

Participant	Meeting	Role[a]
John Ewen[1]	Problem structuring meetings	National decision maker
Stefano Canessa[2]	Problem structuring meetings	Decision analyst
Lynn Adams[3]	Both	National decision maker, Expert
Danielle Shanahan[4]	Both	Local decision maker
Kari Beaven[4]	Both	Expert
Rachel Selwyn[4]	Both	Expert
Ellen Irwin[4]	CVOI Meetings	Expert
Jo Ledington[4]	CVOI Meetings	Expert
Kate McInnes[3]	CVOI Meetings	Expert
Troy Makan[3]	CVOI Meetings	Expert
Patricia Brekke[1]	CVOI Meetings	Expert
Anna Santure[5]	CVOI Meetings	Expert
Caitlin Andrews[6]	CVOI Meetings	Expert
John Stewart[7]	CVOI Meetings	Expert

[a]"Expert" denotes those who provided judgments; also denoted are those who have ultimate authority for decision making at national or local scales; one additional participant with decision analysis skills participated in the problem structuring meetings.
[1]Zoological Society of London
[2]Bern University
[3]Department of Conservation
[4]Zealandia
[5]Auckland University
[6]Tufts University
[7]Supporters of Tiritiri Matangi

Table 4.2 Table of alternative hypotheses ($n = 22$) about what may be causing stagnant population growth of Hihi at Zealandia. Note that hypotheses 9a-9e are combinations of other hypotheses in the table.

Hypothesis number	Hypothesis description	Hypothesis Name
1	Predation by native species Kārearea (New Zealand Falcon, *Falco novaeseelandiae*) or Ruru (Morepork, *Ninox novaeseelandiae*) is reducing adult survival.	Native Species Predation
2	Birds are moving outside of the fence into surrounding neighborhoods to feed and are being depredated, reducing adult and fledgling survival.	Dispersal Predation
3	Birds are dispersing away from Zealandia and thus are lost to the population, functioning from the standpoint of the population as a reduction in survival.	Dispersal
4	Inbreeding depression is reducing survival and/or breeding success and thereby dampening population growth.	Inbreeding
5	A male-skewed sex ratio is resulting in harassment of females by males, which reduces female survival or breeding success.	Sex Ratio
6	Weather events, specifically cold temperatures in the early breeding season, reduce breeding success and survival of females.	Weather
7	Disease, either aspergillosis or others (e.g., Toxoplasmosis, trematodes, Plasmodium sp., avian malaria, internal or external parasites), is reducing adult survival, fledgling survival, or breeding success.	Disease
8	Current habitat conditions result in poor nutrition (quality or quantity of food) and reduced survival.	Habitat
9a	Weather events (Hypothesis 6) are causing females to approach feeders at a higher rate, where they are harassed by males (Hypothesis 5), which is reducing female survival and breeding success.	Weather + Sex ratio
9b	Inbreeding depression (Hypothesis 4) is increasing the disease susceptibility of Hihi (Hypothesis 7), thereby reducing survival and breeding success.	Inbreeding + Disease
9c	Weather events (Hypothesis 6) are causing stress to females, making them susceptible to disease (Hypothesis 7), reducing female survival.	Weather + Disease
9d	A male-skewed sex ratio (Hypothesis 5) is increasing the rate of female dispersal out of Zealandia (Hypotheses 2 and 3), thereby reducing female survival.	Sex Ratio + Dispersal
9e	The current habitat conditions (Hypothesis 8) are such that birds are dispersing out of Zealandia to find food (Hypotheses 2), reducing adult and fledgling survival.	Habitat + Dispersal
9f	Current habitat conditions and poor nutrition (Hypothesis 8) are increasing Hihi's susceptibility to disease (Hypothesis 7), reducing survival.	Habitat + Disease
10	Hihi get killed from hitting fences, reducing survival.	Fence Mortality
11	Hihi chicks are being fed wasps, causing internal trauma from stingers and leading to death, reducing overall chick and fledgling survival.	Wasp Ingestion
12	Hihi are consuming poisoned baits, either through primary or secondary poisoning, causing reduced adult survival.	Poison Baits
13	Hihi are being caught in mammalian traps and other control tools, reducing adult survival.	Mammal Control
14	Wasps are limiting nectar and insects, reducing survival.	Wasp Competition
15	Competition with mice for insects and seeds is reducing survival.	Mice Competition
16	Inter and intraspecific competition for supplemental feeding resources is reducing female survival.	Inter/intraspecific Competition
17	Hihi rearing is phenologically asynchronous with invertebrate prey availability, leading to poor survival.	Phenology

Table 4.3 Table of alternative management actions that may be promising under various alternative hypotheses. See Table 4.2 for hypothesis descriptions.

Alternative hypothesis number	Alternative hypothesis name	Alternative management actions
1	Native Species Predation	Alter nest box design, install protective structures, alter placement of nest boxes
2, 3	Dispersal Predation and Dispersal	Reduce feeding outside of reserve, increase non-native predator control around reserve
4	Inbreeding	Transfer birds from wild or other reintroduced populations, exchange birds between populations
5	Sex Ratio	Translocate excess males, exclude males from feeder locations, translocate additional females
6	Weather	Design nest boxes that protect against cold temperatures
7	Disease	Increase hygiene at feeders and nest boxes
8	Habitat	Plant a variety of food sources, supplement with additional food types, wait for forest to mature
10	Fence Mortality	Alter design of fence hood
11, 14	Wasp Ingestion, Wasp Competition	Active search and destroy of wasps, change management of supplemental food, change supplemental feeder style
12	Poison Baits	Minimize time bait is available
13	Mammal Control	Change trap management, reduce trap entrance sizes, change trap locations, change trigger pressure
15	Mice Competition	Provide supplemental insect and seeds, increased mouse control
16	Inter/intraspecific Competition	Alter supplemental feeder design or placement
17	Phenology	Provide supplemental insect prey
9a, 9b, 9c, 9d, 9e, 9f	Combination hypotheses (see Table 4.2)	Any of the above actions depending on specific interaction
General alternative		Continue supportive management and supplement the population as needed
General alternative		No more supplemental releases
General alternative		Discontinue supplemental feeding
General alternative		Translocate excess males to learn about potential reintroduction sites for Hihi populations
General alternative		Translocate all Hihi to another site, i.e., terminate Zealandia's Hihi program

Table 4.4 Scoring rubric for *magnitude of uncertainty*, *relevance (a), relevance (b),* and *reducibility* for Zealandia Hihi constructed value of information (CVOI) expert elicitation. Note than *relevance* is calculated as *relevance (a)* multiplied by *relevance (b)* then rescaled to 0 – 4. CVOI is calculated as the product of *magnitude of uncertainty* and *relevance*.

Score	*magnitude of uncertainty*	*relevance (a)*	*relevance (b)*	*reducibility*
0	There is little to no uncertainty regarding the degree of impact this hypothesis is having on the population, based on current knowledge, theory, personal experience, or other rationale.	Even if true, this hypothesis has no to very little impact on the population.	Management actions will have no to very little impact on the hypothesized effect.	The uncertainty about this hypothesis is unlikely to be reduced within a relevant management timeframe due to the expected cost, logistics, and ability to obtain necessary data.
1	There is low to moderate uncertainty regarding the degree of impact this hypothesis is having on the population.	If true, this hypothesis has a small to moderate impact on the population, with an impact that is approximately 25% compared to the total range of variability in population trend.	Management actions will reduce the hypothesized effect on the population by approximately 25%.	The uncertainty about this hypothesis could be reduced by approximately 25% within a relevant management timeframe based on the expected cost, logistics, and ability to obtain necessary data.
2	There is moderate uncertainty regarding the degree of impact this hypothesis is having on the population.	If true, this hypothesis has a moderate impact on the population, with an impact that is approximately 50% compared to the total range of variability in population trend.	Management actions will reduce the hypothesized effect on the population by approximately 50%.	The uncertainty about this hypothesis could be reduced by approximately 50% within a relevant management timeframe based on the expected cost, logistics, and ability to obtain necessary data.
3	There is moderate to high uncertainty regarding the degree of impact this hypothesis is having on the population.	If true, this hypothesis has a moderate to large impact on the population, with an impact that is approximately 75% compared to the total range of variability in population trend.	Management actions will reduce the hypothesized effect on the population by approximately 75%.	The uncertainty about this hypothesis could be reduced by approximately 75% within a relevant management timeframe based on the expected cost, logistics, and ability to obtain necessary data.
4	There is little to no information currently available to discern the degree of impact this hypothesis is having on the population.	If true, this hypothesis has a large impact on the population, with an impact as large as the total range of variability in population trend.	Management actions will mostly or completely reduce the hypothesized effect.	The uncertainty about this hypothesis could be mostly or completely reduced within a relevant management timeframe based on the expected cost, logistics, and ability to obtain necessary data.

Table 4.5 Mean and estimated standard error (SE) for each elicited component of constructed value of information (CVOI). Mean values were calculated across experts (n = 9) for each hypothesis (n = 22) using the second round of expert responses. SE values were estimated through bootstrapping. See Table 4.2 for hypothesis descriptions. Hypotheses with the highest mean for each component are shown in bold.

Hypothesis Number	Hypothesis Name	*magnitude of uncertainty*	*relevance (a)*	*relevance (b)*	*relevance*	CVOI	*reducibility*
1	Native Species Predation	2.00 (0.35)	0.78 (0.26)	0.11 (0.11)	0.06 (0.02)	0.22 (0.05)	2.00 (0.43)
2	Dispersal Predation	1.78 (0.21)	**2.67 (0.23)**	1.11 (0.19)	0.78 (0.14)	1.39 (0.30)	2.33 (0.34)
3	Dispersal	2.00 (0.32)	2.22 (0.31)	0.67 (0.16)	0.36 (0.10)	0.61 (0.24)	2.22 (0.31)
4	Inbreeding	1.89 (0.26)	1.33 (0.16)	2.00 (0.27)	0.67 (0.12)	1.31 (0.29)	**2.67 (0.44)**
5	Sex Ratio	2.00 (0.35)	1.89 (0.37)	**2.56 (0.32)**	**1.39 (0.28)**	2.11 (0.71)	2.00 (0.35)
6	Weather	1.78 (0.31)	1.67 (0.23)	1.22 (0.34)	0.58 (0.16)	0.83 (0.33)	2.11 (0.32)
7	Disease	2.56 (0.22)	1.89 (0.30)	1.33 (0.22)	0.72 (0.14)	1.75 (0.39)	1.44 (0.36)
8	Habitat	2.56 (0.28)	1.56 (0.32)	1.44 (0.32)	0.61 (0.17)	1.58 (0.45)	1.22 (0.14)
9a	Weather + Sex ratio	2.78 (0.32)	1.22 (0.21)	1.11 (0.24)	0.28 (0.10)	0.86 (0.28)	1.33 (0.22)
9b	Inbreeding + Disease	2.78 (0.26)	1.44 (0.23)	1.78 (0.34)	0.75 (0.16)	2.22 (0.47)	0.78 (0.14)
9c	Weather + Disease	**3.00 (0.15)**	1.33 (0.27)	0.56 (0.23)	0.22 (0.08)	0.75 (0.25)	0.44 (0.17)
9d	Sex Ratio + Dispersal	**3.00 (0.22)**	1.67 (0.28)	2.00 (0.39)	1.00 (0.22)	**2.67 (0.68)**	1.00 (0.28)
9e	Habitat + Dispersal	2.78 (0.27)	2.11 (0.33)	1.00 (0.22)	0.64 (0.14)	1.89 (0.42)	1.22 (0.21)
9f	Habitat + Disease	2.67 (0.28)	1.78 (0.34)	1.11 (0.34)	0.58 (0.18)	1.67 (0.49)	0.89 (0.20)
10	Fence Mortality	0.33 (0.15)	0.11 (0.11)	0.22 (0.21)	0.00 (0.01)	0.00 (0.00)	2.22 (0.57)
11	Wasp Ingestion	1.56 (0.22)	0.89 (0.11)	1.78 (0.38)	0.39 (0.10)	0.53 (0.18)	2.00 (0.35)
12	Poison Baits	1.33 (0.36)	0.44 (0.16)	1.33 (0.30)	0.19 (0.06)	0.36 (0.10)	1.78 (0.37)
13	Mammal Control	0.33 (0.16)	0.11 (0.10)	1.67 (0.36)	0.08 (0.04)	0.00 (0.02)	2.56 (0.46)
14	Wasp Competition	2.11 (0.25)	1.56 (0.27)	2.11 (0.25)	0.94 (0.17)	1.92 (0.42)	1.22 (0.21)
15	Mice Competition	2.11 (0.25)	0.89 (0.24)	2.11 (0.33)	0.53 (0.15)	1.14 (0.35)	1.11 (0.25)
16	Inter/intraspecific Competition	2.22 (0.21)	1.22 (0.21)	0.89 (0.30)	0.36 (0.10)	0.89 (0.24)	1.33 (0.22)
17	Phenology	2.67 (0.32)	1.56 (0.24)	0.56 (0.32)	0.22 (0.13)	0.64 (0.35)	1.11 (0.25)

Table 4.6 Percent of 1000 bootstrapped samples that fell into each priority quadrant for each hypothesis, i.e., highest priority, high priority, medium priority, and low priority. The 4 priority rankings were determined based on the median of the CVOI and *reducibility* means across the hypotheses. The hypotheses that fell into the highest priority quadrant scored above the median on both CVOI and *reducibility*, whereas those in the low priority quadrant scored below the median on both CVOI and *reducibility*. Hypotheses in the high priority quadrant scored above the median on CVOI but below in *reducibility* and hypotheses in the medium priority quadrant scored above the median on *reducibility* but below on CVOI. The median value of CVOI means was 1.01 and the median value of the *reducibility* means was 1.39. Percentages that are bold indicate the priority ranking based on the mean. See Table 4.2 for hypothesis descriptions.

Hypothesis Number	Hypothesis Name	Highest Priority	High Priority	Medium Priority	Low Priority
1	Native Species Predation	0.00%	0.00%	**93.50%**	6.50%
2	Dispersal Predation	**85.80%**	0.10%	14.10%	0.00%
3	Dispersal	13.70%	0.10%	**86.10%**	0.10%
4	Inbreeding	**78.50%**	0.40%	20.80%	0.30%
5	Sex Ratio	**93.90%**	5.20%	0.80%	0.10%
6	Weather	35.80%	0.40%	**62.60%**	1.20%
7	Disease	**53.10%**	42.00%	3.40%	1.50%
8	Habitat	9.80%	**73.00%**	1.30%	15.90%
9a	Weather + Sex ratio	15.10%	20.90%	27.80%	**36.20%**
9b	Inbreeding + Disease	0.00%	**97.80%**	0.00%	2.20%
9c	Weather + Disease	0.00%	5.20%	0.00%	**94.80%**
9d	Sex Ratio + Dispersal	7.20%	**92.20%**	0.00%	0.60%
9e	Habitat + Dispersal	16.60%	**70.10%**	3.60%	9.70%
9f	Habitat + Disease	0.30%	**73.60%**	0.00%	26.10%
10	Fence Mortality	0.00%	0.00%	**92.80%**	7.20%
11	Wasp Ingestion	2.10%	0.10%	**92.80%**	5.00%
12	Poison Baits	0.00%	0.00%	**85.40%**	14.60%
13	Mammal Control	0.00%	0.00%	**98.90%**	1.10%
14	Wasp Competition	21.00%	**77.00%**	0.50%	1.50%
15	Mice Competition	4.60%	**41.20%**	7.40%	46.80%
16	Inter/intraspecific Competition	1.80%	3.00%	40.00%	**55.20%**
17	Phenology	1.80%	9.70%	9.40%	**79.10%**

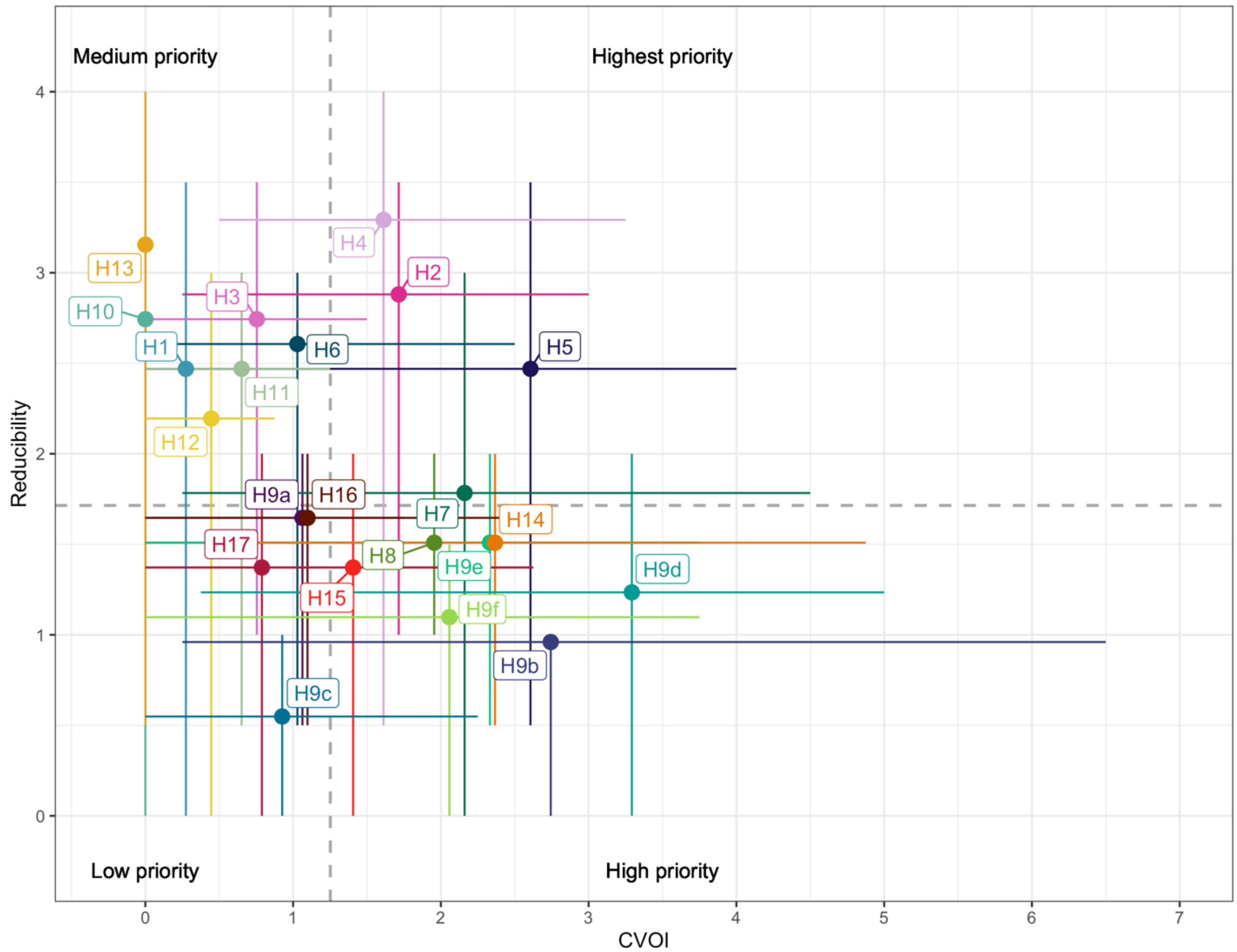

Figure 4.1 Mean constructed value of information (CVOI; *x*-axis) and *reducibility* (*y*-axis) values across experts ($n = 9$) for 22 alternate hypotheses about what may be causing the stagnant population growth of Hihi at Zealandia Wildlife Sanctuary. Dots for each hypothesis are means across experts, vertical bars are the 90% confidence intervals for *reducibility*, and horizontal bars are the 90% confidence intervals for CVOI. Confidence intervals were obtained from the elicited values. Horizontal and vertical gray dashed lines are the median values across all hypotheses for CVOI and *reducibility*. The gray dashed lines, medians, divide the plot into 4 priority quadrants based on the median values for CVOI and *reducibility*: highest priority, high priority, medium priority, and low priority. The hypotheses that fell into the highest priority quadrant scored high in both CVOI and *reducibility*, whereas those in the low priority quadrant scored low in both CVOI and *reducibility*. Hypotheses in the high priority quadrant scored high in CVOI but low in *reducibility* and hypotheses in the medium priority quadrant scored high in *reducibility* but low in CVOI. See Table 4.2 for hypothesis names and descriptions.

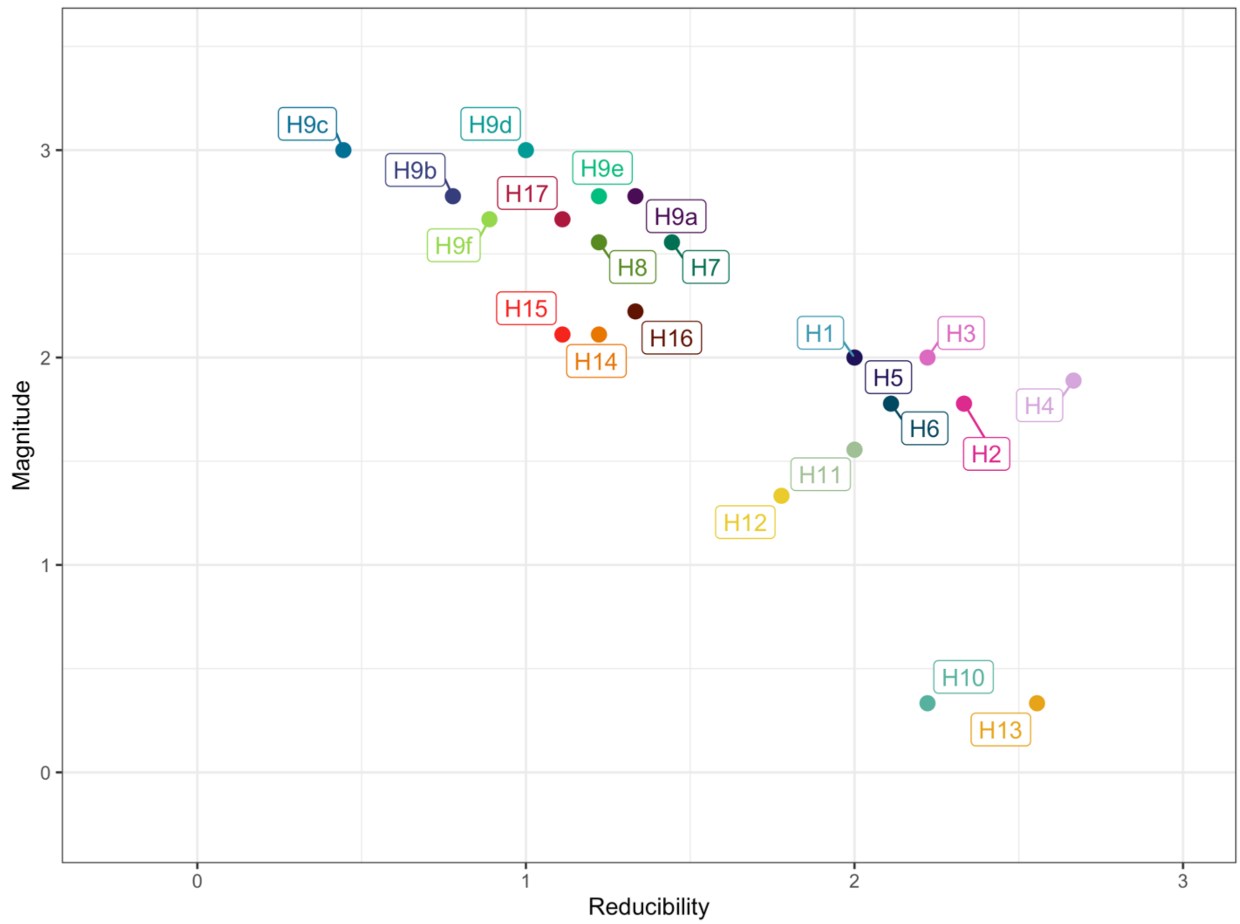

Figure 4.2. Mean *reducibility* (x-axis) and *magnitude of uncertainty* (y-axis) values across experts (n = 9) for 22 alternate hypotheses postulating the cause of stagnant population growth of Hihi at Zealandia. Dots for each hypothesis are the means across experts. See Table 4.2 for hypothesis names and descriptions.

Chapter 5. Reintroduction Strategy Development and Evaluation for a Threatened Grassland Passerine using Decision Analysis

Publication history: This study was co-authored with Gary L. Slater, Abby E. Bratt, Scott F. Pearson, and Sarah J. Converse. At the time this dissertation was published, this chapter was not in review with a journal.

5.1 Abstract

Conservation translocations, including reintroduction, are increasingly common management actions used to conserve species and restore ecosystem function. Determining the best way to reintroduce a species is challenging due to the complexity of management alternatives and the uncertain outcomes of novel management actions. Decision-analytic methods applied to reintroduction problems can help overcome these challenges, offering decision makers a framework for improving performance on management objectives. We engaged in a decision-analytic process to develop a framework for reintroduction of a threatened grassland passerine, the Streaked Horned Lark (SHLA; *Eremophila alpestris strigata*), in Washington State, USA. Here, reintroduction has the potential to expand breeding sites for a migratory metapopulation that exists in a highly fragmented landscape. Our goal was to develop a framework that would be useful for making decisions about reintroduction efforts, and that could be adapted as needed throughout the reintroduction planning process. We developed the framework in a collaborative process involving decision makers, stakeholders, and scientific experts. We generated a variety of management alternatives that captured options throughout the entire reintroduction process, including the source of SHLA, the reintroduction site, the reintroduction method, and post-release management actions. To evaluate these actions in terms of their ability to meet species conservation goals, we developed a metapopulation model. To parameterize the model, we

elicited expert judgements about demographic parameters under the different reintroduction strategies. Expert judgments were combined with empirical demographic information to predict site-level abundance and persistence. Predicted reintroduction site persistence and abundance were highest for strategies involving adult breeding pairs released with their dependent young and strategies involving larger numbers of releases. Existing breeding sites were minimally impacted by serving as a source for reintroductions. Our framework offers insight about the development and evaluation of reintroduction strategies for metapopulations in highly fragmented landscapes.

5.2 Introduction

Conservation translocations, including reintroductions, reinforcements, and conservation introductions, are a class of conservation action that is increasingly used to conserve threatened species and restore ecological function in degraded ecosystems (IUCN/SSC, 2013; Seddon et al., 2012, 2007). Reintroductions involve intentionally translocating animals to areas in which they historically occurred but from which they have been extirpated (Seddon et al., 2012; Seddon and Armstrong, 2016). Reintroductions tend to be both risky and expensive (Brichieri-Colombi and Moehrenschlager, 2016) and have produced relatively poor outcomes in the past (Griffith et al., 1989; Seddon and Armstrong, 2016). However, the field of reintroduction biology has grown rapidly in the last few decades, resulting in improved knowledge about effective practices (Moehrenschlager and Lloyd, 2016) and more robust approaches for grappling with the complexities of these decisions (Converse et al., 2013a; Moehrenschlager and Lloyd, 2016; Parker et al., 2012).

A valuable framework for identifying reintroduction strategies that will best achieve management objectives is decision analysis, or structured decision making (SDM; Gregory et al., 2012; Hemming et al., 2022; Keeney, 1982; Runge et al., 2020). An SDM process can provide decision makers with a clear view of the decision problem and a transparent and defensible justification for the decision (Gregory et al., 2012; Runge et al., 2020). In an SDM process, decisions are deconstructed into their component parts, including (1) a definition of the decision problem, (2) management objectives, (3) management alternatives, (4) models that predict the effects of alternatives in terms of the objectives, and (5) an approach for identifying the preferred alternative (Gregory et al., 2012; Runge et al., 2020). The deconstruction of the decision allows for the analysis of each component separately, facilitating the identification and application of decision-analytic tools to overcome impediments. Reintroduction decisions tend to have particular types of impediments, including multiple potentially competing objectives (Converse et al., 2013b), complex alternatives, and substantial uncertainty.

One of the major challenges in reintroduction decisions is the complexity of the available alternatives. Alternatives are typically built by considering various combinations of actions in different categories, including where to translocate the species to, where to source individuals from, how many individuals to release, the demographic composition of animals to release, the seasonal timing of release, the method of release, and the management actions to apply after release, among others (Canessa, 2015; IUCN/SSC, 2013). Alternatives of this type are often referred to as strategy-based alternatives (Converse and Grant 2019). Simply identifying the set of available strategies may require substantial time and attention.

The predictive modeling step of SDM is intended to allow for the exploration of outcomes under various strategies (Converse et al., 2013a; Converse and Armstrong, 2016). However, modeling is often substantially constrained by uncertainty resulting from incomplete information about species biology, ecology, and response to novel management actions

(Armstrong and Reynolds, 2012; Converse et al., 2013a; Converse and Armstrong, 2016; IUCN/SSC, 2013). Expert elicitation, a formalized approach for obtaining expert judgement in cases where direct empirical data are unavailable, offers a way to overcome data limitations in conservation problems (McBride et al. 2012, Martin et al. 2012a). Consistent with the SDM process, expert elicitation is structured, explicit, and transparent, allowing predictive models to be updated with empirical information once management actions are applied.

Thus, two of the major challenges of reintroduction decisions, the complexity of alternatives and the uncertainty of predictions, can be approached using existing tools. However, a remaining challenge is how to effectively combine these tools when uncertainty exists about many aspects of a strategy. This requires a tractable way of accounting for multiple uncertainties simultaneously in modeling, while not overwhelming the cognitive capacity of experts by requiring them to reply to excessive numbers of elicitation questions.

The Streaked Horned Lark (SHLA; *Eremophila alpestris strigata*) is a ground-nesting grassland passerine and a subspecies of Horned Lark (*Eremophila alpestris*). SHLA are found only in the U.S. states of Washington and Oregon, where they are listed as threatened under the U.S. Endangered Species Act (U.S. Fish and Wildlife Service, 2021). Threats to SHLA include small population genetic effects (e.g., inbreeding depression), skewed sex ratio, predation, human disturbance, and high susceptibility to demographic and environmental stochastic events (U.S. Fish and Wildlife Service, 2021, 2019). Their reliance on sparse grassland habitat for breeding has resulted in SHLA populations establishing breeding territories on human-dominated landscapes, e.g., airports, agricultural fields, and dredge deposit sites. Because of this, SHLA are particularly vulnerable to anthropogenic threats that can impact their survival and reproduction (Anderson and Pearson, 2015). In the south Puget lowlands (SPL) region of Washington State, the species exists in a highly fragmented metapopulation with few breeding sites and limited movement between sites (Bratt, 2023; Pearson et al., 2005a). SHLA recovery goals are unlikely to be met in this region by focusing solely on currently occupied sites because they fail to address the problems arising from small, isolated breeding populations that are at risk of extirpation resulting from stochastic events (U.S. Fish and Wildlife Service, 2019). Therefore, the establishment of additional breeding sites through reintroduction has been identified as a priority for recovery (U.S. Fish and Wildlife Service, 2019).

Here, we develop a decision-analytic framework to inform the selection of a reintroduction strategy for SHLA in the SPL. Through a collaborative group process, we developed a definition of the decision problem and identified a key management objective on which to focus the analysis. We developed a large number of alternative strategies to capture all aspects of the reintroduction process. We built a predictive metapopulation model incorporating empirical estimates of demographic parameters from occupied SHLA sites along with expert judgments regarding new sites and other aspects of the reintroduction process. We demonstrate an approach to combining expert judgments about individual aspects of strategies to develop predictions for complete reintroduction strategies. Our model predicts the outcome of reintroduction strategies in terms of individual sites, offering information about the overall metapopulation, and accounting for dynamics at sites that are used to source individuals for reintroduction. Finally, we use the results of the predictive model to identify release strategies that appear promising. Our results can support decisions by management agencies regarding reintroduction of SHLA, while providing a template for other reintroduction decisions of this type.

5.3 METHODS

5.3.1 PARTICIPANTS AND WORKSHOP FORMAT

A collaborative workshop process was used for problem framing. Participants invited to the working group included decision makers, stakeholders, and scientific experts, with representatives from Ecostudies Institute, U.S. Fish and Wildlife Service, Washington Department of Fish and Wildlife, Oregon Department of Fish and Wildlife, Joint Base Lewis McChord, Center for Natural Lands Management, U.S. Federal Aviation Administration, and Thurston County Planning Commission. The initial problem structure, i.e., problem definition, objectives, and alternatives, was developed through remote workshop meetings following a rapid prototyping approach (Garrard et al., 2017), in which the SDM steps are progressed through in quick succession, iteratively revisiting each step to refine components as needed.

5.3.2 PROBLEM DEFINITION

Initial discussion of the decision problem was focused on determining the spatial scope of the decision problem. Four regional SHLA breeding metapopulations are recognized. In Washington state, there are three regions where SHLA breeding populations occur: sparsely vegetated dune and sand spit sites along the Pacific coast in southern Washington; islands in the Columbia River; and grassland, agricultural, military, and airport sites in the SPL (Stinson, 2016). In the Willamette Valley of Oregon, a single SHLA breeding population occurs on grassland, agricultural, and airport sites (Anderson and Pearson, 2015). The majority of SHLA across the four metapopulations overwinter in the Willamette Valley or the Washington Coast (Pearson et al., 2005b).

The workshop participants determined that the decision should focus on the SPL. The SPL region is where habitat fragmentation is most severe and is most likely restricting colonization of new sites. A decision framework developed for the SPL also has relevance to reintroduction efforts in other regions. With that in mind, the decision problem was identified as: how should managers undertake reintroductions of SHLA to unoccupied sites within SPL?

5.3.3 Objectives

The working group articulated two fundamental objectives for SHLA reintroduction to SPL: 1) maximize the number of SHLA populations (breeding sites) and 2) minimize cost. These objectives were intended to align with the recovery goals specified in the draft recovery plan for SHLA (U.S. Fish and Wildlife Service, 2019). We recognize minimizing cost as a fundamental objective, which may require further analysis prior to a final decision, but focus here on the objective to maximize the number of SHLA population in the SPL, as working group members saw the prediction of population outcomes as the fundamental impediment to the decision.

5.3.4 Alternatives

We developed a strategy table with the working group and used it to develop reintroduction strategies by selecting elements from each of several strategy components (Gregory et al., 2012). These components included the source (occupied sites in SPL), location of release (unoccupied suitable sites within SPL), number of individuals to release in a year, number of years to perform releases, age and timing of release (e.g., SHLA life stage and season), method of release (e.g., soft vs. hard release), and post-release management actions (e.g., supplemental feeding; Table 5.1). We combined age and timing of release, method of release, and post-release management actions to form sets of initial release alternatives (Table 5.2). Similarly, we combined the number of individuals released in a year and the number of years additional releases occurred to form a set of release size alternatives (Table 5.3). We developed reintroduction strategies as the combination of source site, reintroduction site (the location of release), initial release alternatives, and release size alternatives. Below, we describe each of the components independently, that when combined, make up a reintroduction strategy.

The source sites included in the strategy table were either ‘prairie’ sites or ‘airport’ sites. After further discussion, decision makers expressed an interest in evaluating only those reintroduction strategies where SHLA were sourced from airport sites, since airport sites are associated with higher levels of anthropogenic mortality through airplane strikes and mowing activities (Anderson and Pearson, 2015). We focus the rest of the analysis using only airport sites as the source for SHLA reintroduction.

Five release sites, with the potential to be suitable after restoration, were identified by the working group. These sites included Glacial Heritage, Mima Mounds, Violet Prairie, Scatter Creek, and West Rocky Prairie. Glacial Heritage is comprised of land owned by the Washington Department of Fish and Wildlife (WDFW) and Thurston County. Glacial Heritage is open to the public only one day a year and restoration efforts have been ongoing since 1996 by Center for Natural Lands Management (Washington Department of Fish and Wildlife, 2020). Mima Mounds is a natural preserve area managed by the Washington Department of Natural Resources, with public access restricted to wildlife viewing (Washington Department of Natural Resources, n.d.). Scatter Creek and West Rocky Prairie are wildlife areas managed by WDFW. They offer public recreation opportunities, including hunting and wildlife viewing (Washington Department of Fish and Wildlife, 2020). Violet Prairie was acquired by WDFW in 2022 and currently does not allow use by the public (Washington Department of Fish and Wildlife, n.d.). The potential release sites are mapped in Figure 5.1.

Initial release alternatives were generated by combining the strategy table components of age and timing of release, method of release, and post-release management actions (Table 5.1). Using these components, we developed 12 initial release alternatives (Table 5.2) to evaluate at each of the five potential release sites, using birds sourced from airport sites. Three initial release alternatives involved release of breeding pairs with dependent young (i.e., 5- to 6-day-old nestlings) from their first brood in May or June (identified as 'ADY'; A1 – A3). Four alternatives involved release of breeding pairs during the pre-breeding season in April ('ABP'; A4 – A7). Four alternatives involved release of independent young one to two months post-fledging in August or September ('IND' A8 – A11). The last alternative involved release of adults with independent young in August or September ('IND+AD' A12). The release method for each initial release alternative was either soft or hard release. Soft release involves capturing individuals, moving them to the release site, and keeping them in pens for two weeks prior to release. Hard release involves capturing individuals, transporting them to the release site, keeping them in an enclosure for the first night, and then releasing them the next day. Post-release management activities included supplemental food following release, placing SHLA decoys around the release site and playing conspecific calls, or avian predator management. See Table 5.2 for additional details.

To evaluate how release size and additional reinforcement releases would impact outcomes, we developed nine release size alternatives. For each initial release alternative and potential release site combination, we considered initial release abundances of 12, 20, or 30 SHLA, assuming equal numbers of males and females. Three release size alternatives did not include additional releases after the initial release and only altered the initial release abundance ('None' combinations, Table 5.3). The other release size alternatives considered supplemental reinforcement where 10 additional SHLA were released in years 2, 3, or 4, with the number of releases depending on the initial release abundance. Release size alternatives where additional releases occurred also included nest supplementation after the reintroduced population was established. Nest supplementation involved supplementing active nests with either nestlings or eggs from source sites ('SUPP N' or 'SUPP E', respectively). See Table 5.3 for details.

5.3.5 Predictive model

Bratt (2023) developed a two-sex, age-structured, multi-site integrated population model (IPM) to estimate demographic rates and abundance at occupied SPL sites, and we used the structure and parameter estimates from this model as the basis of the predictive model for evaluating reintroduction strategies. The pre-breeding census model included two age classes: one-year-olds, N_1, that hatched in the previous year, and adults two years or older, N_{ad}. We modeled the abundance at each year t for first-year SHLA at occupied site s, $N_{1,s}$, as:

$$N_{1,s,t} \sim Poisson\left(f_s * (N_{1,s,t-1} + N_{ad,s,t-1}) * \varphi_{1,s,t}\right) \tag{1}$$

where the number of first-year SHLA in the current year at site s is modeled as a function of the number of fledglings produced at time t, i.e., $f_s * (N_{1,s,t-1} + N_{ad,s,t-1})$, that survive to 1-year of age with time- and site-specific probability $\varphi_{1,s,t}$. We modeled the number of adult SHLA in each year t at occupied site s, $N_{ad,s,t}$, as:

$$N_{ad,s,t} \sim Binomial\left(N_{1,s,t-1} + N_{ad,s,t-1}, \varphi_{ad,s,t}\right) \tag{2}$$

where the number of adults in the current year at occupied sites, $N_{ad,s,t}$, is dependent on the abundance of SHLA that survived with time- and site-specific probability $\varphi_{ad,s,t}$. We projected the predicted dispersal of individuals between occupied sites in a given year using the movement process described in (Bratt, 2023).

We used the demographic prediction model described above to simulate abundance at each occupied SPL site. Individuals were sourced from occupied airport sites for reintroduction to a single reintroduction site. We assumed that individuals would be sourced from the airport site with the largest abundance of the given release strategy's age class, and within each simulation, we tracked which airport site SHLA were sourced from.

We used an individual-based model (IBM) to simulate outcomes for released individuals during the year of release, thereby combining features of both an IPM and an IBM for prediction (Petracca et al., 2024). The year after translocation, reintroduced individuals' fates were compiled by age class and included in the overall age- and site-structured population model using the form of Eqs. 2 and 3. Temporal stochasticity was included by following the methods described in McGowan et al. (2011). Under all initial release alternatives, released individuals progressed through various stages during their initial year at the reintroduction site: first month survival, second month survival, and dispersal. For adults released early in the breeding season (ADY A1 – A3 and ABP A4 – A7), we randomly assigned the order of breeding and dispersal after adults survived the second month, i.e., whether reintroduced individuals would 1) attempt to breed, then potentially disperse or 2) potentially disperse, then attempt to breed. If additional releases occurred, we used the same process as the initial release. Figure 5.2 (a) provides an illustration of the temporal sequence of events for occupied SPL sites and the reintroduction site. Figure 5.2 (b) and (c) illustrate the various stages individuals progressed through from the time of release until the following year in the IBM.

Translocations have been documented to result in post-release effects, in which survival or reproduction are reduced relative to baseline values for days to months post-release. To account for these effects, for each stage in the IBM, we modeled the outcome of survival as a

Bernoulli distribution with an age- and site-specific survival probability multiplied by a proportional reduction in survival in the first or second month. Take for example, an adult individual released during the pre-breeding season. We modeled the survival of individual adults in their first month after release as:

$$n_{ad,i} \sim Bernoulli(\varphi_{ad,r} * \psi_{ad,1}) \quad 3$$

where $n_{ad,i}$, the individual adult, survives their first month with probability $\varphi_{ad,r} * \psi_{ad,1}$. The site-specific survival probability $\varphi_{ad,r}$ is the probability of survival of adults at reintroduction site r and $\psi_{ad,1}$ is the proportional reduction of survival during the first month post-release given a release strategy. We allowed post-release effects on survival to extend for up to two months, using the same structure, but a different proportional reduction, $\psi_{ad,2}$ in the second month.

We modeled the number of offspring produced by pairs using a Poisson with a rate equal to the reintroduction site-specific fecundity, f_r, multiplied by the proportional reduction of fecundity for a given initial release alternative. Breeding during the year of release was only modeled if the pair were both alive and present at the reintroduction site. Offspring were included as individuals in the IBM and we predicted their survival and dispersal (Figure 5.2 (b)). We modeled whether individuals dispersed away from the reintroduction site as Bernoulli-distributed random variable with age- and timing-specific dispersal probabilities for a given initial release alternative.

For SUPP N and SUPP E, the nest supplementation component in release size alternatives, we simulated the number of eggs or nestlings produced at source sites and translocated up to four per pair to breeding pairs at the reintroduction site. Clutches were constrained to remain at or below a maximum of four and the number of breeding aged adults dictated the number of cross-fosters possible in each simulation.

We included density dependence in our model by reducing fecundity to 0 if the abundance at a site was greater than or equal to the site-specific carrying capacity. We approximated carrying capacity, K, at each site s as

$$K_s = \frac{Area_s * Proportion\ Suitable * Proportion\ Overlap}{Territory\ Size} \qquad 5$$

where $Area_s$ is the area of site s, $Proportion\ Suitable$ is the average proportion of suitable habitat at any given site, $Proportion\ Overlap$ is the average proportion of SHLA breeding territory that overlaps with other SHLA breeding territories, and $Territory\ Size$ is the average breeding territory size in SPL.

5.3.6 Expert elicitation

Empirical data about demographic rates were available for nine occupied sites in SPL. We employed expert elicitation to develop predictions for reintroduction sites and under initial release alternatives. Expert elicitation took place in two phases. The first phase focused on eliciting long-term demographic parameters at a given reintroduction site, e.g., the values reflecting site-specific parameter rates after establishment. The second phase focused on eliciting post-release effects on survival, fecundity, and dispersal under each initial release alternative. Experts were identified and invited to participate in the elicitation based on their knowledge of SHLA populations and habitat, general knowledge about Washington's SPL sites, and management experience with SHLA. Experts that participated in the elicitation are listed in Table A5.

For both phases of elicitation, we used a modified Delphi process (MacMillan and Marshall, 2006) and elicited 4-point parameter estimates (Speirs-Bridge et al., 2010) in which experts are asked to provide their best judgement, a minimum value, a maximum value, and their confidence level that the true value falls between the minimum and maximum values (Speirs-Bridge et al., 2010). We used a combination of in-person and remote meetings to conduct elicitations. In both phases, we onboarded the experts to the process, worked through a practice question, provided instructions, and discussed the parameters we were eliciting (Hanea et al., 2017; Hemming et al., 2018). For remote meetings, we provided experts with a fillable spreadsheet to report their values and experts conducted each round of elicitation offline and independently. Following the modified Delphi process, we held discussions between each round of elicitation. We conducted two rounds of elicitation and used the values from the second round in the model described above.

During the first phase of expert elicitation, we elicited survival and fecundity parameters at each potential reintroduction site assuming an established population, i.e., in the absence of post-release effects. To familiarize the experts with the potential release sites, the experts and facilitators (SJC and HAS) visited an occupied SHLA site and each of the five potential reintroduction sites as a group. Shortly after the site visits, we held an in-person elicitation session where we asked experts to provide survival and fecundity values at potential reintroduction sites. We also elicited parameters required to approximate carrying capacity (Eq. 5) at any SPL site. Direct information is available for the area of each site and empirical information has been collected about the average breeding territory size in Anderson and Pearson (2015). Given that, experts were asked to provide values for $Proportion\ Overlap$ and

$Proportion\ Suitable$ parameters (see Table 5.4 for the parameters elicited and their definitions).

During the second phase of elicitation, we asked experts to provide values for the short-term post-release effects under each initial release alternative (Table 5.2). We asked experts to provide values reflecting the proportional reduction from establishment level age-specific survival, e.g., the proportional reduction in adult survival during the first month, $\psi_{ad,1}$, from Eq 3 above. We elicited the proportional reduction in establishment level fecundity during the year of release and in the following year. To capture whether individuals would be likely to disperse away from the reintroduction site after release, we elicited the age-specific probability of dispersal. Lastly, we elicited the age-specific probability that SHLA would return to the reintroduction site in the year after release. See Table 5.4 for a list of parameters that were elicited.

For each expert and parameter, we estimated parameters of beta or gamma distributions to represent each expert's belief. Beta or gamma parameters were estimated by minimizing the sum of square errors of the quantiles and best values by experts. We modeled all parameters that had support over $(0,1)$, e.g., survival probabilities, using a beta distribution and we modeled all parameters with support from 0 to greater than 1, e.g., fecundity rates, using a gamma distribution. All distributions were converted to a 99% confidence interval, to standardize distributions across experts. We combined distributions to create a mixture distribution for each parameter, with each expert's distribution contributing equally to the mixture distribution (see Chapter 3 for the general form of the mixture distributions). We used the mixture distributions of parameters from both phases of elicitation to parameterize predictive models.

5.3.7 Model implementation

We simulated the SPL metapopulation with a single reintroduced site under each of the reintroduction strategies (i.e., reintroduction site, initial release alternative, and release size alternative; Tables 5.2 and 5.3). For each reintroduction strategy, we projected the population forward 21 years. For each year in each simulation, we drew 15,000 samples from the parameter-specific mixture distributions and drew 15,000 posterior estimates from the IPM model output. All code was written and implemented in R software (R Core Team, 2022). Code and expert elicitation facilitation materials can be found on GitHub (https://github.com/sipeha/Dissertation_Chapter_5).

5.3.8 Assessment

The predictive model outputs for the reintroduced site and the source site were visualized by plotting the mean abundance and the 95% quantiles around the mean over the 21-year timeframe. We evaluated the uncertainty associated with each strategy and considered the risk of poor outcomes by identifying reintroduction strategies that resulted in 95% quantiles that overlapped 0 for reintroduction sites and source sites. The probability of persistence was calculated by summing the number of samples where the abundance of adults was greater than 0 in a year and dividing it by the total number of samples.

5.4 Results

Here, we present the results of the predictive model simulations for SHLA reintroduction strategies in SPL. We report the probability of persistence and estimated abundance by reintroduction site, age of release, release type, post-release management actions, and release size alternative. All reintroduction strategies resulted in 95% quantiles around the mean adult abundance that overlapped 0 (Figures 5.4 – 5.6). There were only minor differences between release size strategies that included nest supplementation, i.e., SUPP N or E (Table 5.3), and here we focus just on the number of individuals released under a release size alternative. Lastly, we report on the estimated abundance at the occupied SPL sites, comparing the outcomes when individuals are sourced for reintroduction to the outcome under the Status Quo strategy.

The reintroduction site with the highest predicted probability of persistence was Glacial Heritage and the site with the lowest was West Rocky (Figure 5.3). Similarly, Glacial Heritage was predicted to have the highest mean adult abundance at the end of the timeframe and West Rocky was predicted to have the lowest (Figures 5.4 – 5.6). Initial release alternatives that involved releasing adult breeding pairs with dependent young (ADY A1 – A3) resulted in the highest predicted probability of persistence (Figure 5.3). Similarly, the mean adult abundance was predicted to be highest under ADY A1 – A3 alternatives (Figures 5.4 – 5.6). Initial release alternatives that involved releasing adult breeding pairs during the pre-breeding season (ABP A4 – A7) resulted in slightly lower mean adult abundance and probability of persistence than ADY A1 – A3 (Figures 5.3 – 5.6). The lowest predicted probability of persistence resulted from initial release alternatives that involved releasing independent young (IND A8 – A11; Figure 5.3). Additionally, IND A8 – A11 resulted in the lowest mean adult abundance across all initial release alternatives (Figure 5.4 – 5.6).

Initial release alternatives that involved soft release generally resulted in higher predicted probability of persistence (Figure 5.3) and slightly higher mean abundance than hard release alternatives (Figures 5.4 – 5.6). Post-release management actions resulted in slightly higher predicted probability of release than when none were included in an initial release alternative (Figure 5.3).

The predicted probability of persistence was highest if release size alternatives included reinforcement releases after the initial release, regardless of the initial release abundance. If reinforcement releases did not occur in a release size alternative (‘None’ combinations, Table 5.3), larger initial release abundances resulted in a higher predicted probability of persistence. For release size alternatives where no additional releases occurred, the mean predicted abundance at reintroduction sites increased with increasing initial abundance. When release size alternatives included additional releases, the mean abundance was higher than without additional releases. A smaller initial release abundance and more years of reinforcement releases resulted in slightly higher predicted probability of persistence (Figure 5.3). In terms of abundance, there were only minor differences between release size alternatives that included additional releases for initial release alternatives ADY A1 – A3 and ABP A4 – A7 (Figures 5.5 and 5.6).

The predicted abundance at occupied SPL sites was not drastically different at the end of the timeframe with and without reintroduction (Figure 5.7). When only an initial release took place, i.e., no reinforcement releases occurred, the impact on SPL abundance was minimal and trends were similar to the status quo. Reinforcement releases caused the airport abundance projections to decrease temporarily with the sourcing of SHLA for reintroduction, then increase for a few years, prior to eventually leveling off to similar abundance as the status quo. Interestingly, SPL sites that were not sources (i.e., prairie sites) also showed the same temporary fluctuations in abundance as source sites. This temporary region-wide perturbation in abundance was likely due to the movement component in the SPL site projection model responding to loss of individuals from source sites (Figure 5.7). The probability of persistence for SPL sites was approximately 1 under all reintroduction strategies, hence the probability of persistence for the metapopulation was determined by the probability of persistence for the specific reintroduction alternative.

5.5 DISCUSSION

Here, we developed a decision-analytic framework for evaluating reintroduction strategies for SHLA in Washington State. The reintroduction strategies that we generated were comprehensive, including all aspects of the reintroduction process. We developed predictive models that used empirical data from the currently occupied sites and expert judgments for the reintroduction strategies, and our models tracked the impact of reintroduction on the overall SPL population. Our predictive models captured how the release process impacted translocated individuals and explicitly incorporated post-release effects. The predictions offer the chance to identify which strategies appear promising for achieving reintroduction success while maintaining the currently occupied sites in the SPL region. Further, this framework is fully adaptable to evaluate various combinations of strategy components and can be updated with empirical information as new data become available.

Identifying reintroduction strategies that mitigate post-release effects is key for reintroduction success (Armstrong and Seddon, 2008). Here, we found that releasing a larger number of SHLA was important for persistence if no additional reinforcement releases took place (Figures 5.3 and 5.4); however, additional releases resulted in the highest predicted probability of persistence (Figures 5.3). Releasing a large number of individuals is expected to safeguard against post-release effects by ensuring there is an adequate number of individuals that survive and breed (Fischer and Lindenmayer, 2000; Wolf et al., 1998), avoiding small population effects (Armstrong and Wittmer, 2011; Deredec and Courchamp, 2007). We also found that initial release alternatives with soft release and post-release management actions resulted in higher persistence. Using a soft release approach may provide SHLA with a chance to acclimatize to the new habitat and minimize post-release dispersal (Tetzlaff et al., 2019). Our results suggest that the age and timing of release may be the most crucial aspect in determining whether SHLA will persist through the initial post-release period. Hence, through considering a range of reintroduction strategies, we were able to identify which aspects result in persistence at the reintroduction site (Armstrong and Seddon, 2008).

Our results indicated that initial release alternatives involving adult breeding pairs with dependent young (ADY A1 – A3) outperformed all other initial release alternatives. The probability of persistence was high for ADY alternative release strategies when additional reinforcement releases occurred (Figure 5.3). Selecting a site that performs well under various release alternatives would allow for release of SHLA using a variety of alternatives. For example, managers could initially reintroduce SHLA with a combination of ADY and adult breeding pair (ABP A4 – A7) alternatives. In our analysis, one site outperformed the rest: Glacial Heritage. Releases at Glacial Heritage resulted in larger predicted site-level abundance and higher probability of persistence than all other sites.

The management objective we focused on in this framework was to maximize the number of SHLA populations in SPL. Hence, in addition to evaluating reintroduction strategies and whether they resulted in an established population, we were also concerned with evaluating how reintroduction strategies impacted the currently occupied sites. Fortunately, SHLA are a relatively well monitored-species in Washington State and we were able to employ the structure and parameter estimates of a previously developed IPM to predict outcomes of reintroduction in terms of currently occupied sites (Bratt, 2023). Our results suggest that the currently occupied SPL populations would be minimally impacted by serving as sources for reintroductions using the numbers of sourced individuals we considered in our analysis. At this point in reintroduction planning, it appears likely that reintroduction efforts to establish an additional SHLA population will not significantly impact the occupied site populations in SPL.

To parameterize predictive models for reintroduction, we elicited expert judgement. Expert elicitation offers a way to inform initial release attempts during this early stage of framework development (Martin et al., 2012; McBride et al., 2012). Because expert elicitation is explicit and structured, the predictive models in our framework can be updated once more information is available (Ellison, 2004). We elicited demographic parameters at each reintroduction site separately from post-release effects to reduce the cognitive task required of experts. Further, instead of eliciting post-release effects in terms of site-specific demographic parameters, we elicited the proportional reduction of establishment level demographic parameters. Together, the elicited values provided information about entire reintroduction strategies and minimized the number of values we asked experts to provide. Our approach to eliciting individual aspects of strategies allowed us to combine the elicited uncertainty of post-release impacts with that of site-specific demographic parameters.

Reintroduction decisions will be more robust if all forms of decision-relevant uncertainty are considered (Converse et al., 2013a; Regan et al., 2002). Here, we incorporated parametric uncertainty, as well as demographic and temporal stochasticity in our predictive models (Regan et al., 2002). To capture the parametric uncertainty in expert judgements, we approximated statistical distributions based on values provided by experts and used these to generate outcomes (MacMillan and Marshall, 2006; Speirs-Bridge et al., 2010). For occupied SPL sites, we used the full range of empirically estimated parameter values, in the form of posterior distributions, from the IPM developed by Bratt (2023). Demographic stochasticity was included in all aspects of the predictive model to capture the random variations in the population and the impacts of small population size (Lande, 1988). Uncertainty around the predicted abundance under each reintroduction strategy included 0 (Figures 5.4 – 5.6) indicating that there is a meaningful amount of risk under each strategy. Hence, a devoted risk analysis will be required to determine the level of risk decision makers are willing to accept about reductions in the source population or failure of the reintroduced population (Canessa et al., 2020; Runge et al., 2020).

Our framework for evaluating reintroduction strategies is ultimately a work in progress and will require additional refinement prior to a decision about which reintroduction strategy to implement. We assumed that reinforcement releases would use the same release alternative as the initial release, but this need not be the case. Evaluating different combinations of release alternatives may provide valuable insight prior to implementation if some age classes are more difficult to capture than others (e.g., capturing breeding pairs with dependent young may be more technically challenging than capturing breeding pairs). Additionally, the number of individuals to release in a year could be further honed by identifying agency-mandated limits, e.g., through discussion with US Fish and Wildlife Service. The framework we developed here can easily be adjusted to integrate various initial release or release size alternatives. A next step for this decision framework will be considering the objective of minimizing cost and analyzing the tradeoffs between the population objective and the cost objective (Converse et al., 2013b; Gregory et al., 2012).

We developed a framework that can be used to assess various combinations of reintroduction strategies and their outcomes in terms of the SPL metapopulation. We demonstrated an approach for combining expert judgements about individual aspects of alternatives that allowed us to make predictions for complete reintroduction strategies. The work we presented offers decision makers and stakeholders a cognitively tractable framework representing the decision problem. Due to the complexity inherent in reintroduction problems, using SDM to inform initial reintroduction attempts has the potential to improve future outcomes by improving the integration of all available information in a deliberative process. The reintroduction problem we present here shares challenges that are present in many other reintroduction problems, e.g., complex alternative structures, unknown species response to management actions, and limited information. Here we offer an example of how these challenges can be overcome when developing a decision framework for reintroduction of species with high breeding site fidelity and species that are reliant on human-mediated processes to maintain suitable habitat.

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5.8 Tables and Figures

Table 5.1 Strategy table outlining components and actions for Streaked Horned Lark (SHLA) reintroduction in the south Puget lowlands of Washington, USA. Potential actions are organized by categories, when one or more elements from each category can be selected to form alternative strategies.

Category	Source population	Location of release	Number of individuals to release	Number of years to release	Age and timing of release	Method of release	Post-release management
Actions	Airport sites	Mima Mounds	0	1	Adults, pre-breeding season (April)	Hard release[1]	Supplemental food[3]
	Prairie sites	Glacial Heritage	12	2	Adults with dependent young (May – June)	Soft release[2]	Conspecific playbacks
		Violet Prairie	20	3	Independent young (August – September)	Cross-fostering	Decoys
		West Rocky	30	3+	Adults, post-breeding (August – September)		Predator management[4]
		Scatter Creek			Egg supplementation		Combinations of the above
					Nestling supplementation		

[1]Hard release: birds are captured and moved, then spend one night in a pen and are released the next day.
[2]Soft release: birds are captured and moved to the release site where they spend 2 weeks in an enclosure, then they are released.
[3]Supplemental food: food is available after birds are released and is provided in such a way that predator attraction is minimized.
[4]Predator management: after release, effort spent to eliminate predator resources, such as perches, that support avian predators (crows, ravens, raptors).

Table 5.2 Initial release alternatives for Streaked Horned Lark (SHLA) reintroduction to five potential release locations Washington State's south Puget lowland region (Figure 5.1). Initial release alternatives were developed from the alternative release strategy table (Table 5.1) by selecting elements from the categories including age and timing of release, release method, and post-release management. Initial release alternatives are grouped by the age of SHLA that are released. Each of these release strategies is applicable to the release size alternatives in Table 5.3.

Alternative		Age and timing of release	Release method	Post-release management
Adults with dependent young and soft release alternatives (ADY)	Alternative 1	Breeding pairs with dependent young (May – June)	Soft release	None
	Alternative 2	Breeding pairs with dependent young (May – June)	Soft release	Supplemental food
	Alternative 3	Breeding pairs with dependent young (May – June)	Soft release	Predator management
Adults during pre-breeding season and soft release (ABP)	Alternative 4	Breeding pairs, pre-breeding season (April)	Soft release	None
	Alternative 5	Random males and females (forced pairs), pre-breeding season (April)	Soft release	None
	Alternative 6	Breeding pairs, pre-breeding season (April) with a focus on at least 1 in the pair being a 1st year bird	Soft release	None
	Alternative 7	Breeding pairs, pre-breeding season (April)	Soft release	Conspecific calls and decoys
Independent young, 1-2 months post-fledge (IND)	Alternative 8	Independent young, 1-2 months post-fledge (August – September)	Soft release	Supplemental food
	Alternative 9	Independent young, 1-2 months post-fledge (August – September)	Soft release	Conspecific calls and decoys, and supplemental food
	Alternative 10	Independent young, 1-2 months post-fledge (August – September)	Hard release	Predator management
	Alternative 11	Independent young, 1-2 months post-fledge (August – September)	Hard release	Supplemental food
Independent young, 1-2 months post-fledge and adults (IND+AD)	Alternative 12	Independent young, 1-2 months post-fledge, with 20% of the cohort adults (August – September)	Hard release	Conspecific calls and decoys

Table 5.3. Release size alternatives for Streaked Horned Lark reintroduction in Washington State. The release size alternatives were developed from the alternative strategy table (Table 5.1) by selecting elements from the categories of number to initially release, the number of years to release, and the number of individuals to release past the initial year. We also included whether nest supplementation, of nestlings or eggs, would take place in these alternatives. Each of the release size alternatives are applicable to the initial release alternatives in Table 5.2 at the five potential reintroduction sites.

Alternative reinforcement strategy	Number of SHLA to initially release	Number of years to release after initial year	Number of individuals released for reinforcement	Nest supplementation
None	12	None	None	None
12, RI+SUPP N	12	Years 2, 3, and 4	10	SUPP N[1]
12, RI+SUPP E	12	Years 2, 3, and 4	10	SUPP E[2]
None	20	None	None	None
20, RI+SUPP N	20	Years 2 and 4	10	SUPP N[1]
20, RI+SUPP E	20	Years 2 and 4	10	SUPP E[2]
None	30	None	None	None
30, RI+SUPP N	30	Year 3	10	SUPP N[1]
30, RI+SUPP E	30	Year 3	10	SUPP E[2]

[1]SUPP N: Within species cross-fostering with nestlings, after reintroduced population is established.
[2]SUPP E: Within species cross-fostering with eggs, after reintroduced population is established.

Table 5.4 Table of elicited parameters and parameter definitions for Streaked Horned Lark (SHLA) reintroduction in Washington State, USA.

Parameter	Definition
Adult survival	Average annual survival probability of adults (birds that are 1 year or older)
Hatch-year survival	Average annual survival probability of hatch-year birds (birds that are less than 1 year old)
Nest success	Probability that a nest produces one or more fledglings
Nests per pair	The number of nesting attempts per adult per year
Fledglings per nest	The number of fledglings produced per successful nest
Proportion overlap	The average proportion of breeding territory that overlaps with other breeding territories
Proportion suitable	The proportion of suitable habitat on average at any given site

Parameter	Definition	Alternative
Dispersal A	Probability that released adults leave the release site prior to the end of the breeding season or prior to normal migration timing.	A1 – A7, A12
Dispersal B	Probability that released hatch-year birds (eggs, nestlings, or independent young depending on the alternative) leave the release site prior to the end of the breeding season or prior to normal migration timing.	A1-A3, A8-A12
Dispersal C1	Probability that released adults will return to the release site in the next breeding season, given they are alive to return and given they did not stay at the release site until the end of the breeding season.	A1 – A7, A12
Dispersal C2	Probability that released adults will return to the release site in the next breeding season, given they are alive to return and given they did stay at the release site until the end of the breeding season.	A1 – A7, A12
Dispersal D	Probability that released hatch-year birds (eggs, nestlings, or independent young depending on the alternative) will return to the release site in the next breeding season, given they are alive to return	A1 – A3, A8 – A12, SUPP N and E
Adult survival A	Proportional reduction in adult survival in the first month after release if the birds stay at the release site.	A1 – A7, A12
Adult survival B	Proportional reduction in adult survival in the second month after release if the birds stay at the release site.	A1 – A7, A12
Hatch-year survival A	Proportional reduction in hatch-year survival in the first month after release if the birds stay at the release site.	A1 – A3, A8 – A12
Hatch-year survival B	Proportional reduction in hatch-year survival in the second month after release if the birds stay at the release site.	A1 – A3, A8 – A12
Fecundity A	Proportional reduction in fecundity during breeding attempts in the year of release	A4 – A7
Fecundity B	Proportional reduction in fecundity in subsequent breeding attempts during the year of release	A1 – A3
Fecundity C	Proportional reduction in fecundity in the year after the initial release year, i.e., the reduction in fecundity in the next breeding season, if birds are alive and return to the release site	A1 – A12
Fecundity D	Proportional reduction in the probability that foster nestlings or eggs with successfully fledge, given foster parents are established at the release site.	SUPP N and SUPP E
Fecundity E	Proportional change in the fledgling probability per individual pair with foster nestlings or eggs added to the nest (contingent on keeping the clutch within 1 – 6 range)	SUPP N and SUPP E

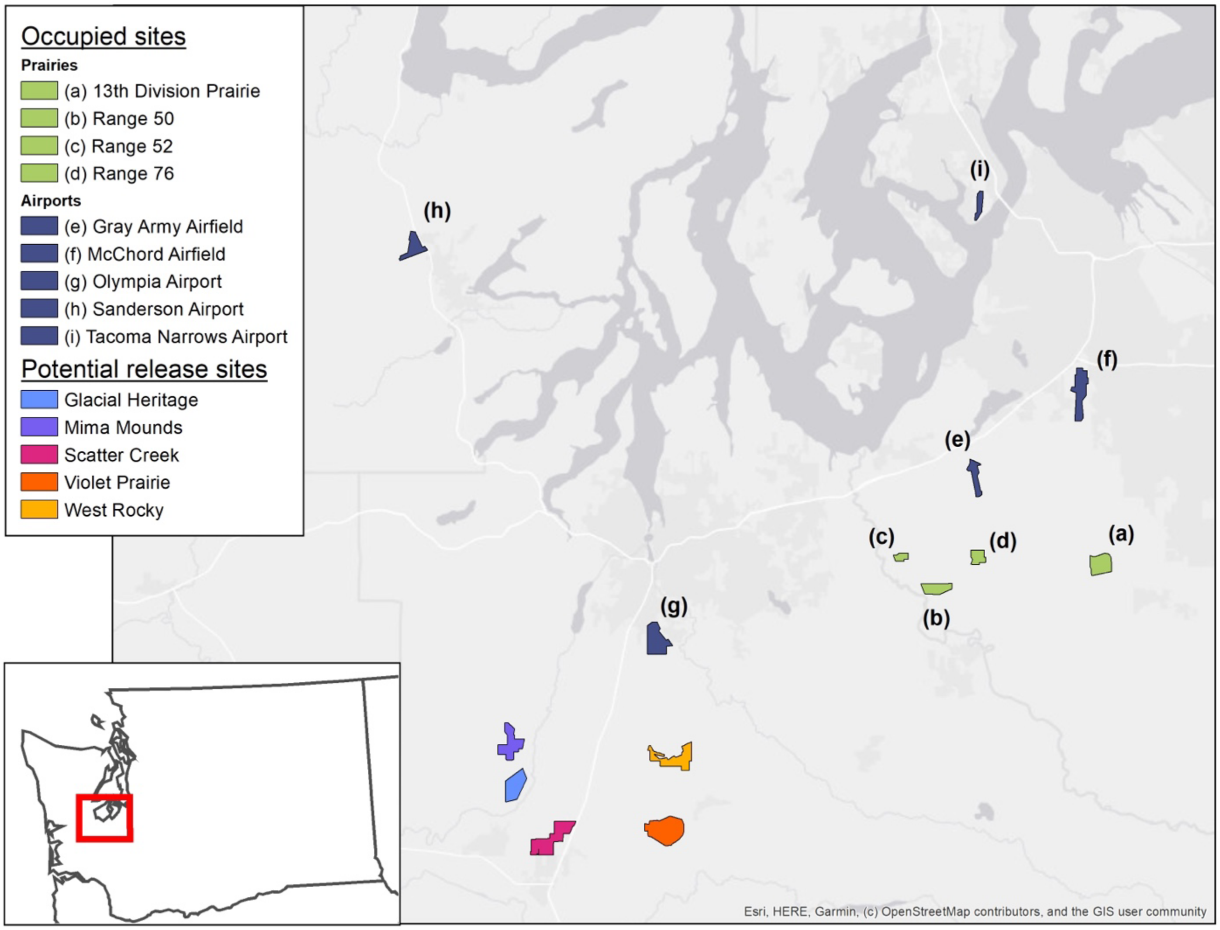

Figure 5.1. Map showing currently occupied sites and potential release sites for Streaked Horned Larks in the south Puget lowland region of Washington State, USA. Occupied sites that are found on airports include (e) Gray Army Airfield, (f) McChord Airfield, (g) Olympia Airport, (h) Sanderson Airport, and (i) Tacoma Narrows Airport. All other occupied sites are prairie sites.

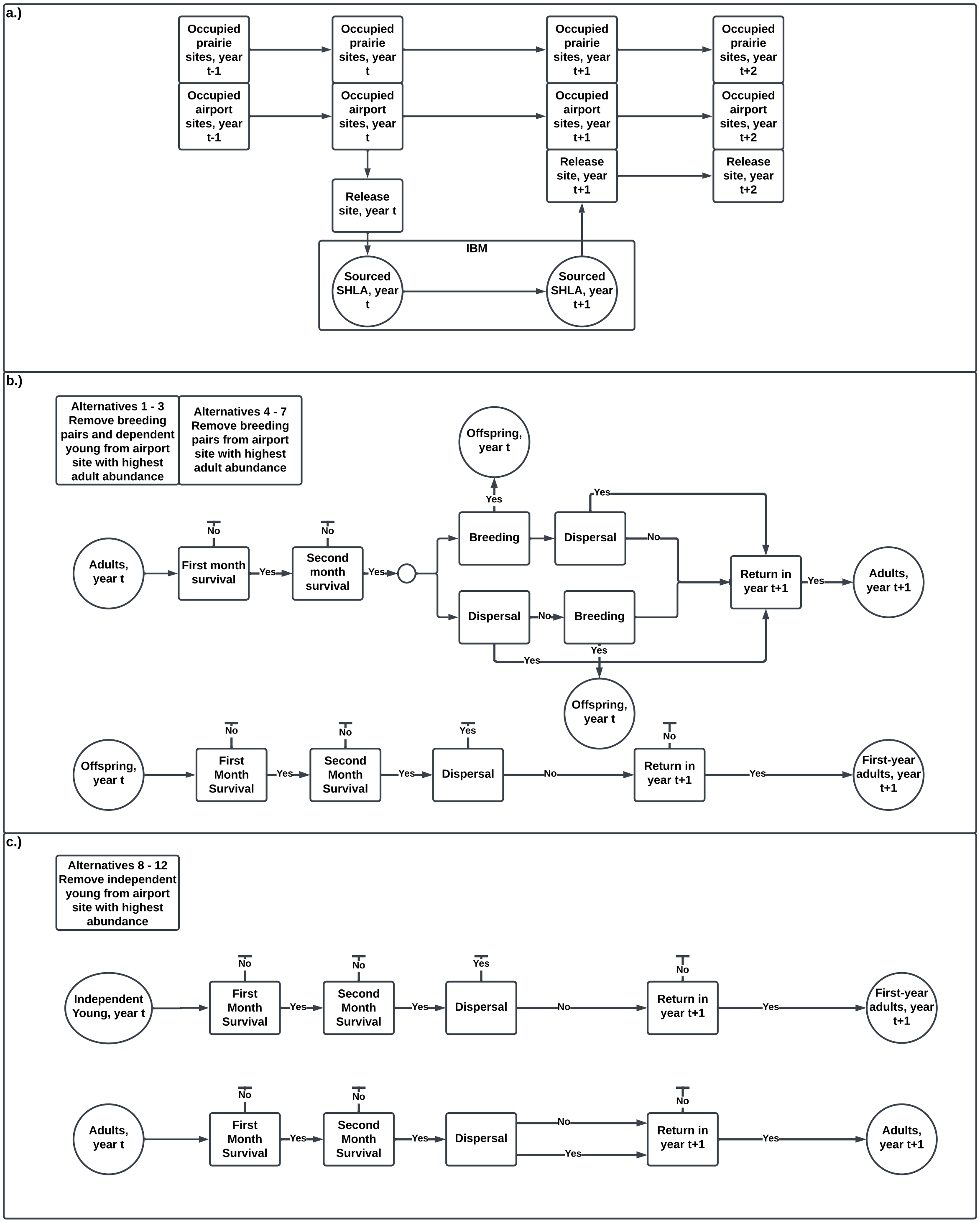

Figure 5.2 Conceptual diagram of population model and various stages that individuals are modeled as going through during their initial year of release and the following year in the predictive model. (a) The basic structure of the overall population model predicts the abundance of SHLA at both prairie and airport sites. (b) and (c) are the various stages that individuals are modeled through in the individual model component given an initial release alternative.

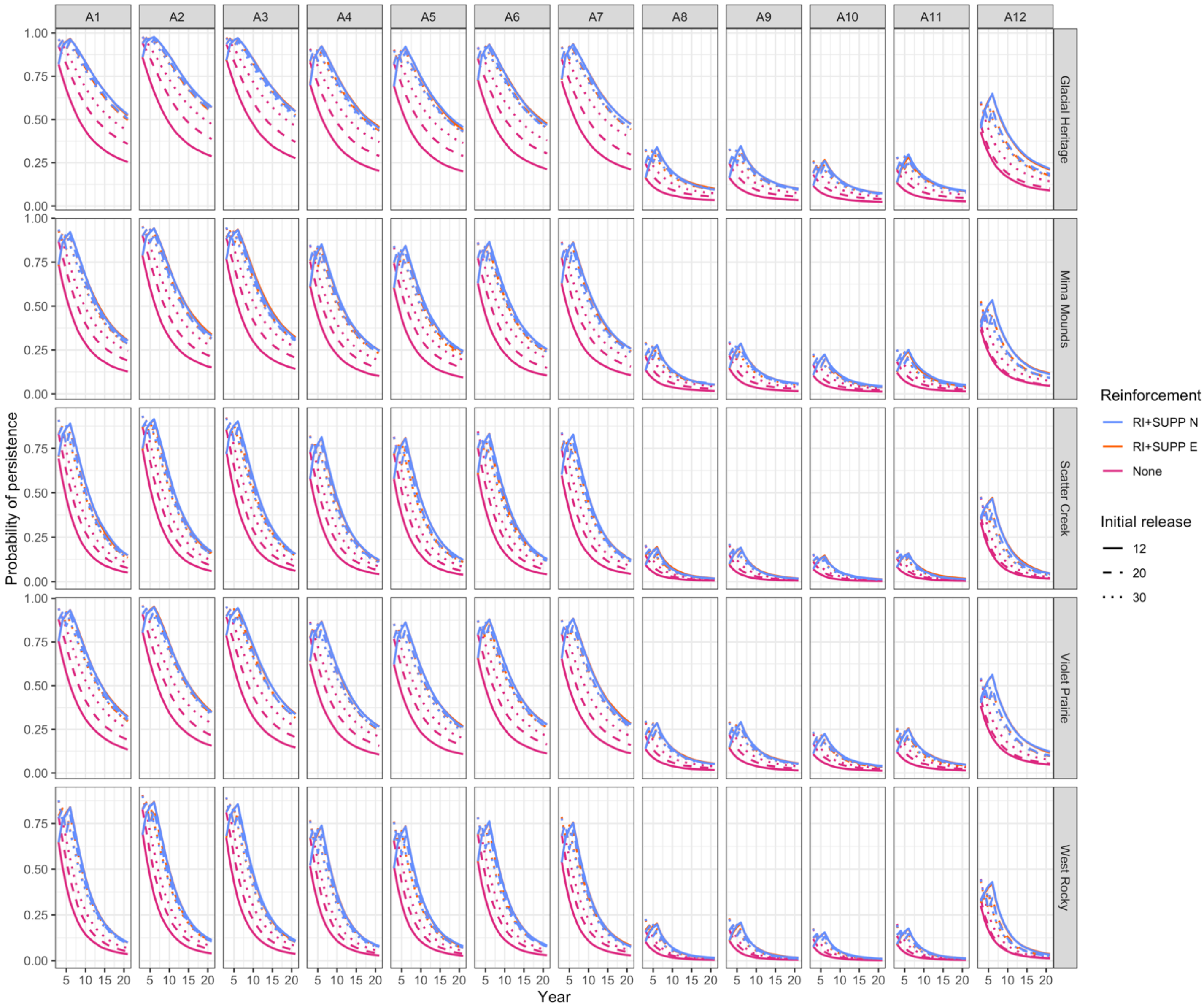

Figure 5.3 Probability of persistence across five reintroduction sites under initial release alternatives (A1 – A12, see Table 5.2) and under various release size alternatives (see Table 5.3). Note that supplementation with SUPP N and SUPP E resulted in approximately similar probabilities of persistence for many reintroduction strategies.

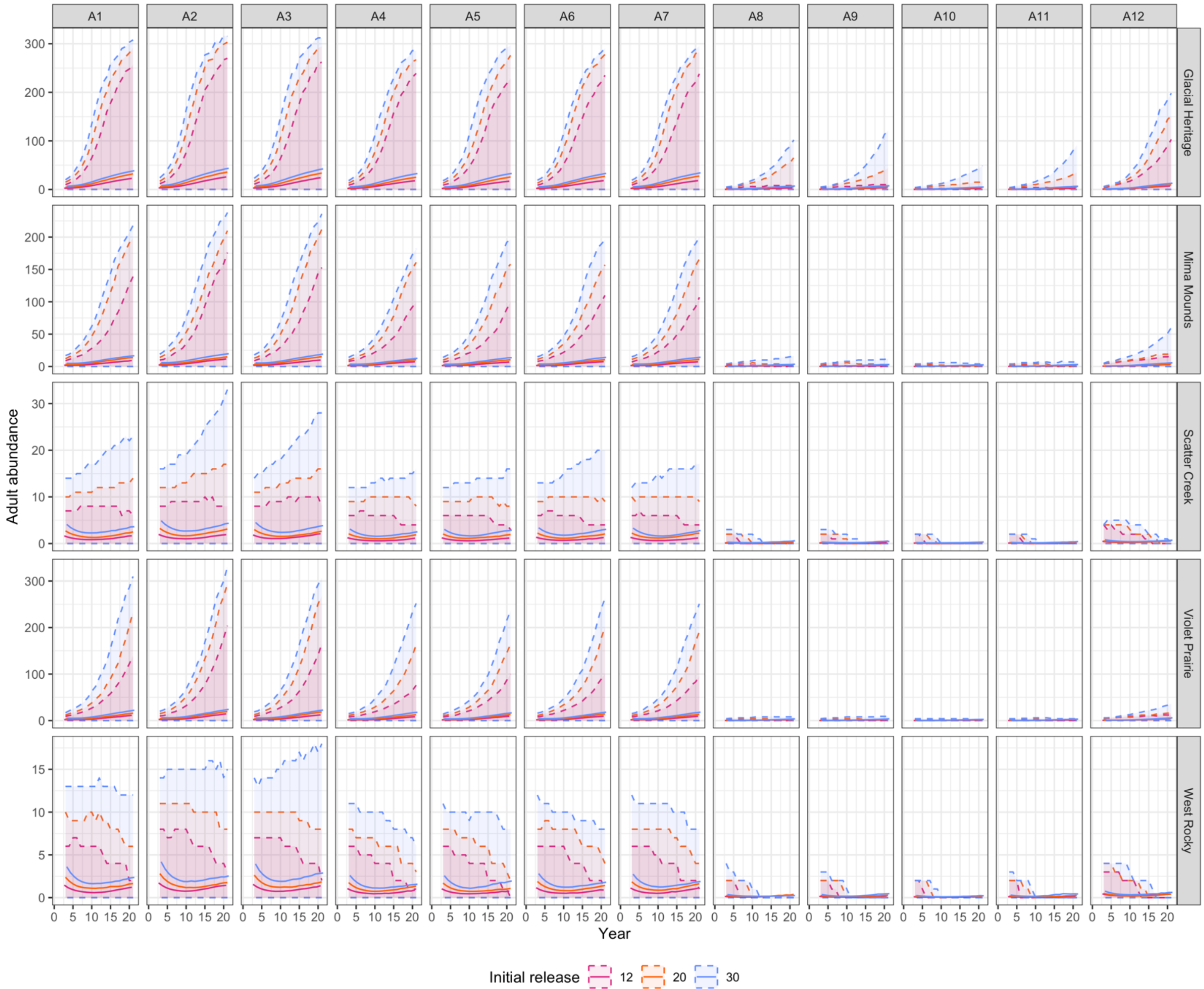

Figure 5.4 Abundance at potential reintroduction sites across all initial release alternatives (A1 – A12, Table 5.2) shown here with the release size alternatives where only the initial release occurs ('None' combinations in Table 5.3). Mean abundance is shown for each initial release abundance with 95% quantiles around the mean.

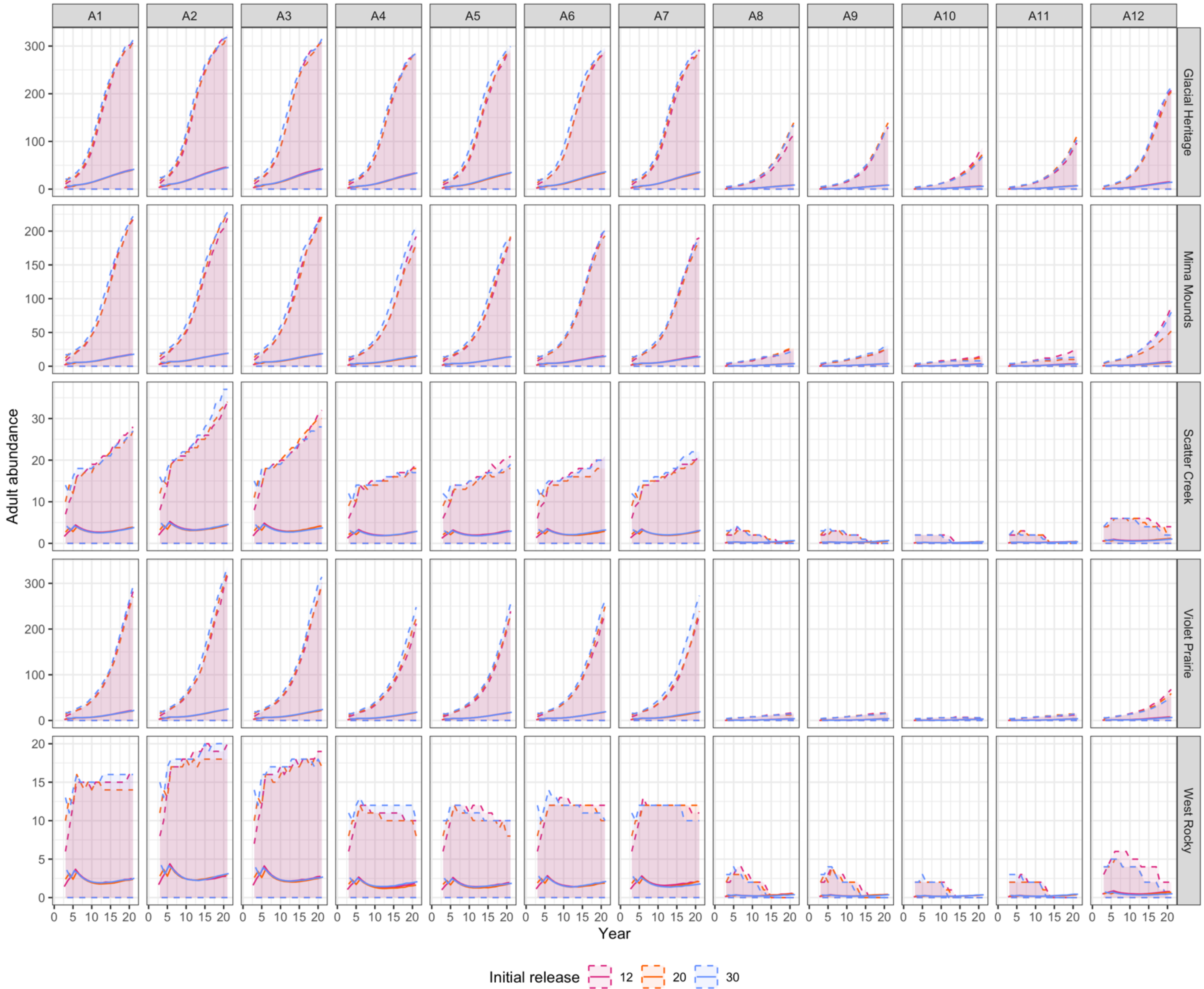

Figure 5.5 Abundance at potential reintroduction sites across all initial release alternatives (A1 – A12, Table 5.2) shown here with the release size alternatives where additional reinforcement releases occur and include nest supplementation with nestling (SUPP N, i.e., 'RI+SUPP N' combinations in Table 5.3). Mean abundance is shown for each initial release abundance with 95% quantiles around the mean.

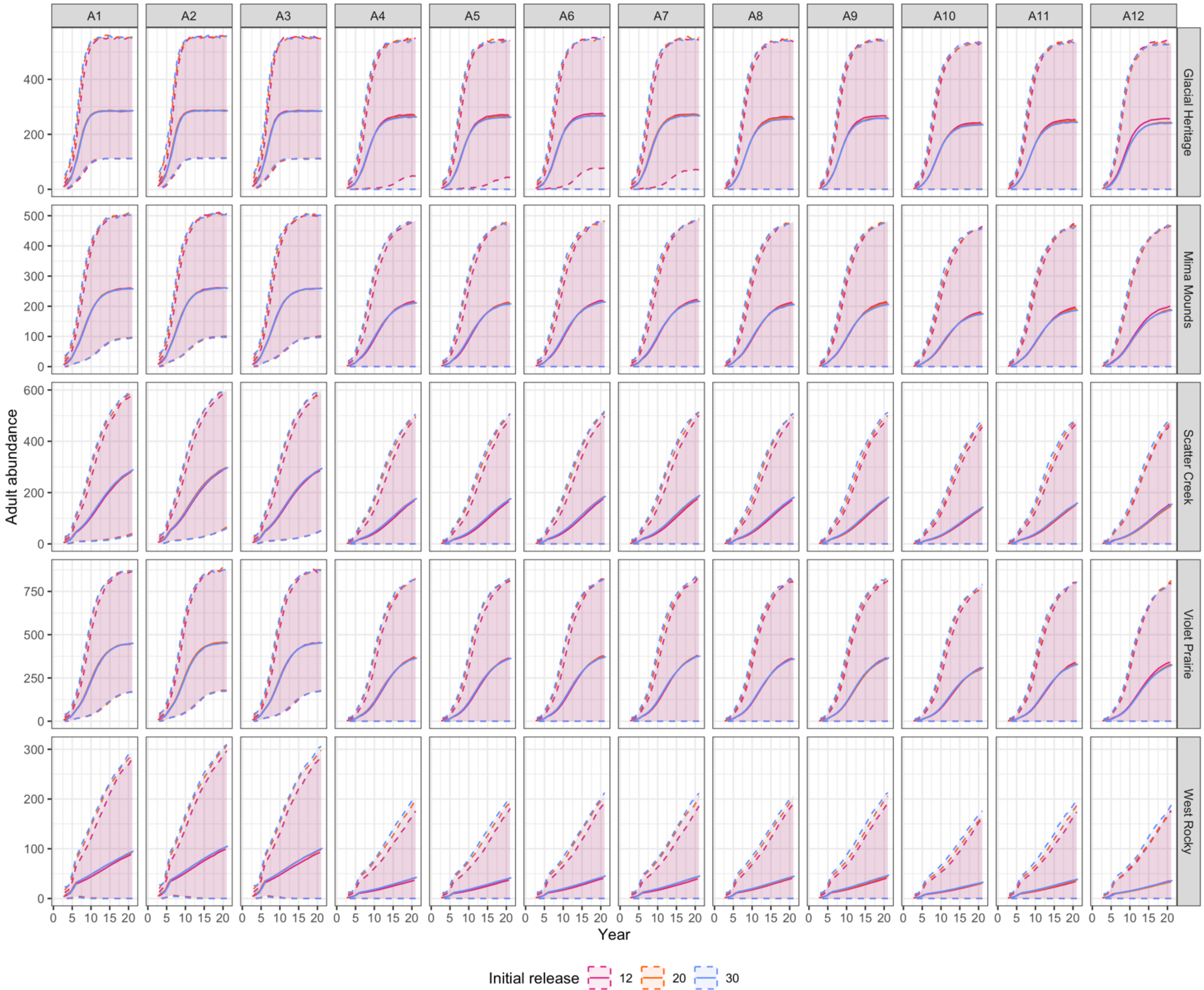

Figure 5.6 Abundance at potential reintroduction sites across all initial release alternatives (A1 – A12, Table 5.2) shown here with release size alternatives where additional reinforcement releases occur and include nest supplementation with eggs (SUPP E, i.e., 'RI+SUPP E' combinations in Table 5.3). Mean abundance is shown for each initial release abundance with 95% quantiles around the mean.

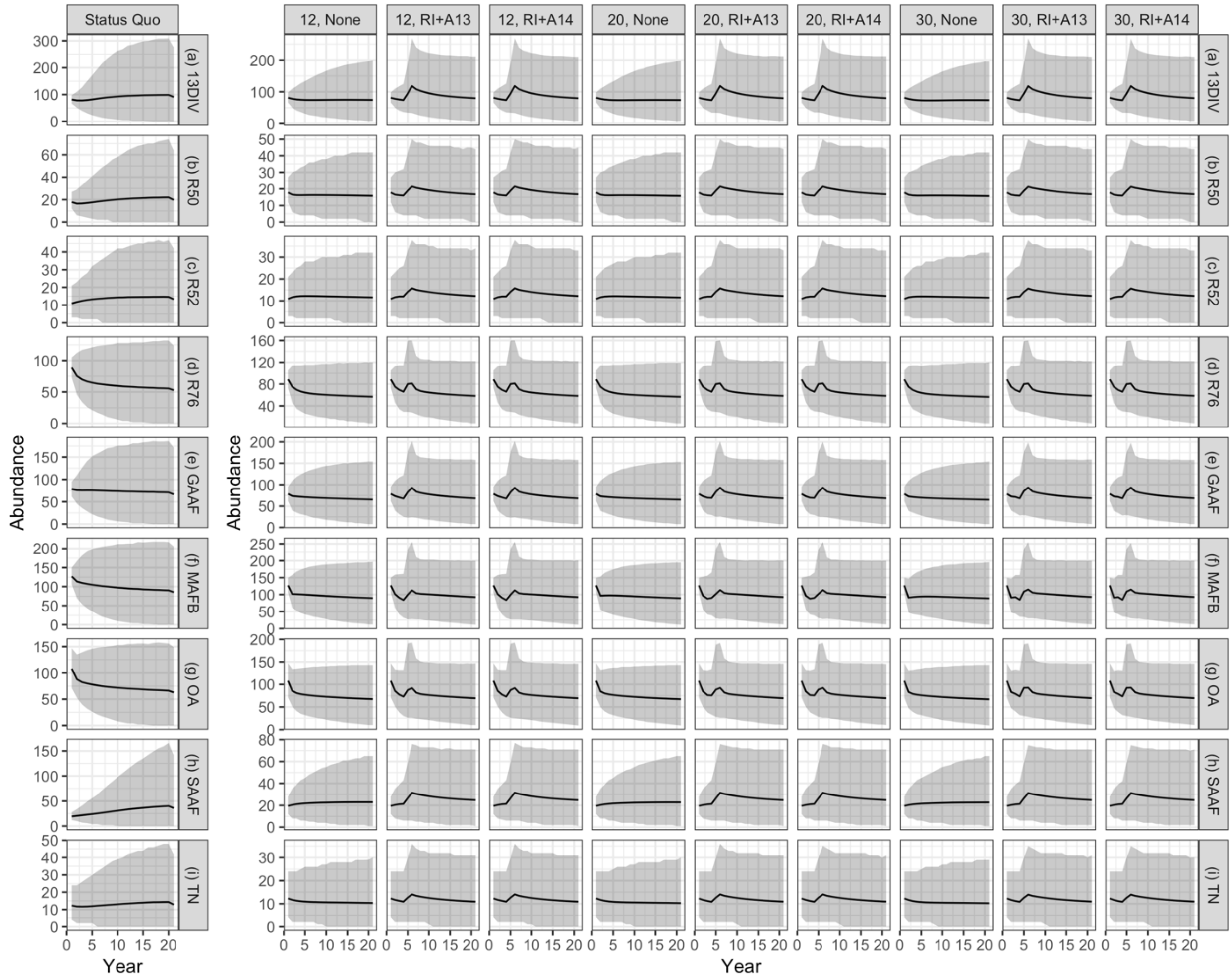

Figure 5.7 Streaked Horned Lark abundance at currently occupied south Puget lowlands sites under various reintroduction strategies where individuals are sourced from airport sites at different rates (Table 5.3). The left panel shows the predicted population mean and 95% quantiles around the mean if no reintroduction occurs. The right panel shows the predicted mean and 95% quantiles around the mean for the release size alternatives in Table 5.3. Airport sites are (e) – (i), as shown in Figure 5.1.

5.9 Appendix 5

Table A5. Expert elicitation participants, affiliation, and phase in which they participated. During Phase 1 we elicited site-specific demographic parameters assuming the population was established. During Phase 2, we elicited the temporary, post-release effects of translocation under various initial release alternatives.

Scientific Expert	Affiliation	Elicitation phase
Gary Slater	Ecostudies Institute	Both
Abby E. Bratt	University of Washington	Phase 1
Scott Pearson	Washington Department of Fish and Wildlife	Both
Timothy Leque	Center for Natural Lands Management	Phase 1
Hannah Anderson	Washington Department of Fish and Wildlife	Phase 2