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Improving the Recovery Efforts of Threatened Species

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Declaration

I, Gemma Taylor, confirm that the work presented in this thesis is my own. Where information has been derived from other sources, I confirm that this has been indicated in the thesis

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Abstract

Reintroductions are becoming an increasingly popular tool for threatened species management and broader scale restoration projects. Reintroductions require a series of important decisions to be made from planning and implementation through to post-release establishment and persistence of populations. Decision making in reintroduction is frequently impeded by high levels of uncertainty. Linguistic, epistemic and aleatory uncertainties often lead to a failure to meet project objectives. This has led to repeated calls for setting clear objectives and using these to focus monitoring in a way that allows applied science to support management.

Viewed in this way, applied science can naturally assist the decision making process. It is important to reduce only the uncertainties that will help inform the choice between two or more possible actions. These can be reduced through targeted monitoring and research. The failure of applied science to approach research in this way is one possible explanation for the ‘research –implementation gap’ that persists in conservation biology. Throughout this thesis I use decision analytic tools to evaluate and inform the discipline of reintroduction biology. Decision analytic tools are increasingly being utilised in diverse fields of resource management. The benefits for more formally incorporating decision science into conservation biology are obvious and repeatedly lauded, yet it remains unclear how much the approach is used to ensure applied science is truly informing management, particularly in the growing discipline of reintroduction biology.

Overall, my PhD intends to promote the application of formal decision tools to threatened species management and showcase how it can reduce uncertainty and

support decision making specifically in reintroductions. In using the Regent Honeyeater recovery actions as a case study, I will evaluate whether management actions to recover the species are working, as well as highlighting areas where resources can be targeted to reduce the uncertainties that influence management decisions, rather than wasting it on those that are not relevant.

Impact statement

Key impacts and outputs:

- Publication in the journal *Trends in Ecology and Evolution*
- Publication in the journal *EMU-Austral Ornithology*
- Identified novel threats to captive bred and released Regent Honeyeaters
- Increased awareness and engagement through local and regional media attention
- Generated AUD $15,000 of additional funding for conservation-focussed research

My research methods and results have showcased how applied science can directly benefit conservation practitioners who need to make difficult decisions regarding the management of a critically endangered bird species in Australia. The science of reintroduction biology has repeatedly been encouraged to better support reintroduction practice and I start by reviewing the reintroduction literature and show that reintroduction biology still has scope to better support decision makers in this field. This paper was published in the journal *Trends in Ecology and Evolution* (Taylor *et al.,* 2017). I then answer these calls for more effective applied science by working directly with the recovery team for the Regent Honeyeater *(Anthochaera phrygia)* to conduct strategic research that is embedded in management.

The management of this species has included multiple reintroduction events and my research has focussed on understanding the factors influencing the survival of the birds once they have been released and the factors limiting their reproductive success in the wild. I undertook the first intensive study of captive bred Regent Honeyeater breeding success which was a key uncertainty of the recovery team and documented for the first time native sugar gliders depredating nests. It was featured in local, regional and national media including Australian Geographic, radio stations and news channels (Prime 7). This chapter has been published in the journal *EMU* (Taylor *et al.*, 2018). These results revealed worryingly low reproductive outputs from captive bred birds that are now free-ranging and were immediately designated as the primary concern for this species.

I then showcase the use of decision analytic tools and bespoke decision trees to facilitate the decision making process for the recovery team regarding the post release management aimed at improving the breeding success for future releases. This gave the recovery team the opportunity to consider multiple management alternatives and I was able to further focus my research to answer their key questions. I did this by designing and trialling nest protection methods; tree collars and nest cages on artificial nests.

This thesis has important consequences for the critically endangered Regent Honeyeater, threatened species management in Australia and the wider reintroduction biology community. My research responded to long-standing calls to improve conservation outcomes by bridging applied science and conservation practice. I identified and filled research gaps specifically relating to the Regent Honeyeater and through collaborative work, have helped focus future management for this species as well as showcasing to the global conservation community an approach that can be employed to help managers achieve recovery objectives.

Table of Contents

List of figures

List of tables

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This PhD is dedicated to my Grandad, Eric Norris,

and my son, Reuben Massarano

Chapter 1 ~ Introduction

The challenges of threatened species recovery

Attempting to recover populations of threatened species is the key focus of biodiversity conservation. Recovery, however, is made difficult because in most cases there are numerous contributors to a species decline and our knowledge of these and the species we are attempting to manage is frequently poor. Given that threatened species are already subject to the process causing their decline, most conservation efforts are post-hoc, rather than preventative (Scheele *et al.,* 2018). Even when we have a good understanding of the threats we may be uncertain as to how best to manage them. Furthermore, recovery efforts require substantial resources over prolonged periods, requiring strong management structures with effective leadership and long-term commitment (Lindenmayer 1999; Black *et al.,* 2011). In essence therefore, successful recovery will require overcoming challenges of both uncertain biological systems and the management structures (including political and economic) within which recovery programs operate. Unfortunately, these scenarios are all too frequent as conservationists face an increasing extinction crisis (Tilman *et al.,* 2017).

Uncertainty is unavoidable in decisions about all forms of conservation actions. However, despite the growing body of work recognising the importance that uncertainty plays (Runge *et al.,* 2011; Polasky *et al.,* 2011; Kujala *et al.,* 2013), the scientific literature rarely recognises nor clearly defines the forms it can take. Three types of uncertainty are commonly referred to: *aleatory, epistemic* and *linguistic* uncertainty (Regan *et al.,* 2008; Runge *et al.,* 2011; Kujala *et al.,* 2013; Mccarthy 2014). The first, aleatory uncertainty, also called statistical uncertainty, refers to demographic and environmental stochasticity; the intrinsic and extrinsic variance influencing population growth. Threatened species management largely focuses around dealing with a combination of both these processes. They can be divided into two non-mutually exclusive paradigms, the '*declining population*' and '*small population*' paradigms (Caughley 1994).

The declining population paradigm deals with the extrinsic drivers of population declines caused by environmental fluctuations (environmental stochasticity). These factors show small to moderate spatiotemporal variation and influence the growth rate of a population (Kendall 1998; Saether *et al.,* 1998). Most environments are dynamic and will show both predictable and unpredictable fluctuations over time (Foley 1994; Caughley 1994). Small populations however are much more vulnerable to any level of environmental perturbation and the continuous variations they experience are capable of reducing numbers further, to the point where other demographic and density dependent factors accelerate the time to extinction (Engen *et al.,* 1998).

The small population paradigm focusses on the intrinsic factors (demographic stochasticity) such as body size and reproductive rate of a species that increase the risk of extinction when the population has already declined to low numbers (Caughley 1994; Asquith 2001). These factors cause realised population growth rates to vary from expected rates due to birth and death rates of individuals, sex ratios and dispersal (Lande 2009; Jeppsson and Forslund 2012). Birth and death rates are probabilistic and will cause fluctuations within a population (Legendre *et al.,* 1999; Lande 2009; Rajakaruna *et al.,* 2013). This is an important concept to understand and acknowledge in the recovery of a small population as it increases the chances of extinction and creates uncertainty regarding these key vital rates when modelling population viability. In large populations these demographic rates will fluctuate around an equilibrium and the effects of the variance in these parameters between individuals decreases with increasing population size. Of concern in small populations is that any changes in this variance can make the population more vulnerable to extinction (Clout *et al.,* 2002). When populations are at a very small size they may also suffer an associated problem of negative density dependence where individual fitness is reduced due to a loss of the benefits of conspecifics (known as an Allee effect) (Stephens and Sutherland 1999; Boukal and Berec 2002). Populations that are subject to Allee effects have a lower growth rate when the density of individuals is low, especially if certain behaviours such as breeding, foraging, hunting, social thermoregulation and predator defence are more effective with the cooperation of others (Stephens and Sutherland 1999; Courchamp *et al.,* 1999; Deredec and Courchamp 2007; Jeppsson and Forslund 2012).

Other concerns related to small, threatened populations are genetic drift and inbreeding depression (Caughley 1994; Stephens and Sutherland 1999). Small populations experience greater effects from genetic drift than larger populations as random sampling of parental alleles to offspring through reproduction results in loss of alleles at low frequency (the fewer chances for an allele to be passed on the more likely it is to be lost). In addition to drift, there is an increased frequency of closely related individuals breeding with each other through random mating and in small populations which have experienced a bottleneck this is a more frequent occurrence. Offspring of related parents will show more homozygosity, increasing the risk of deleterious recessive genes being expressed and/or reducing the expression of heterozygote advantage (Tanaka

2000; Szulkin and Sheldon 2007; Keller *et al.,* 2012). Genetic drift and inbreeding can affect populations differently depending on their interaction with the external environment, and it has been found that 'stressful' environments, such as those with extreme weather and disease, can heighten inbreeding depression and the variability of fitness within a population (Tanaka 2000; Armbruster and Reed 2005). Biodiversity loss is a symptom of these intrinsic and extrinsic processes all interacting together, and effective conservation actions will need to consider threats within both paradigms: those inherent to a small population, as well as understanding why it is a small population in the first place.

Linguistic and epistemic uncertainties relate more specifically to the management difficulties inherent to threatened species conservation. Linguistic uncertainty refers to the imprecise language that is often used when communicating, consequently confusing the whole decision process (Regan *et al.,* 2008; Kujala *et al.,* 2013). In conservation words such as 'self-sustaining', 'risk' and 'success', are used regularly and appear in many species management plans, usually when stating aims and objectives (Sarmenta *et al.,* 2001; Decāk *et al.,* 2005; Lovaszi 2012). However, without a clear definition of such terms specific to the context in which they are being applied, there is potential that they will become a source of vagueness and ambiguity.

Epistemic uncertainty refers to that which arises from a lack of knowledge or data on the species and the system in question and will vary between people and organisations. It unavoidably pervades all areas of species conservation, hinders management decisions and reduces outcome efficacy, but can be somewhat eliminated through the inclusion of different areas of expertise as well as focussed data collection (Regan *et al.,* 2008; Runge *et al.,* 2011; Kujala *et al.,* 2013). One type of epistemic uncertainty is measurement error, which arises from what appears to be random variation in the measurement outcomes of sampling techniques, observation and equipment, meaning that repeated measurements of the same quantity will show statistical variance about a mean (Regan *et al.,* 2008). Parametric uncertainties pertain to those generated from parameter values and the preciseness of the initial system model, and this can be augmented by measurement error. This can be reduced by focussed monitoring and data collection to improve our knowledge on the species and system parameters we are using to build the models (Fieberg and Jenkins 2005). Sensitivity analysis can be used to indicate which parameters are having the greatest impact on the model estimates. A second type of epistemic uncertainty is model or structural uncertainty which in contrast, arises as we never completely understand how components of the system interact with each other, for example a predator prey relationship, or how a native competitor may interact with the reintroduced species. Therefore, even in the most well designed and clearly articulated of management plans, uncertainty will influence our ability to make decisions (Milner-Gulland and Shea 2017).

It is clear that conservationists have a multitude of complex problems to solve when faced with recovering a population which is heading towards extinction. It is therefore important to not only have a clear understanding of the factors that are driving the decline, but also ensure the management team is fully supported with multidisciplinary expertise, including researchers and other stakeholders, and decisions are made within a strategic and transparent framework (Scheele *et al.,* 2018; Schwartz *et al.,* 2018).

"*Most wildlife management problems start out as biological problems but eventually become people problems*" (Teague, 1979 in Jacobson and McDuff, 1998). Species conservation is driven by human values and therefore cannot be solved solely through scientific methods. The success of threatened species recovery attempts can be hindered by poor communication and management planning (Battisti 2017; Schwartz *et al.,* 2018), ineffective monitoring (Legg and Nagy 2006a; Ewen and Armstrong 2007), 'evidence complacency' (Sutherland and Wordley 2017), failure to engage with the public (Jacobson and McDuff 1998) and a general divide between scientific researchers and conversation practitioners (Pullin *et al.,* 2004; Gardner *et al.,* 2018). The role of applied scientists is to conduct robust, hypothesis driven research aimed at reducing the uncertainties mentioned above, and then to communicate this to managers so they can make informed decisions. Managers therefore have the responsibility of using this research to make decisions and plan conservation projects, however here is where communication and planning issues can arise. The collection of more data is often proposed as a management action when faced with a decision that has no clear answer. Monitoring can easily be a waste of valuable time and money if it is not aimed at reducing uncertainty that hinders a choice between two or more possible actions. Even when data collection is aimed at reducing uncertainty, it can all too often focus on the more manageable uncertainties, but not those that are important and controllable (Milner-Gulland and Shea 2017). The formal integration of monitoring into the management program intuitively ensures that the effort invested in monitoring is effective and useful to decision and policy makers. This process explicitly facilitates learning by designing the monitoring to allow feedback between the management alternatives and the predictions. Conservation managers must then make a concerted effort to engage with applied scientists and use the evidence that is available, reducing what has been coined ‘the research –implementation gap’ (Flaspohler *et al.,* 2000; Ormerod *et al.,* 2002; Knight *et al.,* 2008; Lauber *et al.,* 2011); something that has long been recognised as needing attention in conservation management (Pullin *et al.,* 2004; Sutherland and Wordley 2017). Uncertainties can be ignored if a solution to reduce them is not obvious, however this inevitably leads to the development of a more dogmatic approach to decision making, instead of using experiments to support and inform decisions (Martínez-Abraín and Oro 2013; Milner-Gulland and Shea 2017).

Each recovery program faces its own set of challenges, yet there is a deficiency in literature specifically on what contributes to a successful recovery, making learning from others and predicting outcomes difficult. Recent literature within conservation science has sought to evaluate the characteristics of management that are connected to both successful and unsuccessful projects, citing the identification and removal of threats as the major predictor of success (Manolis *et al.,* 2008; Crees *et al.,* 2016). Strong leadership has also been identified as key to effective management, and a lack of leadership within conservation science has been recognised as contributing to failed projects (Dietz *et al.,* 2004; Manolis *et al.,* 2008; Black *et al.,* 2011). Some of the characteristics of strong and successful leadership have been identified as: the ability to integrate knowledge and values from multiple stakeholders, learning from experimentation and the ability to negotiate through conflict (Dietz *et al.,* 2004; Manolis *et al.,* 2008; Black, *et al.*, 2011).

One approach that can be employed to help attain these leadership qualities and improve collaborative work between managers and applied scientists is the use of decision analysis tools. In the next section I introduce these tools and describe how they can be applied to management decisions in a conservation context.

Decision analysis to support threatened species recovery

Decision analytic tools offer great potential in achieving structured support for conservation planning and a means to foster collaborative decision making between multiple stakeholders (Schwartz *et al.* 2018). Decision analysis, developed from risk assessment and probability theory, structures the process by which decisions are made under uncertainty (Caughley 1994). It has been advocated as a method for conservation managers since the 1980's (Maguire 2004) and has been widely applied in natural resource management (Gregory and Failing 2002; Espinosa-Romero *et al.* 2011), yet seldom been applied in the field of conservation translocations (Struhsaker and Siex 1998; Bennett *et al.* 2012; Xia *et al.* 2014). This is gradually changing and efforts to incorporate decision analysis into practical conservation are encouraged with courses such as those offered to conservation practitioners by the National Conservation Training Centre (USFWS).

It draws on powerful analytical tools to provide managers with information on the best recovery action among available alternatives to achieve stated objectives (Burgman and Yemshanov 2013). Decisions made in this way work systematically through the elements common to all choices: problem framing, objectives, alternatives, consequences and trade-offs (Figure 1) (Hammond *et al.* 2002).

Decision analysis can help in multiple-objective decision making by providing a means for optimising across them in a highly transparent way. The first step is defining the problem scope, ensuring that everyone understands the context of the issue and that any vagueness and ambiguity surrounding language, key terms, people and the roles of each party is removed (linguistic uncertainty) (Davies *et al.* 2013). Second, is clearly stating the fundamental objectives and their associated performance measures (i.e. metrics managers can use to compare alternative actions) (Espinosa-Romero *et al.* 2011; Mccarthy 2014). Objectives reflect our values: for example, the objectives may be species specific (to re-establish a historic range or establish a captive breeding population), they may be habitat specific (the removal of invasive species) or they may be social (to increase public awareness, encourage community involvement) (Ewen *et al.* 2014). More than likely the objectives (fundamental and means) encapsulate elements of all these values, especially when multiple stakeholders are involved. Fundamental and means objectives differ; fundamental objectives reflect the end goal that is desired to be achieved and means objectives are the ways in which they are achieved. The third step is the development of alternative management options. The previous stage of defining objectives should facilitate the creation of numerous alternatives, something which is not always considered and is a major drawback in other decision making approaches which do not offer the chance to develop alternatives, as it is difficult to place value and importance on an action if it cannot be explicitly compared to a set of other proposed actions. Management alternatives need to be the following: 1) a complete solution to the problem, 2) directly comparable, 3) value focussed, 4) fully specified, 5) internally coherent and 6) distinct (Gregory *et al.,* 2012). With these requirements for alternatives, and the elimination of those that do not satisfy the chosen objectives, a concentrated set of alternatives should remain to be evaluated (Espinosa-Romero *et al.,* 2011).

Once a set of possible alternative actions have been generated it is important to estimate the consequences of applying each of them against the objectives, using the defined performance measures. Consequences should present all the information needed to highlight the uncertainties of each alternative. This process enables stakeholders to contend with the trade-offs that need to be made when deciding for or against competing alternatives (Converse *et al.*, 2013; Ewen *et al.*, 2014; Gregory *et al.*, 2012). This step can be difficult as is it involves a tradeoff and compromise on values between someone, or a stakeholder group. It needs to remain clear that this stage is about assigning weights and optimising the alternatives that best achieve the objectives, objectives that were chosen and agreed on by all involved. Approaching conservation planning this way naturally facilitates the development and implementation of the monitoring program component.

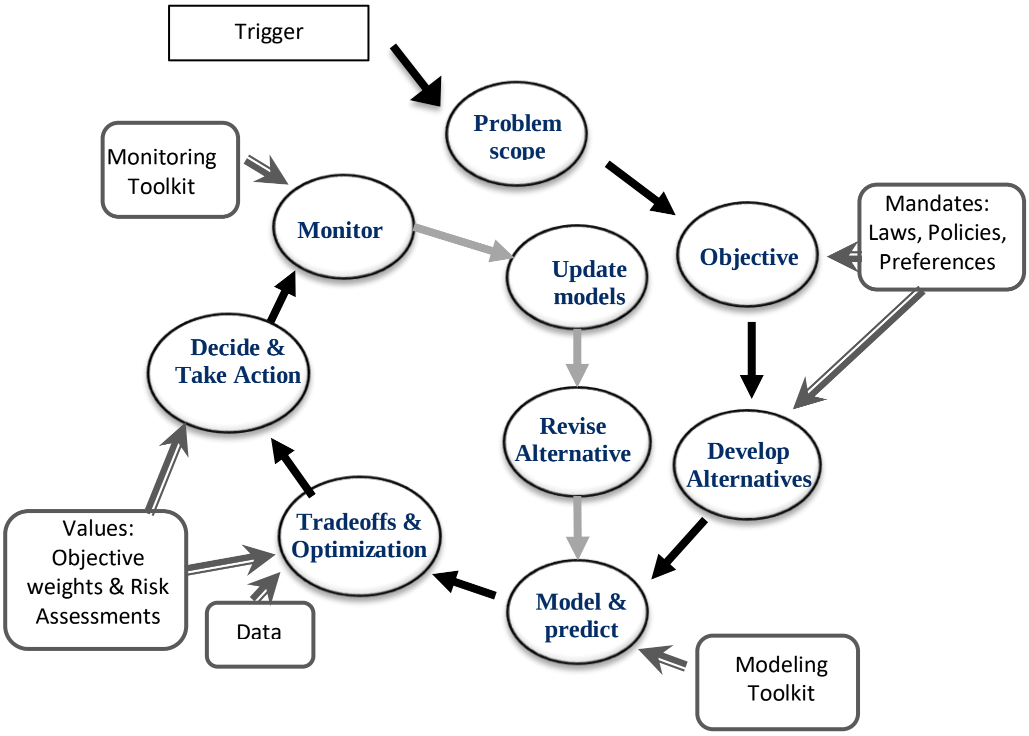

Figure 1: Schematic showing stages of Structured Decision Making (modified from Gregory, 2012)

Those involved in the decision analysis process will have some sort of predetermined idea of the system: the species, the habitat and any other factors which contribute to their conceptual model. This process encourages groups to articulate clearly what it is they are trying to achieve, and to estimate how well they can do so by implementing given actions. At this stage the uncertainties in the data used to compare possible alternatives will be highlighted, and how they influence our ability to select between options. Sensitivity analysis can be used at this point to measure how important the implications of these uncertainties are under certain management actions, and which uncertainties are most important to reduce.

The management of threatened species is an area that can benefit from a structured approach to decision making. One area of conservation which has particularly high levels of uncertainty and often involves multiple stakeholders is species reintroductions.

Conservation Translocations

Conservation translocations are an important type of management used to recover threatened species (Seddon and Armstrong 2016). When species translocations are carried out for population conservation purposes they form part of a spectrum of conservation translocations (IUCN, 2013). For example, species can be moved to areas within their indigenous range or outside of it: *assisted colonisation* refers to the situation where a species is intentionally moved into an area it may or may not have historically occupied, most topically so that range restricted species can persist under climate change (Chauvenet *et al.,* 2013). *Ecological replacements* use non-native species as a substitute for an extinct native species to re-establish or maintain ecosystem functions (Seddon *et al.,* 2014). When a species is released into an area which is known to be part of its indigenous range, it can be defined as either a *reintroduction*, if the species is currently extirpated, or as a *reinforcement*, when an endangered population is enhanced with conspecifics (Seddon *et al.,* 2014; Malone *et al.,* 2018).

Despite the growing use of reintroduction as a conservation tool and the increasing numbers of species recovery plans that include captive breeding for release as part of their management, there continues to be a high failure rate. One reason for this is that reintroductions inherently have a lot of uncertainty and require a series of important decisions to be made from the planning and implementation through post-release establishment and persistence of populations. Poorly informed decisions lead to inefficient management and failure to meet objectives. As mentioned in previous sections, this, coupled with biological uncertainty, can severely hinder achieving successful outcomes. In addition, the majority of failed reintroductions go unpublished either in the grey or scientific literature, resulting in the underlying mechanisms causing failure to remain poorly understood and inhibiting any opportunity to learn (Parlato and Armstrong 2013; Ewen *et al.,* 2014).

Historically both pre and post-release monitoring in reintroduction attempts have either been lacking completely, or inadequately planned (Legg and Nagy 2006b; Ewen and Armstrong 2007; Ewen *et al.,* 2014). More recently, reintroduction plans have heeded the advice to include monitoring and the majority have a post-release monitoring component. However, this component has tended to be unfocussed and used in reflective analysis of the reintroduction attempt (Armstrong and Seddon 2008) rather than answering *a priori* hypotheses to reduce uncertainty. This has led to repeated calls for setting clear objectives and using these to focus monitoring in a way that allows applied science to support management (Muths and Dreitz 2008; Parker *et al.,* 2013; Taylor *et al.,* 2017). This need for improvement on planning and implementing focussed monitoring lends itself well to employing decision tools to help with this process (Converse *et al.,* 2013; Canessa *et al.*, 2016; O'Donnell *et al.*, 2017) and more research in the recovery of threatened species are showcasing how decision analysis can be implemented. For example O'Donnell *et al.,* (2017) use structured decision making to develop recovery plans for Flatwoods salamander species (*Ambystoma bishopi* and *A. cingulatum*), specifically addressing uncertainties regarding breeding and habitat restoration. They detail the steps taken, which included workshops with stakeholders, and describe how decisions were made and implemented as a result of employing a formal decision framework. Conroy *et al.,* (2008) give a detailed example of how contentious decisions, with multiple stakeholders who have conflicting values and beliefs can be addressed, using Hector's dolphin *(Cephalorynchus hectori),* an endangered species native to New Zealand, as an example.

In this thesis I review the use of decision analytic approaches for threatened species management, specifically reintroductions and reinforcements, and present an example of how it can inform management and target specific uncertainties through focused monitoring and experimental design. The following sections introduce the species and site used throughout this thesis as a case study.

The Regent Honeyeater case study

The Regent Honeyeater (*Anthochaera phrygia)* is a monotypic passerine species endemic to southern Australia (Driskell and Christidis 2004; Liu *et al.,* 2014). Under national and state jurisdiction in Australia the species is listed as threatened and it is classified as Critically Endangered in the IUCN Red List (Birdlife 2013; Ingwersen et al., 2015). The Regent honeyeater was once widely distributed in the woodland belt from the Adelaide region in South Australia to southern Queensland, about 100 km north of Brisbane (Ingwersen *et al.,* 2015). However, now it is believed to be locally extinct in South Australia and western Victoria and rare in Queensland (Oliver *et al.,* 1998; Thomas 2009), Figure 2. Historical accounts of the species have described 'flocks of thousands' (Geering and French 1998), yet surveys in the 1980s estimated a population number of around 1500 individuals, whilst more recent surveys estimate only about 100 breeding pairs (Crates et al., 2018).

Ecology

Regent honeyeaters currently occur in box-ironbark eucalypt woodland and lowland coastal forests, also dominated by a eucalyptus species (*Eucalyptus robusta),* within which they prefer the wetter, fertile and more riparian areas (Thomas 2009). Their diet consists mostly of nectar from a few key mistletoe and eucalypt species, including Mugga Ironbark (*Eucalyptus sideroxylon),* Yellow Box *(E. melliodora),* White Box *(E. albens)*, Yellow Gum (*E. leucoxylon) and* Box Mistletoe *(Amyema miquelii)*, preferring larger, mature trees which produce greater quantities of nectar. Despite long term colour banding of captive and wild birds and regular surveys at key habitat sites, their movements remain poorly understood. They respond to food resources and seasonal movements appear to coincide with flowering events of key feeding species; however this is spatially and temporally unpredictable (Ford *et al.,* 2001). Some individuals have been observed returning to the same breeding sites, whereas others have been recorded at different sites from year to year, suggesting long distance travel of up to 534 km (Oliver 1998b; Powys 2010; Ingwersen *et al.,* 2015). Large concentrated flocks have been observed around nectar sources in the box-ironbark woodlands during the main breeding season, which is typically from August to January.

Figure 2 -Map showing historical range of Regent Honeyeaters from South Australia to southern Queensland (shaded). Circled areas show current known breeding sites within Victoria and New South Wales

Threats

The ultimate driver of decline in regent honeyeater numbers is loss and fragmentation of eucalypt woodland habitat, which in turn has led to a series of additional threatening processes that has further exacerbated decline. The habitat crucial to the regent honeyeater and many other passerine woodland species now has the highest density of threatened and declining bird populations in Australia (Major *et al.,* 2001). In recent history southern Australia has experienced extensive clearing of woodland and conversion to agricultural land and sheep grazing pastures because of the highly fertile soil (Mac Nally *et al.,* 2000; Ford *et al.,* 2001; Watson *et al.,* 2011). The box-ironbark woodlands are situated on auriferous rocky soils and were also cleared during the gold-mining rush in the 1800s when the timber was used to support the subterranean mines. This woodland now only covers about 15% of its original area (Mac Nally *et al.,* 2009). For Regent Honeyeaters in particular, it seems that fragmentation negatively impacts their ability to disperse and follow the flowering events of nectar species, which are spatially and temporally dynamic throughout the year (Ford *et al.,* 1993; Ford *et al.,* 2001). As a consequence of fragmentation and further habitat degradation, Regent honeyeaters are more at risk from inter-specific competition from other nectar feeding species, such as the noisy miner *(Manorina melanocephala),* the noisy friarbird *(Philemon corniculatus)* and red wattle bird *(Anthochaera carunculata)* (Ingwersen *et al.*, 2015).

Management

In 1994 the national Regent Honeyeater Recovery team was formed (Cooke and Munro 2000; Powys 2010) with the goal of increasing the numbers of regent honeyeaters so that there is a viable wild breeding population (Ingwersen *et al.,* 2015). Members of the recovery team represent multiple organisations and institutions including; Birdlife Australia, Taronga Zoo, New South Wales and Victorian Government departments and Australia National University. Members have a range of expertise and some have been working with the species for decades. The recovery team does not currently include members from any indigenous organisations.

In order to achieve the recovery goal, three main management strategies have been employed: habitat restoration, noisy miner control, and captive breeding and release. The first species action plan was developed in 1994 and a second in 1999, of which a review concluded significant improvements were needed in the knowledge of the species ecology and it identified areas of research needing further attention. The captive breeding started in 1995 when 18 young nestlings were taken from the wild and raised in captivity at Taronga zoo, Sydney, and captive bred chicks were born within the first year (Liu et al., 2014). In 2000 the first release of nine captive birds with radio tracking harnesses occurred at Capertee Valley, New South Wales; however, only three are known to have survived the first month. In 2008, after improvements were made to the harness design, a further 27 birds were released in Chiltern National Park, Victoria. This was followed by 43 in 2010 and 38 in 2013, 77 in 2015 and 101 in 2017, with each release timed to coincide with flowering events (Liu et al., 2014; Birdlife International 2015).

Challenges for Species Recovery

The Regent Honeyeater Recovery Program serves as an exemplar for a project facing both numerous biological uncertainties, as well as uncertainties associated with a multi-stakeholder management team. There is much uncertainty regarding the movements of this semi-nomadic species and what is causing its patchy occurrence across south east Australia. Their survival is heavily dependent on the availability of the key food source - native nectar from eucalypts and other species such as *Callistemon* sp. – but heavy landscape modification has reduced food availability and disturbed the pattern of flowering (Kelly and Mercer 2005; Bennett 2016). Regent Honeyeaters are regularly sighted in locations such as the Capertee Valley and Hunter Valley in New South Wales during the Austral winter months, but very little is known about their preferred habitat during the summer months. Their small population size, spread over a potentially extremely large range, has meant that accurately quantifying their population, and measuring population change, has been difficult (Crates *et al.,* 2017). Very little was known about the reproductive success of wild birds, or of those released from captivity.

The management of the Regent Honeyeater has been hindered by many of the issues mentioned in the earlier sections. Much of the research on the species was conducted in the 1980s with no new data collected or analysed until recent years. Extensive post release monitoring was conducted each year in Chiltern Mt-Pilot National park, but no analysis of these data had been performed in order to update knowledge and inform future management. The main organisations responsible for the planning of the release are an NGO and a regional government department, with 1-2 personnel from each working on the project, as well as on many other projects. Until recently science has not been well integrated into the management and decisions have been driven by experience and anecdotal evidence. The contribution of applied scientists in the past has been sporadic, and in the last four years the team working on the species has expanded to include scientists from academic institutions and new data have been collected on their range occupation and breeding (Crates *et al.,* 2017; Taylor *et al.,* 2018).

Study site

Chiltern-Mt Pilot National Park (36°7’59”S 146°36’4”E) in North Eastern Victoria has been the chosen site for the releases of captive bred Regent Honeyeaters from Taronga Zoo in the years 2008, 2010, 2013, 2015 and 2017, as historically Regent Honeyeaters used this area for breeding and wild birds are still observed here. It is 21,600 ha of primarily regrowth open box-ironbark forest located dominated by species such as *E. sideroxylon*, red stringybark (*E. macrorhyncha*) and box eucalypts (*E. albens, E. macrocarpa and E. polyanthemos*) (Harrisson *et al.,* 2014). Approximately 4,600 ha in the northern section of the park comprises the core habitat for regent honeyeaters and where the releases have taken place. The location of Chiltern-Mt Pilot National Park is on the border of Victoria and New South Wales. Historically, this area has been very important for local indigenous peoples as well as a meeting site for travelling groups. The country to the north east of the Murray River in New South Wales belongs to the Wiraduri people. This is the largest group in New South Wales and their country stretches as far as the west of the Blue Mountains, encompassing much of the historical habitat of the Regent Honeyeater. Currently there is dispute over who has native title rights in Chiltern and the surrounding area; however at present it is believed to be situated on the traditional land of the Dhudhuroa-Waywurru and Pangerang people (Blake and Reid 2002).

Although it was only designated as a national park in 2002, the Box-Ironbark forest of Chiltern has a long history of anthropogenic disturbance. There is evidence of traditional use such as scarred trees and rock art, which are still present. In the 1850's the area was heavily altered by European settlers and the majority of the forest was cleared for gold mining and timber (Parks Victoria, 2008). Despite a history of significant degradation, Chiltern Mt-Pilot National Park is considered to have the most intact Box-Ironbark faunal assemblages in Victoria, with the highest number of birds, mammals and reptile species recorded for the type of woodland. It is also cited as the most important site in Victoria for the threatened Squirrel Glider (*Petaurus norfolcensis*) (Parks Victoria, 2008). The northern section of the park is embedded in a mosaic of agricultural land, which directly adjoins the park boundary. It is also divided by the Hume freeway, with the township of Chiltern (population 1200) situated between the two sections. This has resulted in significant fragmentation and increased detrimental impacts such as feral cats and dogs, invasive grasses and other flora, and overgrazing (Parks Victoria, 2008). Large wildfires, most notably in 2003 and 2015, have reduced the number of breeding Barking Owls *(Ninox connivens)* and Powerful Owls *(Ninox strenua)*, which would have been the apex predators in the area (Parks Victoria, 2008).

Although the plight of the Regent Honeyeater is geographically localised, the causes of its decline - habitat loss and fragmentation - and the issues that are faced by managers when trying to recover the population are those faced worldwide by the conservation community. Research for this PhD will not only inform species specific management plans, but help showcase how better decisions can be made in the future. It will permit a pathway for science to contribute directly to conservation management, fostering a more robust cohesion between applied research and managers.

The chapters in this thesis aim to showcase how decision analytic tools can help improve the recovery efforts for threatened species. The chapters were planned specifically with the aim of, first, highlighting areas in the field of reintroduction biology which are failing to meet reintroduction objectives, and second, applying decision tools, working alongside a threatened species recovery team to demonstrate how they can be applied to real-world conservation problems.

In ***chapter 2***, I review the reintroduction literature from the past 20 years to assess how well applied science in reintroduction biology is employing decision theory to inform management, specifically by testing between two or more possible management actions. This chapter has been published in the journal *Trends in Ecology and Evolution* (Taylor *et al.,* 2017).

In ***chapter 3***, I analyse the short-term survival of captive-bred and released Regent Honeyeaters using monitoring data from releases in 2010, 2013, 2015 and 2017. I test *a priori* hypotheses that post release survival is influenced by the age, sex and weight of individual birds at release, as well as the effect of wearing a radio transmitter. This information is crucial to help the recovery team understand which management actions are positively impacting the survival of birds post-release, and how planning for future releases can be improved.

In ***chapter 4,*** I reduce uncertainty around the factors that are causing poor breeding success in the captive bred and released regent honeyeaters in Chiltern Mt-Pilot National Park. I collected the first data on breeding effort for this population, revealing novel threats and providing further insight into the factors limiting reintroduction success. This information was crucial for management planning and essential for initiating the decision making process as described in chapter 5. This chapter has been published in the journal *Emu* (Taylor *et al.,* 2018).

In ***chapter 5,*** using the information gathered in chapter 4, I present a worked example of how a decision analysis approach can be employed to capture and quantify the uncertainty surrounding whether to use nest protection measures on the captive released cohort in 2017 and potential future releases. I used expert elicitation and decision trees quantitatively to represent the uncertainty attached to different management actions, and used these results to conduct informative trial experiments.

In ***chapter 6***, I summarise the key findings in my thesis, and describe how they are relevant to threatened species management not only for Australian wildlife, but for global conservation issues.

Chapter 2 ~ Is reintroduction biology an effective applied science?

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Abstract

Reintroduction biology is scientific research aimed at informing translocations and the reintroductions of threatened species. I review two decades of published literature to evaluate whether reintroduction science is evolving in its decision-support role, as called for by advocates of evidence-based conservation. Reintroduction research increasingly addresses *a priori* hypotheses, but it remains largely focused on short-term population establishment. Similarly, studies that directly assist decisions by explicitly comparing alternative management actions remain a minority. A small set of case studies demonstrate full integration of research in the reintroduction decision process. I encourage the use of tools that embed research in decision-making, particularly the explicit consideration of multiple management alternatives since this is the crux of any management decision.

From reintroduction biology to reintroduction practice

In the face of unprecedented biodiversity losses, effective strategies for the conservation of threatened species are urgently required (Lauber et al., 2011; Sutherland et al., 2012; Bainbridge, 2014). Among conservationists, there is almost universal agreement on the need for evidence-based management decisions and for science that supports conservation decision-making (Sutherland et al., 2004). However, management decisions remain primarily based on the application of experience without careful evaluation of evidence (Dicks et al., 2014; Walsh et al., 2015; Cook, 2016). For conservation management to be truly evidence-based the science should be embedded within the management problem to facilitate the choice of a best management action. Conservation science generally seeks to undertake research aimed at providing information to help choose management actions; this role should provide better outcomes than would be achieved otherwise and is our interpretation of applied science. However, most published conservation studies are not always explicit about how the information they present should be used by decision makers, and thus might not achieve a complete connection between basic and applied science (McNie, 2007; Cvitanovic et al., 2015). In general, science can support management by (i) predicting the consequences of management actions based on available evidence, (ii) reducing uncertainty around choices between alternative actions, and (iii) providing specialist tools to help select the best action for a given set of objectives. Successful examples in conservation range from experimentally testing non-lethal predator exclusion methods to protect shorebird colonies (Maslo & Lockwood, 2009) to developing software for optimal design of nature reserves at the continental scale (Ball et al., 2009).

The science of reintroduction biology showcases well these general criticisms. Reintroduction is a globally important form of conservation management, but reintroduction programs are complex and require numerous decisions, all of which are subject to uncertainty. This uncertainty in turn makes it difficult to select the ‘best’ set of actions, frequently resulting in poor choices that have been blamed for the low success of past reintroduction efforts (Wolf et al., 1998; Fischer & Lindenmayer, 2000; Germano et al., 2014). Reintroduction biology, first formally recognised as a field of science at a conference in Australia in 1993 and later published as a proceedings in 1995 (Serena, 1995), is increasingly called upon to facilitate those decisions (Seddon & Armstrong, 2016). Several authors have recommended that reintroduction studies should not just collect data from practice and seek patterns *a posteriori*, but focus on the uncertainties that make reintroduction decisions difficult and rigorously evaluate project outcomes with the aim of improvement (Lauber et al., 2011; Armstrong & Seddon, 2008; Sutherland et al., 2010, Ewen & Armstrong, 2007; Lyons et al., 2008; McCarthy, 2014; Canessa et al., 2015).

Indeed, almost a decade ago, two of us published a paper in this journal that outlined the purpose of reintroduction biology as an applied science (Armstrong & Seddon, 2008). In that paper, they argued "that reintroduction biology will progress faster if researchers focus on the questions that need to be answered to improve species recovery and ecosystem restoration. That is, reintroduction biologists should nominate the key research questions then use the best methods available to answer them, rather than addressing the questions that are most easily answered or that lend themselves to the most rigorous science." They then identified ten key questions for reintroduction biology across four levels: population establishment, population persistence, meta-populations, and ecosystems. Recognising that reintroduction biology to that date mostly focused on population establishment, they sought to encourage research across a broader spectrum of concerns. Moreover, they expressed concern that the focus on population establishment reflected the relative ease of research at that level, rather than its actual importance for improving reintroduction outcomes. Therefore, they also recommended that reintroduction biology as an applied science should address *a priori* questions that capture uncertainty directly affecting management decisions. Whether those calls by Armstrong & Seddon (2008) and similar advocates of evidence-based reintroduction (Converse et al., 2013; Moore et al., 2014), including the IUCN Guidelines for Reintroductions and other Conservation Translocations IUCN/SSC, 2013), are being heeded in the growing literature in this field remains to be ascertained.

Here, I evaluate whether the peer-reviewed published literature in reintroduction biology since its inception at a conference in 1993 and first publication in 1995 indicates an increasing effectiveness in supporting reintroduction practice. Accordingly, I seek to understand whether reintroduction studies have (1) broadened their scope beyond population establishment to support problems relating to population persistence, meta-populations, and ecosystems, (2) addressed defined *a priori* questions, and (3) whether these questions clearly provide the scientific evidence required to select a best management action.

I queried the reintroduction literature using the Web of Science citation search engine (23 November 2016 using the University College London institutional login) and specifying the key words: reintroduc* OR re-introduc* Or translocat* in the title field and monitoring OR population modelling OR experiment OR trial OR planning in the topic field and in the research areas of ‘Environmental Sciences Ecology’, ‘Biodiversity Conservation’ and ‘Zoology’ from the years 1995 – 2016 inclusive. I also queried the IUCN Global Re-introduction Perspectives book series (Soorae, 2008; Soorae, 2010; Soorae, 2011; Soorae, 2013, Soorae, 2016) and retrieved any extra peer-reviewed scientific articles cited within those case studies. I only included papers which studied vertebrates and excluded papers that were purely reviews. The search identified 309 peer-reviewed scientific journal articles from Web of Science and an additional 52 peer-reviewed scientific journal articles from the IUCN publications. I read each article fully and carefully evaluated against our criteria. To ensure reliability with categorisation, ten papers were first simultaneously judged between three of the co-authors (GT, SC & JGE) and were consistently categorised. Within the introduction I searched for statements of key questions, hypotheses and objectives and within the methods and results I searched whether or not the outcomes of more than one management action were tested. Although this is not a systematic review I believe it provides a detailed picture of reintroduction biology, with its known bias toward vertebrates (Seddon et al., 2005).

Which level of questions did the paper address?

I found 61% (219/361) of papers addressed questions at the population establishment level, 32% (117/361) at the population persistence level, 4% (16/361) at the metapopulation level, and 3% (9/361) at the ecosystem level (Figure 3). These results mirror the findings in Armstrong & Seddon (2008) who stated that the majority of reintroduction research to that point had focussed on population establishment. Analysis of the temporal trends in our dataset confirmed the lack of a clear change. Between 1995 and 2016, establishment and metapopulation studies decreased and persistence and ecosystem studies increased (in particular, studies addressing persistence in terms of genetic makeup). Multinomial logistic regression confirmed this trend but suggested the yearly rate of change was small and not statistically significant (proportional yearly rate of change, expressed by mean exponentiated regression coefficients: establishment: -1.8%; persistence: 2.2%; metapopulation: -4.9%; ecosystem: 8.3%; *p*>0.05). Most importantly, the proportion of metapopulation- and ecosystem-level studies was still less than 5% by 2017 (Figure 3). Note that although papers will often implicitly look at multiple questions, for the purpose of this review I assigned articles to only one question level, based on what was deemed the primary focus of the study.

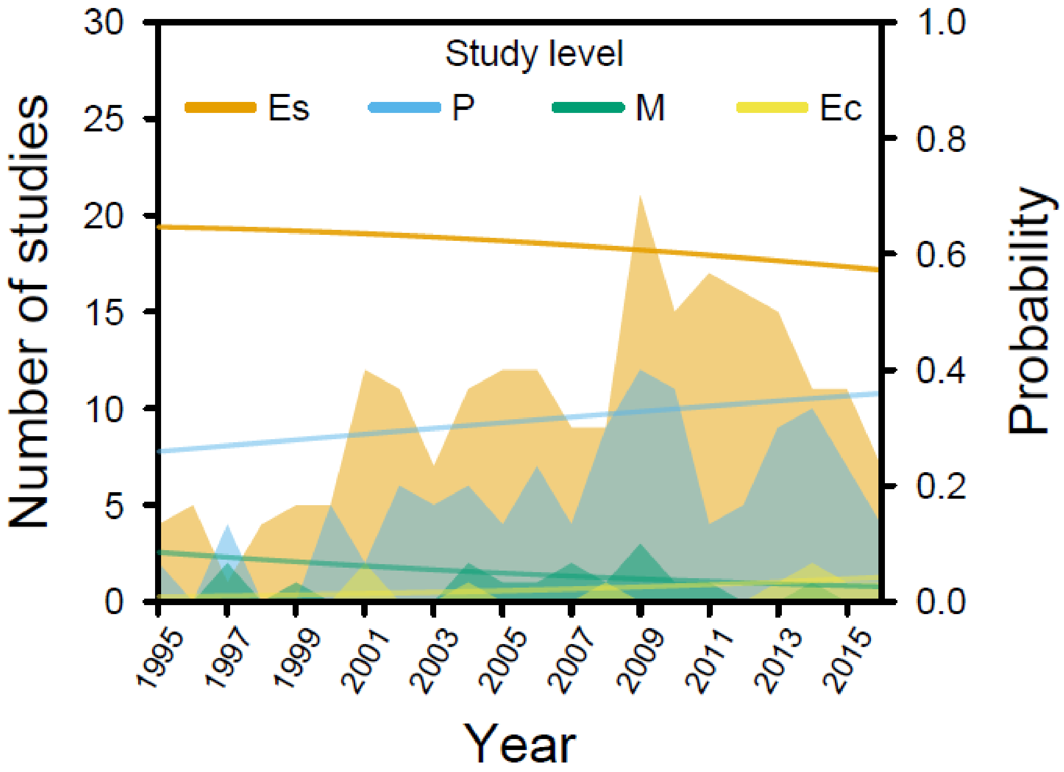

Figure 3 Temporal trend in the level of question addressed (Es=establishment, P=persistence, M=metapop, Ec=ecosystem). Shaded areas are the number of studies in each category each year. Lines are the mean probability of a study falling in each category in a given year, as predicted by multinomial logistic regression (proportional yearly rate of change, expressed by mean exponentiated regression coefficients: establishment: -1.8%; persistence: 2.2%; metapopulation: -4.9%; ecosystem: 8.3%; *p*>0.05).

Is reintroduction literature question- and management-driven?

Armstrong and Seddon (2008) argued that "questions identified *a priori* will increase the amount of useful knowledge obtained from limited conservation funds." If research does not address clearly defined *a priori* questions, it risks being purely descriptive; if it does not directly address uncertainties that are relevant to management, it risks being irrelevant for practical decision making, regardless of its potential scientific interest. To determine the extent that reintroduction literature develops *a priori* management-driven questions, I carried out two analytical steps.

First, each publication was categorised as either clearly stating *a priori* questions or not (i.e. descriptive). Second, while developing questions *a priori* moves us closer to management-driven research, management decisions normally imply a choice between alternative actions (Gregory & Failing, 2002; Moore et al., 2012). Therefore, explicitly discriminating among those actions represents the best support that reintroduction science can provide to decision makers. I categorised each of the 361 reintroduction papers into one of three categories: (A) studies that directly compared the consequences of alternative management actions, either by *a priori* predictive modelling or *a posteriori* analysis of field data (including deliberate manipulation by experiment or adaptive management); (B) studies that analysed results under one management action and assessed them without reference to alternative actions; (C) studies that did not obviously identify or assess a management action, but published scientific information that was considered valuable for conservation.

I found an almost equal split between papers that clearly stated *a priori* questions (49%,176/361), and those that did not (51%,185/361) (Figure 4). Logistic regression suggested a marked increase over the study period: the mean probability that a published study addressed *a priori* questions increased from 24% in 1995 to over 64% in 2016 (Figure 3). Only about one fifth of the reviewed articles (22%, 78/361) presented data comparing two or more management actions to directly support decision making, i.e. were in category A (Figure 5). The majority of research articles (74%, 270/361) were in category B, i.e analysed results of one management action and then made post-hoc recommendations about whether the action was suitable or not. The remaining few research articles (4%, 13/361) were in category C, making no explicit link between research and management. Multinomial logistic regression again confirmed these observed trends, with less than 1% relative yearly changes in all categories

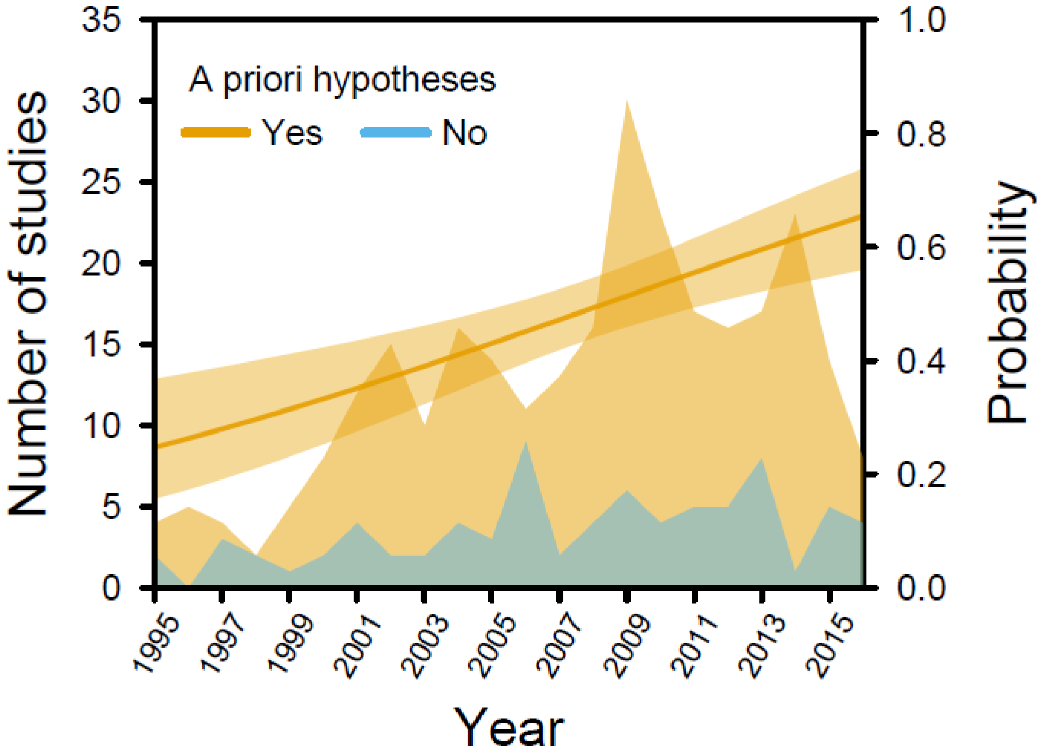

Figure 4: Temporal trend in the treatment of a priori hypotheses (yes/no). Shaded areas are the number of studies in each category each year. The solid line indicates the mean probability of a study addressing a priori hypotheses in a given year, as predicted by logistic regression (the shaded area indicates the 95% confidence interval).

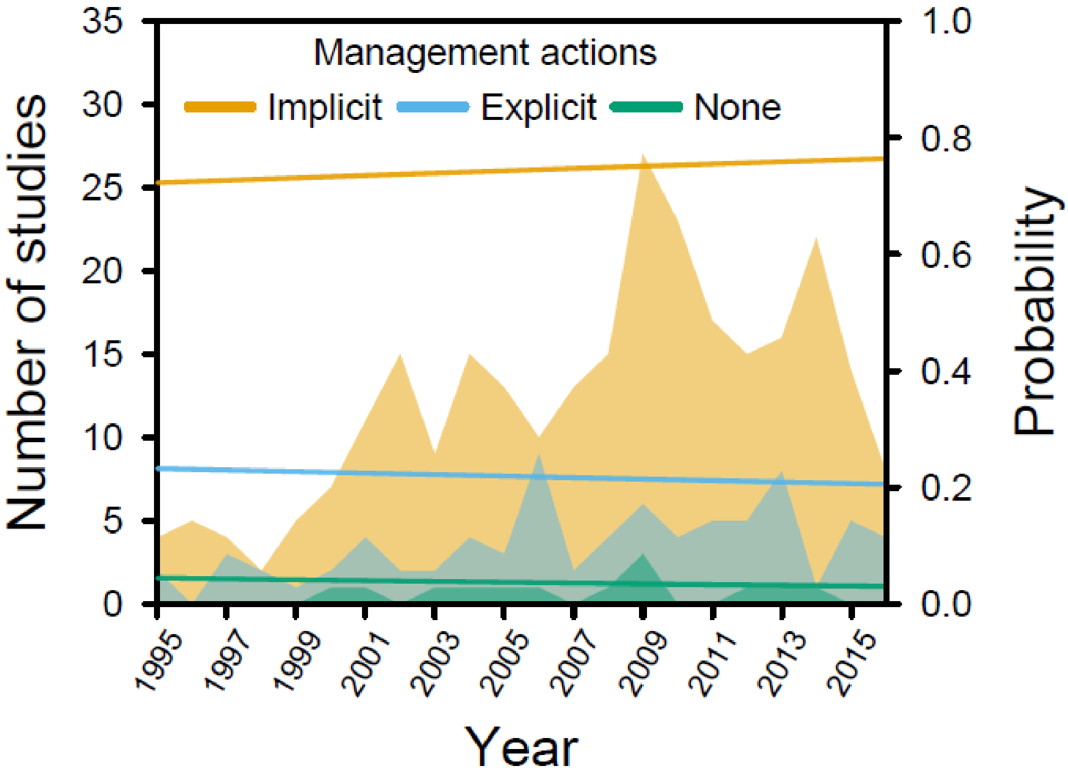

Figure 5: Temporal trend in the level of comparison of management alternatives (explicit/implicit/none). Shaded areas are the number of studies in each category each year. Lines are the mean probability of a study falling in each category in a given year, as predicted by multinomial logistic regression.

Is reintroduction biology supporting reintroduction practice?

Throughout its two-decade history, the science of reintroduction biology has repeatedly been encouraged to better support reintroduction practice (Armstrong *et al*., 1994; Osterman *et al* 2001; Armstrong *et al*., 2007; Armstrong & Seddon, 2008; Sutherland *et al*., 2010; Kemp *et al*., 2015). The publication frequency of reintroduction-related studies continues to increase, making more and more scientific evidence available to support reintroduction practice. However, this is not in itself an indication of better application: reintroduction science will not improve simply by producing more data (Armstrong &

Seddon, 2008). Rather, it requires both scientific learning through experiments, prediction and monitoring, and true integration into reintroduction practice, allowing managers to identify the actions that are most likely to achieve their objectives.

In this regard, our assessment shows that in spite of frequent calls, reintroduction biology is not reaching its full potential in providing the evidence base to support management decisions. For example, resource-demanding and technically challenging metapopulation and ecosystem studies continue to represent only a small proportion of the reintroduction literature. This practical complexity reinforces the need for clear *a priori* thinking; in this regard, it is encouraging to find an increasing proportion of studies focus on answering *a priori* hypotheses. However, whether this latter trend represents a specific improvement of reintroduction biology, or reflects the more general tendency to move away from descriptive studies, particularly in higher-profile peer-reviewed journals, cannot be discerned.

Perhaps the most important of our results is that over the last two decades there has been no appreciable increase in the proportion of studies that provide direct support for management decisions, by explicitly comparing alternative actions. In many such cases, managers and decision makers might be presented with evidence, but it is left to them to translate such information into a management decision. Only a fifth of the studies reviewed directly compared two or more possible actions (or treatment groups), either through predictive modelling prior to any practical implementation, or from interpretation of data from field monitoring or deliberate manipulation as part of the reintroduction. This limitation is likely driven by practical constraints. Many reintroductions focus on highly threatened species, where the potential for learning is limited by small sample sizes and difficulties in replication. However, these limitations reinforce, rather than diminish, the need for a strong theoretical basis for recovery plans and make the alternative trial-and-error approach even more risky (Armstrong *et al.,* 2015). Where active comparison of management actions via experiments is still considered too risky and learning is limited by other practical constraints such as small sample sizes, predictive modelling *a priori* and adaptive management (Canessa *et al.,* 2016a) can still provide guidance. In general, explicit consideration of multiple actions, including “doing nothing” options, can make even studies that directly assess only one action more relevant for management.

To summarise our findings, some encouraging trends are visible in the reintroduction literature: more studies are explicitly addressing *a priori* hypotheses. However, reintroduction biology still has great scope to better support reintroduction practice: broader-scale metapopulation and ecosystem-level studies are still rare, and most importantly, few studies explicitly focus on assisting the choice among alternative management actions, which is the ultimate requirement of decision making. The key to filling this gap is currently represented by a small set of more recent studies that illustrate clearly how to embed conservation science into practice by developing clear *a priori* questions that are immediately relevant to management, explicitly comparing two or more management actions (O’Donnell *et al.,* 2010; Gregory *et al.,* 2012; Converse *et al.,* 2013; Chauvenet *et al.,* 2012; Canessa *et al.,* 2014). An example is given below. I acknowledge that each article in our review was treated equally, regardless of its scale and the number of institutions involved, and that our inferences might have been different to some extent if these factors were taken into account.

Changes still need to occur in *what* reintroduction biology researches (expanding to a broader range of questioning spanning establishment to ecosystems) and in *how* it responds to management needs (by directly embedding within decision making). By targeting uncertainties that are relevant for management, explicitly comparing the expected outcomes of alternative actions, and managing adaptively rather than by trial-and-error, reintroduction biology can best provide the scientific evidence needed to maximise the success of reintroduction practice.

An example of a study that embeds conservation science into practice by developing clear a priori questions that are immediately relevant to management: The benefits of reintroducing ecosystem engineers back into the Australian environment for the management of wildfire (Hayward et al., 2016).

As in other parts of the world, wildfires are a natural occurrence in the Australian environment and have shaped the life-history traits of floral and faunal communities (Bliege *et al.,* 2008). In Australia, burning has been used by indigenous peoples as a traditional ecological management tool for millennia; however uncontrolled wildfires are becoming more frequent and intense, causing enormous economic, social and environmental damage. Australian terrestrial mammals such as the bilby *(Macrotis lagotis),* the numbat *(Myrmecobius fasciatus*), the woylie *(Bettongia ogilbyi)* and the boodie *(Bettongia lesueur)* (Figure 6 panel a) are considered ecosystem engineers as they alter leaf litter accumulation and breakdown. Australia has seen a dramatic decline in small terrestrial mammals, and the loss of these species, particularly fossorial species, has been hypothesised as altering wildfire behaviour through increased leaf litter accumulation. Leaf litter is a hugely combustible material that, when in abundance, can facilitate the spread and intensity of fire. An experimental study by Hayward *et al.* (2016) aimed to determine whether this loss of ecosystem engineers did lead to an increase in leaf litter and therefore an increase in fire intensity and rate of spread. The study was conducted at three Australian Wildlife Conservancy restoration sites where previously extinct fossorial species had been reintroduced into large, exotic-predator-free fenced areas. At these sites, a pair-wise, fence-line comparison was replicated (where outside fence-line represented locations with no reintroduced species). The paired sites inside and outside the fenced areas otherwise had similar vegetation and fire regimes, and data were collected on animal digging pits, leaf litter accumulation and bare ground cover. The McArthur Mk5 Forest fire behaviour model which predicts the probability of a fire starting, rate of spread, and intensity, based on environmental parameters was also applied to these sites. Results showed a significant decrease (24% (95% CI 6–43) in leaf-litter mass inside the fenced areas (in the presence of reintroduced mammal ecosystem engineers) compared to outside (no reintroduced mammal ecosystem engineers) at all the three sites (Figure 6 panel b). The fire-behaviour model also predicted that flame height would be much higher outside (1.41m) of the fenced areas compared to inside (0.37m) and that fire spread would be much faster outside fenced (0.18 km h^{-1}) areas compared to inside (0.12 km h^{-1}), equating to a 74% reduction in flame height and a 33% reduction in the rate of fire spread. This is an example of an experimental study that explicitly tests the outcomes of more than one management alternative (reintroduction of native fossorial species or absence of these species) and answers an ecosystem-level question by highlighting the beneficial impact of these management actions on ecosystem function and restoration.

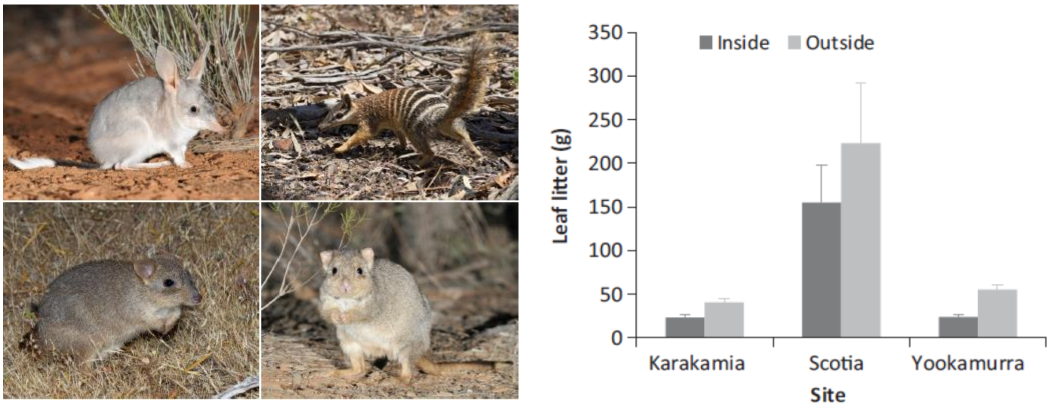

Figure 6: (Panel a) Native Australian mammalian ecosystem engineers; (top left) The bilby (*Macrotis lagotis*), (top right) the numbat (*Myrmecobius fasciatus*, (bottom left) the woylie (*Bettongia ogilbyi*) and (bottom right) the boodie (*Bettongia lesueur*) are considered ecosystem engineers which have the potential to reduce fire intensity and spread due to the alteration of leaf litter accumulation and breakdown where these species (and others) are present (Panel b). Photo credits: Bilby and Boodie - Wayne Lawler/Australian Wildlife Conservancy, Numbat and Woylie - Rohan Clarke

Chapter 3~ Factors affecting short term survival of a captive bred and released passerine bird

Abstract

The success of a reintroduction is determined, in part, by pre-release management. I examined the effect of sex, weight, age, year of release and wearing a transmitter on the survival of captive bred and released Regent Honeyeaters *(Anthochaera phrygia)* in Victoria, Australia. Birds were bred at Taronga Zoo in Sydney and release events occurred in the years 2010, 2013, 2015 and 2017. During these years, a total of 257 Regent Honeyeaters were released with 59% of these fitted with radio transmitters. Results show high levels of survival in the first 12 weeks of release; 68% - 81% of birds were still known alive each year. Survival analysis showed no effect of sex, weight and age on the survival of individuals and the top model suggested that the interaction of year of release and wearing a transmitter has the biggest effect on survival. Monthly survival estimates for birds never fitted with a transmitter were constant across years (0.78 – 0.83), but there was more variation in estimates for those wearing a transmitter (0.76 – 0.94). We also report a higher than expected rate of mortality in the first two days of release in 2017 and provide the hypothesis that transmitters negatively impact survival when environmental conditions are suboptimal and recommend this be explicitly tested in the future. The results for this study provide managers with confirmation that the captive breeding program is effectively producing birds that can

cope with the initial transition from captivity to the wild, but that conditions at the time of release play a crucial role in their short-term survival.

Introduction

The reintroduction of species into the wild is an important tool for the recovery of threatened species and as a component of broader ecosystem restoration. However, historically reintroductions have had low success rates and have often been conducted without sufficient planning to meet the management objectives (Ewen and Armstrong 2007). Reintroductions need a viable source population and are therefore increasingly coupled with *ex-situ* captive breeding programmes, particularly when it is believed that the species cannot be recovered with *in-situ* methods alone (Brichieri-Colombi *et al.,* 2018). *Ex-situ* breeding programmes involve retaining individuals from the wild source population in captivity with the aim of establishing a breeding population as insurance against extinction in the wild (Canessa *et al.,* 2014; Brichieri-Colombi *et al.,* 2018) and are distinguished from programs that hold individuals from the wild only temporarily.

The number of captive breeding projects for threatened species has been estimated at around 489, although this is now likely to have increased (Snyder *et al.,* 1996). Zoos and aquaria are the primary institutions responsible for captive breeding and safe guarding the genetic diversity of threatened species and there are numerous success stories for establishing viable captive populations (Xia *et al.*, 2014; Blanchet *et al.*, 2008; Canessa *et al.*, 2014). However for this management action to be useful and successful in reversing wild population decline, the objective must be a viable wild population as a result of the captive breeding and release program. There are examples of very successful reintroductions, such as the Griffon Vulture (*Gyps fulvus*) in France (Bosé *et al.*, 2007; Gouar *et al.*, 2008), the Black-footed ferret (*Mustela nigripes*) in the Rocky Mountains (Dobson & Lyles 2000) and the Seychelles magpie-robin (*Copsychus* *sechellarum*) (Sutherland *et al*., 2010). However reintroductions still have high failure rates (Fischer and Lindenmayer, 2000; Mathews *et al*., 2005) and it remains unclear how captive breeding and release programs are contributing to stemming biodiversity loss. Establishing a captive breeding population is expensive (Adamski and Witkowski 2007) and can take years to achieve (Xia *et al.,* 2014; Canessa *et al.,* 2016a), but when managed effectively they have the potential to produce individuals that have high survival and reproductive rates once released into the wild.

There are significant risks involved when using captive bred individuals as a source population and the chances of successful population establishment are often lower if captive animals are used compared to directly moving wild sourced individuals (Fischer and Lindenmayer 2000; Williams and Hoffman 2009). Unless all individuals are taken from the wild (Dobson and Lyles 2000) the source population for the captive stock will be a subset of the remaining wild population, potentially resulting in an additional bottleneck and negating some captive breeding objectives (Alcaide *et al.,* 2010). Populations can adapt to captivity when the selective pressures faced in the wild are removed, relaxed and/or replaced due to disease management, predator removal, food and husbandry protocols, resulting in traits that hinder establishment and persistence and recruitment in the wild (Britt *et al.,* 2004; Håkansson and Jensen 2005; Williams and Hoffman 2009). Examples of adaptation to captivity have been demonstrated in the golden lion tamarin (*Leontopithecus rosalia*) (Britt *et al.,* 2004) and the jungle fowl (*Gallus gallus*) (Håkansson & Jensen 2005). Reducing the amount of time a species needs to spend in captivity, or reducing the number of generations born in captivity can minimize the risk of genetic adaptations, however this is not always possible if the cause of decline in the wild has not yet been removed.

Various characteristics of the source population, such as age, sex and body condition, can influence how well individuals survive and reproduce in the wild and managers can use this to improve the reintroduction outcomes. For example a reintroduced population of Saddleback (*Philesturnus carunculatus carunculatus*) in New Zealand showed lower survival rates in adults than sub adults in the first two months of release and it is suggested that younger birds are more likely to settle near the release site (Masuda and Jamieson 2012). Whereas older individuals have been reported as having higher survival rates post–release in Griffon Vulture reintroduction, younger individuals often display greater behavioural plasticity and in theory have the potential to produce more offspring (Sarrazi *et al.*, 2000). On the other hand, older individuals may recruit into the population sooner and therefore make a more immediate contribution to the population. A study on translocated Black Rhinoceros *(Diceros bicornis)* recommended releasing older females as they do not exhibit reproductive delay and have higher offspring recruitment rates (Gedir *et al.,* 2018). There will be logistical and financial constraints with keeping individuals in captivity for longer periods of time. Sex differences in survival and dispersal have also been recorded in reintroduced Bison (*Bison bison*), with lower survival in males than females (Pyne *et al.,* 2010), Burrowing Bettongs (*Bettongia lesueur*) (Parsons *et al.,,* 2002) with males dispersing further than females, and in the Hihi (*Notiomystis cincta*) (Richardson *et al.,* 2010) with females dispersing further post release. Therefore assessing whether the age and sex of released individuals improves survival and reproductive success in the wild must be a key management question and focus of the post-release monitoring in order to explicitly test the hypotheses. Without explicitly testing if these methods contribute to successful establishment, reintroduction outcomes will remain uncertain. In addition, the *a priori* hypotheses must be useful for management, and not simply easy to answer if they are to reduce uncertainty, facilitate learning and increase the success of future reintroductions (Armstrong and Seddon, 2008; Sutherland *et al.,* 2010; Milner-Gulland and Shea, 2017; Scheele *et al.*, 2018; Taylor *et al.,* 2017).

One method which has significantly facilitated the monitoring of individuals once they have been released into the wild is the use of radio telemetry (Barron *et al.,* 2010; Armstrong *et al.,* 2013). Although the use of this technology dates back to the 1960s (Zenzal *et al.,* 2014) ongoing improvements have meant that now smaller free ranging species, especially birds, can be more easily monitored in challenging terrain. This enables a much better understanding of post–release survival, establishment and dispersal (Armstrong *et al.,* 2013), habitat use (Vukovich and Kilgo 2009), and breeding behaviour (Taylor *et al.,* 2018) which can further inform management to improve reintroduction outcomes. However, there is evidence that wearing transmitters can significantly change the behaviour of individuals post-release causing: increased mortality (Lewis *et al.,* 2017), decreased nesting behaviour (Barron *et al.,* 2010), decreased flight time (Zenzal *et al.,* 2014; Hooge *et al.,* 2016) and potential radio frequency radiation (Balmori 2016). This information presents a challenge for decision makers who need to evaluate the trade-off between gaining insightful information about post-release survival and reproduction and the survival cost of fitting transmitters to already threatened species.

In this study I directly address the hypotheses that are important for the management of the reintroduction program for a Critically Endangered Australian passerine species, the Regent Honeyeater *(Anthochaera phrygia)*. The captive breeding program for this species was developed to establish a captive insurance population and to release these individuals into the wild to aid the recovery of the wild population. I test the hypotheses that age, sex, weight and wearing a transmitter will affect short term survival. I predict that heavier birds would have higher survival than lighter birds and that those wearing a transmitter would have lower survival than those not wearing one. I also predict that males and older birds would have higher survival rates in the short term due to generally being larger and stronger than females and smaller individuals. These data were collected in north east Victoria from four separate reintroduction events spanning eight years in which a proportion of the released birds were fitted with radio transmitters using a backpack style harness. By explicitly testing how these factors are affecting the short term survival of captive bred and released Regent Honeyeaters, these results will help the recovery team assess how they plan and focus future management to further improve the reintroduction outcomes.

Methods

Study species and system

The Regent Honeyeater is a small (39-45 g), nectarivorous bird species that occupies Eucalypt woodlands in south-eastern Australia (Driskell and Christidis 2004; Liu *et al.,* 2014). The species has undergone substantial range contraction as a result of habitat loss, habitat fragmentation and the compounding effects of drought (Oliver 1998a; Thomas 2009). Once formerly abundant, there are approximately 100 breeding pairs in the wild (Crates *et al.,* 2017), largely confined to a handful of sites in New South Wales and Victoria. The species is classified as Critically Endangered (Garnett *et al.,* 2011, Birdlife 2013; Ingwersen *et al.,* 2015). As part of coordinated recovery actions for the species, a captive breeding program was established at Taronga Zoo, Sydney, in 1995 to provide both a captive insurance population and to serve as a source for a captive release program that seeks to bolster wild populations. Captive release trials commenced in the year 2000. Here we report on the outcomes of large-scale captive releases that subsequently took place in north-eastern Victoria (36^{o}7'S 146^{o}36'E) in the years 2010, 2013, 2015 and 2017.

All large-scale releases occurred within Chiltern Mt-Pilot National Park in north-eastern Victoria. In the years 2010 and 2013 the birds were released from a site in the northern part of the park, and in the following release years they were released at a site in the southern part of the park (Figure 7). There is very little difference in the larger scale environmental conditions at either site. The box-ironbark woodlands in this region have experienced significant disturbance; cleared for mining during the gold rush of the 19th century, for agriculture and livestock grazing and heavily harvested for timber and firewood. As a result, the remaining remnant and regrowth forest habitats have been reduced to 15% of the original extent and are now nested within a extensively managed agricultural landscape (Mac Nally *et al.,* 2000; Kelly and Mercer 2005). The Chiltern Mt-Pilot National Park consists primarily of re-growth woodlands covering an area of 21,600 ha. Despite past disturbances, the park supports a high number of threatened species considered characteristic of box-ironbark woodlands, and was historically a key foraging and breeding site for wild Regent Honeyeaters (Franklin *et al.,* 1989). Site selection for release within the park was conducted through a combination of desktop spatial analyses, field surveys and expert opinion with the following key criteria: 1) the potential for reliable nectar flows in the weeks following release, 2) a low number of aggressive interspecific honeyeaters, and 3) the presence of a water source. The presence of wild Regent Honeyeaters was also considered desirable.

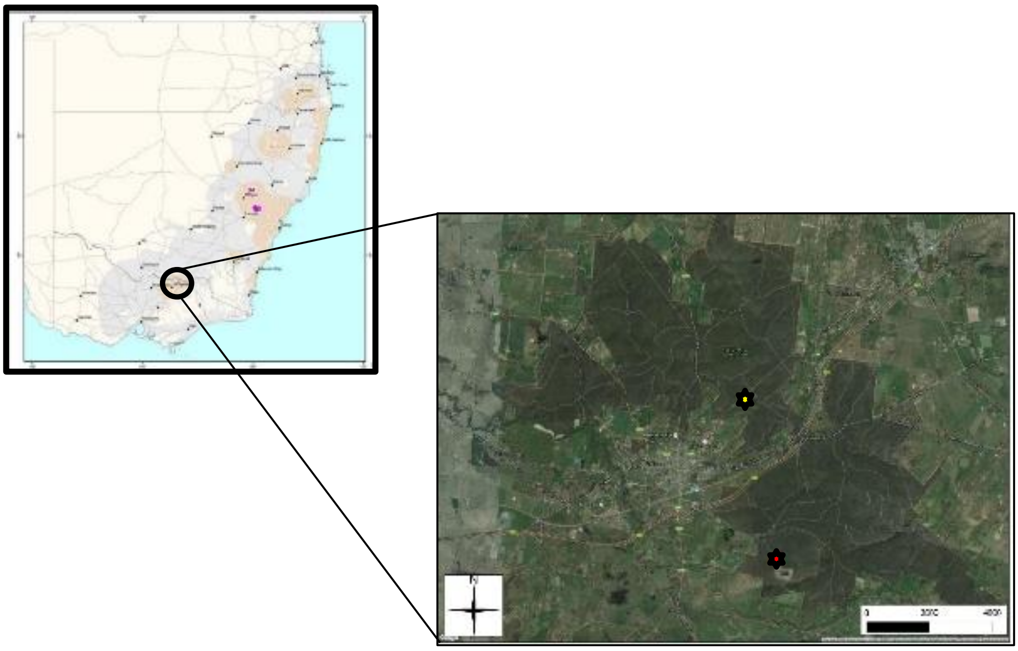

Figure 7 - Map showing release sites at Chiltern Mt-Pilot National Park in north east Victoria.

Release methods and monitoring

All birds selected for release underwent a 30 day quarantine period as well as a pre-release and transport health examination. The birds had been either surgically sexed or the sex was determined by measurements and phenotype before release (males are larger and have brighter yellow plumage) (Oliver *et al.,* 1998). One Regent Honeyeater in the 2010 cohort and another in the 2013 cohort remained unsexed at release. As the hatch date for all captive-bred individuals was known, age at release was recorded in days. All released birds were fitted with a unique combination of colour-bands; an Australian Bird and Bat Banding Scheme-issued metal band and a 'master' colour-band denoting release cohort on one tarsus and two colour-bands on the other (Figure 8a). Upon arrival at the release site, birds were held in temporary soft-sided aviaries for 24-72 hours. During this period all birds were weighed to determine if they were of a suitable mass to carry a radio transmitter post-release. BD-2 Holohill transmitters, transmitting on a unique frequency within the range of 150 to 152 mhz, were fitted to a subset of Regent Honeyeaters using a backpack style harness with a cotton 'weak point' embedded in the harness such that harness and transmitter could be shed in the event of an entanglement (Cheery & Kelly, 2008). Individual transmitters and their harness weighed ~1.8 g and did not exceed 5% of the bird's total body mass. Upon deployment a fully charged transmitter had an anticipated battery life of ~10-12 weeks (Figures 8b & 8c).

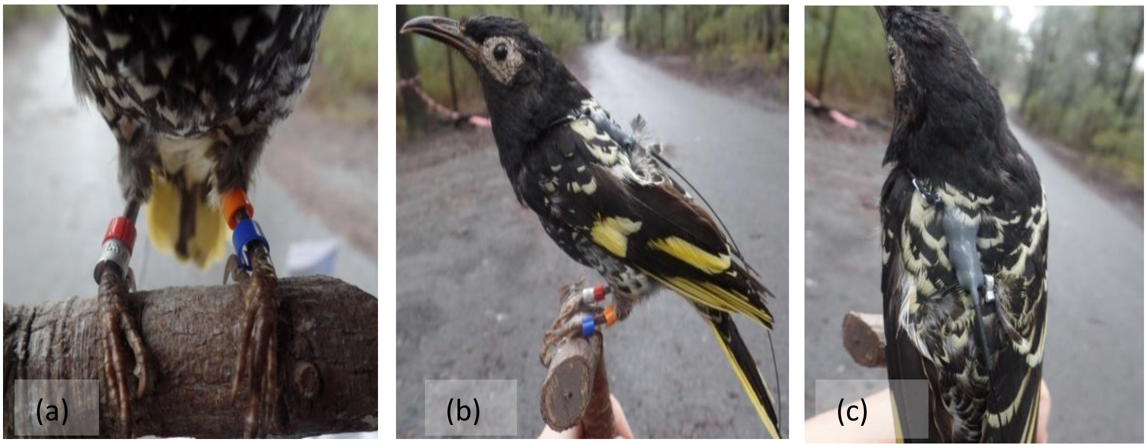

Figure 8: Taxidermy specimen of a Regent Honeyeater fitted with colour bands and BD-2 Holohill transmitter for demonstration purposes; a) band combination showing one metal band and three colour-bands; b) side view of harness and transmitter and; c) rear view

Data collection

Data collection during the post-release monitoring period extended for 19 weeks in 2010, 36 weeks in 2013, 35 weeks in 2015 and 34 weeks in 2017. This was primarily dictated by the continued presence of captive-released Regent Honeyeaters within the national park and focussed monitoring during the breeding season from August-December (See Taylor *et al.,* 2018, chapter 3). Released birds were initially monitored daily for the first month and then at least five days per week for the remaining monitoring period with the objective being to locate all transmitter birds daily and as many additional banded birds as possible.

This project relied heavily on the participation of volunteers and numbers varied daily (from 1 – 15 people a day) and throughout the whole monitoring period. Regardless of the number of volunteers, it was attempted to observe and record as many birds as possible in a day. Volunteer number was not directly linked to monitoring effort and was not included as a predictor variable in the survival analysis as differences in other variables such as quality of binoculars, volunteer experience, age and enthusiasm will have confounded the results. Handheld Australis receivers and 'Yagi' antennae were used in all releases to locate active transmitters. In forested areas radio transmitter signals were detectable at distances of up to 1.3 km though it was possible to extend this range to distances of up to 10 km from higher elevation of 500 m over open ground. If a transmitter signal could not be detected from the forest floor, then higher elevation points around the park were visited and omnidirectional antennas from moving vehicles were used. When a signal was detected we followed it to the source to confirm that that the bird was alive. The identity of detected birds was confirmed by their unique colour-band combination and a GPS waypoint was recorded. Because Regent Honeyeaters routinely associate with congeners tracking of individuals fitted with transmitters also proved to be an effective method to locate banded-only birds. On days with inclement weather, notably fog and rain, when leg bands were not easily identified, a waypoint was only taken for a bird wearing a transmitter if it was clear that the signal was moving. On days when most or all individuals known to be wearing transmitters were located quickly, any remaining field time was devoted to revisiting transmitter-wearing birds and searching novel areas to maximise the probability of detecting those individuals that were not wearing transmitters. On occasions the radio signal would lead us to a downed transmitter. In some instances it was apparent the focal bird had died owing to the presence of the carcase or partial remains. These individuals were recorded as dead in the database on the day of detection. In other instances, inspection of downed transmitters revealed a broken weak point on the harness and therefore mortality was not considered confirmed.

Data analysis

Short term ‘percentage known alive’ has been calculated and reported for the first 12 weeks post release as this is the estimated life span of the radio transmitters and is consistent for all four releases, despite monitoring continuing passed this stage for variable durations across release years. The large multi-year dataset consisted of daily sightings of every individual bird during the post-release monitoring period for each release (2010-2017). Individuals of unknown (n=2) sex were omitted form the dataset. For each release the birds in that release cohort were scored as detected (1) or undetected (0) on each survey day. We were conscious of several modelling complications and potential sources of bias, especially those associated with the effect of radio-tracking on the likelihood of observation. We outline these here, before outlining our modelling approach. The duration that individuals wore a transmitter varied; some wore them for the entire monitoring period, even once the transmitters stopped working, others were recaptured during the monitoring period and fitted, or refitted with a harness and transmitter. In order to reduce the complexity of the dataset and remove several potential sources of bias, we conducted analyses only on birds that either wore transmitters from the time of their initial release, or never at all (banded only). Our fitted effect of [' Tx_r '] is the average effect on survival of having a transmitter fitted *at the time of release*, regardless of how long the tracker lasted. This contrasts with the fitted effect of transmitter status [(‘Tx_t ')] on resighting probability, which is fitted as a time-varying individual covariate and denotes the effect of wearing a transmitter *in the current monitoring interval* on resighting probability.

We considered *a priori* that survival rates may depend on the bird's sex (male or female), body condition (proxied by release weight, grams), age at release (days), transmitter status at release (wearing or not wearing), and year of release (1=2010, 2=2013, 3=2015, 4=2017), and that resighting probability was plausibly impacted by transmitter status (wearing or not wearing) in the current monitoring interval. We therefore fitted CJS models (Cormack, 1964, Jolly, 1965, Seber, 1965) on our reduced dataset, with sex, body condition, age at release, transmitter status at release, and year of release as individual covariates in survival *(Phi)*, and individual transmitter status in the current monitoring interval as a time-varying covariate, and year of release in encounter probability *(P).* The full model matrix, including model specifications for all candidate models, is presented in Table 2. We selected models based on Akaike's Information Criterion (AIC, Akaike, 1974) and we considered models with delta- AICc < 2 to be best supported. Pairwise comparisons between the 'year of release' covariate were also calculated using Bonferroni post-hoc correction. All analyses were performed using R 3.2.2 (R Core Team, 2015) and R package RMark (Laake, 2013).

Results

A total of 257 Regent Honeyeaters were released during four release events in the years 2010, 2013, 2015, and 2017. Of those released, 152 (59%) individuals were fitted with a transmitter (hereafter 'tx-birds') whilst 105 individuals were identifiable only by their unique colour-band combination (hereafter 'banded only birds'). One release cohort had an approximately even sex ratio (1.0:0.925), whilst the other three release cohorts were female-biased (Table 1). Across all years, the mean age of released females was 469 days (90–3398 days) and the mean age of released males was 330 days (95-1628 days). Males are generally heavier than females (mean weight of released males = 44 g +/- 2.7 g; mean weight of released females = 37 g +/-2.7 g; Table 1)

Table 1: Key attributes of male and female Regent Honeyeaters released in each of four captive release cohorts at Chiltern-Mt Pilot National Park, Victoria, Australia in the period 2010-2017.

Year of release	Sex	Total released	wearing tx	Age (days)		Weight (g)		Tx days	
			(% of total)	Mean	+/- sd	Mean	+/- sd	Mean	+/- sd
2010	M	13	11 (92%)	333	237	45.9	2.3	27	44
	F	30	12 (40%)	386	278	39.8	3	39	52
2013	M	16	10 (68%)	403	497	43	2.2	81	73
	F	20	14 (70%)	540	548	38.7	1.7	42	39
2015	M	40	28 (70%)	358	238	43.5	2	54	56
	F	37	29 (81%)	413	250	37.7	3	42	39
2017	M	40	30 (76%)	271	249	44.9	2.7	71	70
	F	61	18(31%)	518	623	36.7	2	25	58

Across the four release cohorts between 68 and 81% of Regent Honeyeaters were known have survived for the duration of the 12 week short-term monitoring period. The trajectory of decline in Regent Honeyeaters known to be alive during the short-term monitoring period was similar for all years (Figure 9). The percentage known to be alive was lower than in previous years and the sharp decline in the first week in 2017 due to a number of known deaths of birds wearing transmitters (n=13), all of which occurred in the first 2 days post-release. Of these, eight were female and five were male, and they were of mixed ages.

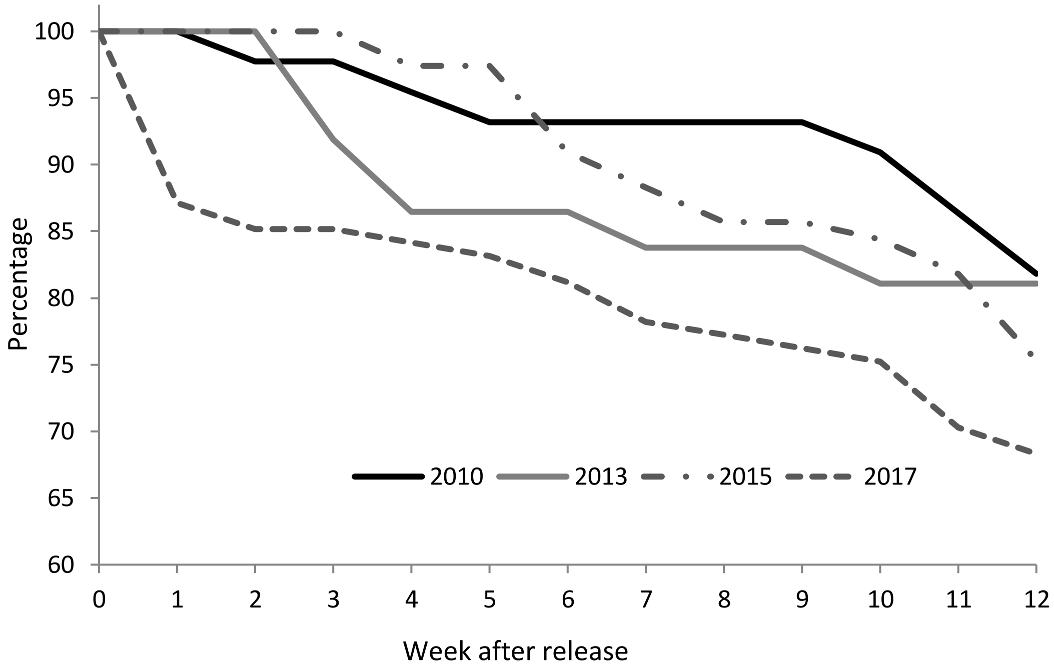

Figure 9: Percentage of Regent Honeyeaters known to be alive up to 12 weeks post release in Chiltern-Mt Pilot National Park.

The top model received 95% of the AICc weight and had a delta AICc of 0, compared to the second top model which received only 4% of the AICc weight and had a delta AICc of 6.32, Table 2. The top model suggests that that the probability of survival (*Phi*) was influenced by the year of release and the interaction of year and wearing a transmitter at the time of release, and that resighting probability *(p)* was influenced on year of release. In all years except 2017, birds wearing a transmitter had slightly higher survival rates than those not fitted with a transmitter (Figure 10). The monthly survival estimates for released Regent Honeyeaters were relatively consistent between years for those that never wore a transmitter (Figure 10). However those wearing a transmitter at the time of release showed more variation in monthly survival rates; 0.94 (95% CI 0.7 -0.91) in 2013 and 0.76 (95% CI 0.65 -0.81) in 2017. After Bonferroni correction, there was no significant difference in survival rates between years for those not wearing a transmitter. Birds wearing a transmitter had significantly higher estimated survival in the year 2013 release than any other years (Z = 5.12 – 5.68; *p* = <0.001 - <0.00001; Bonferroni critical value = 0.00041).

A priori hypotheses that the age at release, weight and sex would influence survival were not supported by any models, although the second ranked model included age and the interaction of year of release. Intuitively we hypothesised that heavier birds would experience higher survival probabilities due to higher fat reserves and improved ability to compete for resources. The mean weights of each cohort were; 2010 (42g +/- 3.9g), 2013 (40.6g +/- 2.8g), 2015 (40.6 +/- 4g), 2017 (40.1g +/- 4.6g), which were significantly different (F(3,254) = 3.236, *p* = 0.02); however, weight did not influence survival.

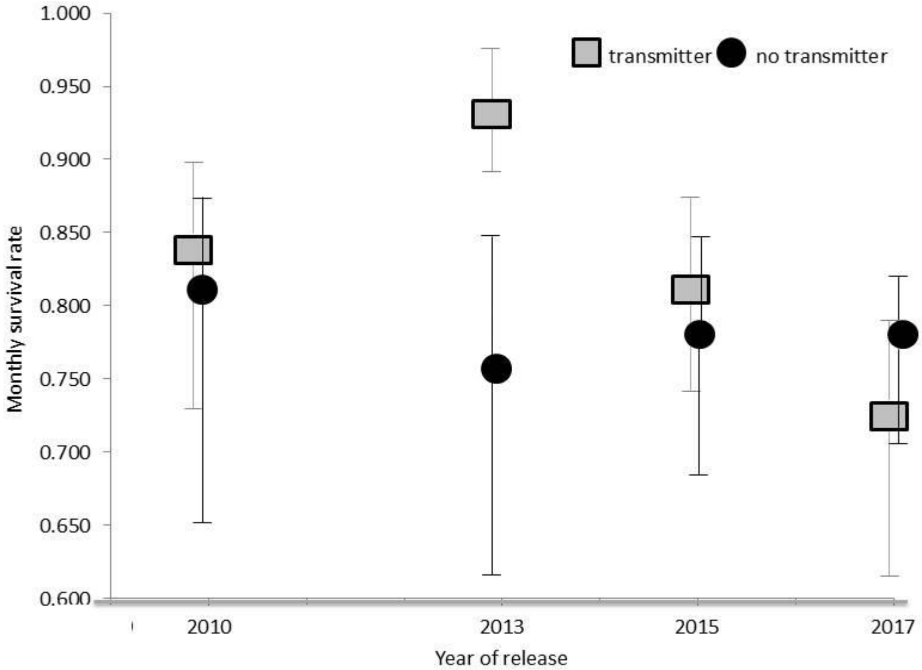

Figure 10: Monthly survival estimate (Phi) of Regent Honeyeaters released into Chiltern-Mt Pilot National Park for the release years of 2010-2017. Birds which wore a transmitter at the start of release (grey square) and those that never wore a transmitter (black circle). Error bars shows upper and lower 95% confidence intervals.

Table 2: Comparison of candidate survival models based on a priori hypotheses for Captive bred and released Regent Honeyeaters into Chiltern Mt-Pilot National Park (2010-2017). The number of predictor variables (*K*), Akaike's Information Criterion (AICc), Akaike's Information Criterion Difference (**Δ**AICc), Akaike weight (model probability) *(w_{ij})* and Deviance. Y1=2010, y2 = 2013, y3 = 2015, y4 = 2017. Tx_r = wearing a transmitter at release , Tx_t = time specific individual matrix of 'is bird wearing a transmitter or not?'. The duration of monitoring data modelled: Y1 = 19 weeks, Y2 = 36 weeks, Y3 = 35 weeks, Y4 = 34 weeks.

Model #	Model description	k	AICc	DeltaAICc	weight	Deviance
1	Phi(~y1 + y2 + y3 + y4 + y1:Txr + y2:Txr + y3:Txr + y4:Txr) p(~y1 + y2 + y3 + y4 + y1)	14	37391.459	0	0.9520803	37363.43
2	Phi(~y1 + y2 + y3 + y4 + y1:age + y2:age + y3:age + y4:age) p(~y1 + y2 + y3 + y4 + y1)	14	37397.787	6.328	0.0402315	37369.758
3	Phi(~y1 + y2 + y3 + y4) p(~y1 + y2 + y3 + y4 + y1)	10	37403.055	11.596209	0.0028879	37383.04
4	Phi(~y1 + y2 + y3 + y4 + weight) p(~y1 + y2 + y3 + y4 + y1)	11	37404.151	12.692243	0.0016695	37382.133
5	Phi(~y1 + y2 + y3 + y4 + Txr) p(~y1 + y2 + y3 + y4 + y1)	11	37404.726	13.267243	0.0012524	37382.708
6	Phi(~y1 + y2 + y3 + y4 + weight + Txr) p(~y1 + y2 + y3 + y4 + y1)	12	37404.89	13.430552	0.0011542	37380.868
7	Phi(~y1 + y2 + y3 + y4 + weight + Txr + Txr:weight) p(~y1 + y2 + y3 + y4 + y1)	13	37406.75	15.291138	0.0004552	37380.725
8	Phi(~y1 + y2 + y3 + y4 + y1:weight + y2:weight + y3:weight + y4:weight) p(~y1 + y2 + y3 + y4 + y1)	14	37408.771	17.312	0.0001657	37380.742
9	Phi(~y1 + y2 + y3 + y4 + y1:sex + y2:sex + y3:sex + y4:sex) p(~y1 + y2 + y3 + y4 + y1)	14	37409.721	18.262	0.0001031	37381.692
10	Phi(~Txr + weight + Txr:weight) p(~y1 + y2 + y3 + y4 + y1)	9	37424.444	32.985452	6.55E-08	37406.432
11	Phi(~Txr) p(~y1 + y2 + y3 + y4 + y1)	7	37424.626	33.166765	5.98E-08	37410.618
12	Phi(~Txr + weight) p(~y1 + y2 + y3 + y4 + y1)	8	37425.415	33.955971	4.03E-08	37409.405
13	Phi(~Txr + weight + sex + Txr:weight) p(~y1 + y2 + y3 + y4 + y1)	10	37426.054	34.595209	2.93E-08	37406.039
14	Phi(~1) p(~y1 + y2 + y3 + y4 + y1)	6	37426.844	35.384836	1.97E-08	37414.838
15	Phi(~weight) p(~y1 + y2 + y3 + y4 + y1)	7	37428.813	37.353765	7.37E-09	37414.805
16	Phi(~Txr + weight + age + Txr:age) p(~y1 + y2 + y3 + y4 + y1)	10	37429.009	37.550209	6.68E-09	37408.994
17	Phi(~Txr : weight : age) p(~y1 + y2 + y3 + y4 + y1)	13	37429.138	37.679138	6.26E-09	37403.113
18	Phi(~Txr : weight : sex) p(~y1 + y2 + y3 + y4 + y1)	13	37429.352	37.893138	5.63E-09	37403.327
19	Phi(~Txr : weight : age * sex) p(~y1 + y2 + y3 + y4 + y1)	21	37440.709	49.249773	1.92E-11	37398.645
20	Phi(~y1 + y2 + y3 + y4 + y1:Txr + y2:Txr + y3:Txr + y4:Txr) p(~Txt)	11	39473.874	2082.4152	0	39451.856
21	Phi(~y1 + y2 + y3 + y4 + y1:age + y2:age + y3:age + y4:age) p(~Txt)	11	39480.157	2088.6982	0	39458.139
22	Phi(~y1 + y2 + y3 + y4) p(~Txt)	7	39485.602	2094.1428	0	39471.594
23	Phi(~y1 + y2 + y3 + y4 + weight) p(~Txt)	8	39486.791	2095.332	0	39470.781
24	Phi(~y1 + y2 + y3 + y4 + Txr) p(~Txt)	8	39487.183	2095.724	0	39471.173
25	Phi(~y1 + y2 + y3 + y4 + weight + Txr) p(~Txt)	9	39487.422	2095.9635	0	39469.41
26	Phi(~y1 + y2 + y3 + y4 + weight + Txr + Txr:weight) p(~Txt)	10	39489.225	2097.7662	0	39469.21

Model #	Model description	K	AICc	DeltaAICc	weight	Deviance
27	Phi(~y1 + y2 + y3 + y4 + y1:weight + y2:weight + y3:weight + y4:weight) p(~Txt)	11	39491.148	2099.6892	0	39469.13
28	Phi(~y1 + y2 + y3 + y4 + y1:sex + y2:sex + y3:sex + y4:sex) p(~Txt)	11	39491.994	2100.5352	0	39469.976
29	Phi(~Txr + weight + Txr:weight) p(~Txt)	6	39505.927	2114.4678	0	39493.921
30	Phi(~Txr) p(~Txt)	4	39506.243	2114.7838	0	39498.24
31	Phi(~Txr + weight) p(~Txt)	5	39506.878	2115.4192	0	39496.874
32	Phi(~Txr + weight + sex + Txr:weight) p(~Txt)	7	39507.35	2115.8908	0	39493.342
33	Phi(~1) p(~Txt)	3	39508.777	2117.3177	0	39502.775
34	Phi(~Txr + weight + age + Txr:age) p(~Txt)	7	39510.472	2119.0128	0	39496.464
35	Phi(~weight) p(~Txt)	4	39510.73	2119.2708	0	39502.727
36	Phi(~Txr : weight : age) p(~Txt)	10	39510.828	2119.3692	0	39490.813
37	Phi(~Txr : weight : sex) p(~Txt)	10	39510.928	2119.4692	0	39490.913
38	Phi(~Txr : weight : age : sex) p(~Txt)	18	39523.127	2131.6682	0	39487.08
39	Phi(~y1 + y2 + y3 + y4 + y1:Txr + y2:Txr + y3:Txr + y4:Txr) p(~1)	10	41757.095	4365.6362	0	41737.08
40	Phi(~y1 + y2 + y3 + y4 + y1:age + y2:age + y3:age + y4:age) p(~1)	10	41763.251	4371.7922	0	41743.236
41	Phi(~y1 + y2 + y3 + y4) p(~1)	6	41768.848	4377.3888	0	41756.842
42	Phi(~y1 + y2 + y3 + y4 + weight) p(~1)	7	41770.137	4378.6778	0	41756.129
43	Phi(~y1 + y2 + y3 + y4 + Txr) p(~1)	7	41770.455	4378.9958	0	41756.447
44	Phi(~y1 + y2 + y3 + y4 + weight + Txr) p(~1)	8	41770.892	4379.433	0	41754.882
45	Phi(~y1 + y2 + y3 + y4 + weight + Txr + Txr:weight) p(~1)	9	41772.7	4381.2415	0	41754.688
46	Phi(~y1 + y2 + y3 + y4 + y1:weight + y2:weight + y3:weight + y4:weight) p(~1)	10	41774.33	4382.8712	0	41754.315
47	Phi(~y1 + y2 + y3 + y4 + y1:sex + y2:sex + y3:sex + y4:sex) p(~1)	10	41775.178	4383.7192	0	41755.163
48	Phi(~Txr + weight + Txr:weight) p(~1)	5	41789.472	4398.0132	0	41779.468
49	Phi(~Txr) p(~1)	3	41789.608	4398.1487	0	41783.606
50	Phi(~Txr + weight) p(~1)	4	41790.27	4398.8108	0	41782.267
51	Phi(~Txr + weight + sex + Txr:weight) p(~1)	6	41790.739	4399.2798	0	41778.733
52	Phi(~1) p(~1)	2	41792.07	4400.6109	0	40750.016
53	Phi(~Txr + weight + age + Txr:age) p(~1)	6	41793.826	4402.3668	0	41781.82
54	Phi(~weight) p(~1)	3	41794.024	4402.5647	0	41788.022
55	Phi(~Txr : weight : age) p(~1)	9	41794.281	4402.8225	0	41776.269
56	Phi(~Txr : weight : sex) p(~1)	9	41794.352	4402.8935	0	41776.34
57	Phi(~Txr : weight : age : sex) p(~1)	17	41806.727	4415.2682	0	41772.685

Discussion

This study tested hypotheses about the factors affecting short-term survival of a captive bred and released passerine species. The reintroduction program has the advantage of a long-term dataset, which spans nearly 10 years and includes daily sighting records for 257 birds, from four releases. This extensive data collection is an unusual opportunity to explicitly analyse whether age, sex, weight and wearing a transmitter impact the reintroduction outcomes in the immediate post-release phase. The percentage of released birds still alive 12 weeks post release was high in all years (68-81%), providing evidence that the captive breeding program was producing birds able to transition into the wild successfully, at least in the short-term.

The top model indicates that environmental conditions at the time of release have a significant impact on the immediate survival of released birds. If conditions are suboptimal, fitting birds with transmitters are likely to cause higher mortality rates than expected. In years 2010, 2013 and 2013 birds wearing transmitters at the time of release had slightly higher survival rates than those not wearing transmitters. These results were unexpected, but similar results have been reported in other studies, such as those found by Richardson *et al*. (2013) in the New Zealand Hihi *(Notiomystis cincta).* This may be as larger, heavier birds are fitted with transmitters and these individuals cope better than smaller, lighter birds when conditions are good; however the effect of weight was not captured or directly tested in the models as larger birds were preferentially fitted with transmitters.

In 2017 there were 13 birds found dead within two days of the first cohort being released; all these birds were wearing transmitters and all banded only birds had been observed alive. This event provided an opportunity to learn and reassess current management. Firstly it shows that even after a number of releases and consistent management routines, unforeseen problems can still occur. It was hypothesised that a lack of nectar availability at the release site caused birds to lose weight overnight and those wearing a transmitter had the added weight to contend with in the mornings, resulting in mortality. In this instance, without the transmitters the dead birds were unlikely to be found on the forest floor and this information would never have been known; however, it is possible that fitting transmitters to these birds did contribute to the mortalities. This supports the need for post-release monitoring despite the confidence of previous releases and results of high short term survival. Fitting birds with transmitters facilitates monitoring and improves the ability to detect birds once released, as well as providing opportunity for public engagement and training. The hypothesis that transmitters negatively impact survival when environmental conditions are suboptimal will need to be explored further by management if using transmitters are planned for future releases. Data on temperature during the period of each release are available from the local weather station (Rutherglen weather station); however temperature alone will not suffice to account for environmental conditions. Short term environmental conditions do not account for flowering intensity and nectar production in Eucalypts (Birtchnell and Gibson 2008), and other variables such as rainfall, soil moisture, humidity, fire intensity and periods of drought will all contribute to nectar availability. We recommend that future studies prioritise longer term environmental data analyses to obtain a clearer understanding of the effect of environmental conditions on the survival of released birds.

This presents a trade-off situation for the recovery team who will need to weigh up the benefits of fitting transmitters which facilitate detection and encourage community engagement, against the monetary cost of fitting transmitters and the risk of mortality when conditions are sub optimal. The data suggest that monitoring would be equally effective without using transmitters as the fitted effect of transmitter status did not affect resighting probability.

Habitat loss and fragmentation is the leading cause of decline in the Regent Honeyeaters and remaining habitat across their range is degraded. The reduction in suitable habitat has led to further threatening processes such increased noisy minors and made them more vulnerable to predation (Crates *et al.*, 2018). Although this study only looks at short term survival immediately post–release, results from this study indicate that conditions at year of release have the greatest impact on survival probability. Knowing that remaining habitat is degraded, future management would benefit from a better understanding of the environmental variables which directly influence short term survival. In doing so, management alternatives, such as providing supplementary feeding (Rooney *et al.,* 2015; Doerr *et al.,* 2017a), can be focussed to mitigate against these factors and increase survival once released.

We hypothesised that the age and sex of the individuals would influence their survival once released into the wild, however models of apparent survival rates showed no support for these variables. These results are comparable to other bird reintroductions where no differences between ages and sexes of individuals were found: North Island Kokako *(Callaeas wilsoni)* (Bradley *et al.,* 2012), The Blue-fronted Amazon Parrot *(Amazona aestiva)* (Lopes *et al.,* 2017) and Griffon Vultures *(G. fulvus)* (Gouar *et al.,* 2008). This is positive news for the recovery team as it shows that current captive breeding pre-release management does not have to attempt to keep birds longer in captivity in order to release older birds or to produce a skewed sex ratio for release, which would be expensive and require additional space. However, a study by Crates *et al.* (2018) found a male bias in the adult sex ratio in wild Regent Honeyeaters (1.18 males per female) and that one in six males are unable to find a mate. They hypothesise this lack of females is limiting population recovery and suggest the strategic release of captive bred females into the wild population. Although my data does not support the hypothesis that females have higher mortality than males and producing more females in captivity is unlikely a feasible option, releasing females where unpaired males are known may be an option the recovery team wish to explore and explicitly test for future releases.

This study provides evidence that the captive breeding of Regent Honeyeaters is producing individuals that transition well from captivity into the wild and have high survival rates in the immediate duration. Excluding year 2017, the percentage of birds still alive after 3 months was above 75%, which is higher than short term survival rates reported from other bird reintroduction studies; 37% in Saddlebacks *(Philesturnus carunculatus)* (Masuda and Jamieson 2012), 50% in Brown Treecreepers *(Climacteris picumnus)* (Bennett *et al.,* 2012) and <50% in Puaiohi (*Myadestes palmeri*)(Soorae 2013). This information is important for management; however research must then stretch to longer term survival and other demographic rates such as reproduction which ultimately influence population persistence and contribute to population recovery.

Breeding species in captivity is an expensive and risky management action, but increasingly required to safeguard populations of species which are facing extinction in the wild. Monitoring is essential to assess reintroduction outcomes and understand if progress is being made towards the reintroduction objectives. Pre-release monitoring of predators and habitat suitability will ensure the chosen release site is appropriate for release. Habitat variables can be monitored after the reintroduction event to continue assessing if the state of the system remains suitable for the released population. Post release monitoring in the immediate phase is extremely useful for managers to monitor survival as a function of demographic rates. When *a priori* hypotheses are integral in the development of the monitoring component, the data collection can explicitly focus on what is most useful for managers and successful outcomes. The information learned will then reduce the uncertainties and can inform better release management.

Chapter 4 ~ Video monitoring reveals novel threat to Critically Endangered captive bred and released Regent Honeyeaters

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Abstract

Nest predation is a primary cause of nest failure in open cup nesting woodland birds and low reproductive success is a common reason that reintroduced species fail to establish in the wild. We used video monitoring to record the breeding outcomes and identify the causes of nest failure in a reintroduced population of the Critically Endangered Regent Honeyeater. We intensively monitored 28 nesting attempts of 13 pairs during the 2015 breeding season and found that the probability of individual nest success was 0.21 (from egg laying to fledging). We report for the first time Sugar and Squirrel Gliders depredating Regent Honeyeater nests. In addition to losses attributed to predation, a high proportion of chicks died in the nest from unknown causes. Our results show that rates of nest initiation and success are low in reintroduced Regent Honeyeaters, and future reintroductions should attempt to mitigate the threat of nest predation. Other sources of nest failure and barriers to nest initiation and egg laying are priority areas for future research.

Introduction

Reproduction is a key vital rate determining the demographics of populations. There are numerous external influences that can reduce nest success in birds (here defined as the proportion of nests that fledge at least one young), including extreme weather (Jovani & Tella 2016), limited resources (Sherley *et al.*, 2014), competition (Frei *et al.,* 2015), brood parasitism (Wei *et al.*, 2015), parasites (Scott-Baumann and Morgan 2015) and anthropogenic habitat disturbance (Ibáñez-Álamo *et al.*, 2015). However, perhaps the most important driver of nest failure is predation (Ricklefs, 1969; Major *et al.*, 2014). Predation risk to eggs, nestlings and attending adults impact a variety of behaviours (e.g. nest construction and position) (Lee & Lima, 2016) and influence the evolution of life history traits (Martin, 1995) to ultimately shape population dynamics and densities (Lahti, 2001). When predation is the main driver of decline for a threatened species, management actions are often focused on reducing predation risks. For example, predator removal (Armstrong *et al.,* 2002) and predator exclusion methods (Major *et al.*, 2014) have been employed to reduce predation and therefore increase adult survival and reproductive output. Seeking to maximise reproductive success is particularly important when attempting to establish or reinforce a population through the release of breeding adults. Predation of nests is known to be a major limiting factor for establishment success in reintroduced populations (Moseby *et al.*, 2015; Ashbrook *et al.*, 2015) and this risk may be further elevated when releasing captive bred individuals due to their naivety to predation pressures in the wild (Moseby *et al.,* 2015).

Here, I report findings from nest monitoring of captive bred and released Regent Honeyeaters (*Anthochaera phrygia)* during an initial post-release breeding event in 2015 at Chiltern-Mt Pilot National Park in northeast Victoria. This species has been subject to intensive recovery actions over the last two decades in response to a precipitous decline in population size. Although once widespread across the woodland belt of south eastern Australia, Regent Honeyeaters have lost >85% of their primary habitat through land clearing (Mac Nally *et al.,* 2000), and sightings are now largely restricted to regions in New South Wales (Bundarra-Barraba, the Hunter Valley and the Capertee Valley) and north-east Victoria (centred around the Chiltern-Mt Pilot National Park). The total wild population was estimated to number 350-400 individuals in 2010 (Garnett *et al*., 2010), with further subsequent declines likely. Previous releases at Chiltern-Mt Pilot in 2008, 2010 and 2013 indicated that nest success was low (D. Ingwersen unpubl. data). As the drivers of this low reproductive success have remained largely unknown, here I sought to explicitly identify the factors limiting nest success.

Methods

Study species and site

The Regent Honeyeater is a Critically Endangered, nectarivorous passerine endemic to south eastern Australia. The breeding season typically occurs from August to January (the Austral spring and summer). Regent Honeyeaters are open cup nesters, with the nest built solely by the female. Regent Honeyeaters lay 2-3 eggs per clutch and have an incubation period of 14 days and a nestling period of ~16 days (Oliver *et al*., 1998). Both parents feed the chicks. Although nests are often spatially aggregated, Regent Honeyeaters do not appear to be synchronised breeders (Oliver *et al*., 1998). Regent Honeyeaters are often associated with riparian habitat during the breeding season (Geering and French 1998; Crates *et al.,* 2017).

Chiltern-Mt Pilot National Park (36°7’59.00”S 146°36’4.00”E), was the chosen release site for all releases (2008, 2010, 2013, 2015 and 2017) because wild Regent Honeyeaters have historically used this area for breeding, and wild individuals are occasionally still observed here. It covers 21,600 ha of primarily regrowth open box-ironbark forest of which roughly 4,600 ha in the northern section comprises the core habitat, predominantly Mugga Ironbark (*Eucalyptus sideroxylon*), for Regent Honeyeaters. The northern section of the park also supports Red Stringybark (*E. macrorhyncha*) and box eucalypts (*E. albens, E. macrocarpa and E. polyanthemos*). It is located on the traditional lands of the Dhudhuroa-Waywurru and Pangerang people (Blake and Reid 2002). Seventy-seven Regent Honeyeaters (36 female and 41 male) of mixed ages (39 were < 1 yr, 31 were between 1 and 2 yrs and 7 were between 2 and 3 yrs) were selected for release from birds bred at Taronga Zoo. None of the birds had prior breeding experience in captivity.

Nest location and monitoring

All released birds were fitted with unique combinations of colour bands. Forty-two of these birds (19 female, 23 male) were also fitted with Holohil systems BD-2 radio transmitters using a backpack style harness incorporating a weak point designed to break when exposed to resistance or wear. The transmitters weighed no more than 5% of the bird’s body weight. The release occurred in April 2015, three months prior to any anticipated breeding events, and timed to coincide with the commencement of seasonal flowering of key eucalypt food plants species. The average battery life of functioning radio transmitters was 10-12 weeks, so transmitters were redeployed at intervals during the release such that at any point in time a selection of birds could be tracked. Over the course of the release 59 of the released birds wore a functioning transmitter at least once, with eight of those refitted with transmitters two or three times.

Established pairs were identified on the basis of intense calling and territory defence by the male, both birds displaying courtship positions (lowered straightened body with wings slightly opened), and subsequent close association when foraging, inspecting potential nest sites, and nest building. Once pairs were identified they were monitored daily and their nest attempts followed. Nest building was confirmed when the birds regularly took material to the same place and a clear base of a nest was seen (they often took one or two sticks to a site before ceasing activities at that site). A complete nest was defined as a nest where adult attendance at that nest continued beyond the nest building stage. All nests were discovered during the early nest building stage providing confidence that, amongst monitored pairs, few if any nests were overlooked. For each nesting attempt I recorded the pair ID, the tree species in which the nest was built, height of nest and nest tree height, distance to water, degree of visual concealment, clutch size and nest outcome (Table 3). I assessed the degree of visual concealment from each cardinal direction at a distance of 2m from the base of the nest tree, acknowledging that nest height can impact the accuracy of our concealment estimate. I estimated the percentage, to the nearest 5%, of the nest that was concealed by foliage with the mean of these four values providing a relative measure of nest concealment.

Modified video surveillance cameras (Network 4 Channel AHD DVR Kit with 4 x 720p Cameras) with DVR monitors were used to monitor ten nests (eight that received eggs and two that did not). Each camera was connected to an 18m cable and fixed to a 6m extendable pole. The batteries and DVR were housed in a 780 x 380 x 380mm cargo box at the base of the tree, minimising the climbing required and therefore disturbance.

Cameras were only deployed on completed nests and then only if they were in a position that allowed easy and safe access to the tree with minimal disturbance to the breeding pair. Cameras were always positioned 3-4 metres from the nest, which still enabled good quality footage to identify predators. No vegetation or other potential forms of concealment were modified. After a camera had been installed, nests were observed from a distance of 10+ m once per day. All video footage was studied in order to record any rare behaviours, and if it appeared the nest had been abandoned (e.g. absence of the parents or reduced visitation rates), I scrutinised the video footage to identify the time and cause of predation. Nestlings that were found dead in the nest were stored at ~4^{o}C and air-freighted to Taronga Zoo for post-mortem (n=3 chicks from two broods). In one instance, footage showed the adults removing dead chicks and this allowed recovery of the bodies.

Statistical analysis

An initial basic model for constant daily survival rate (DSR) from laying to fledgling or failure of Regent Honeyeater nests (based on a 30 day nesting period) was estimated using the R-package 'RMark' v2.2.2 (Laake *et al.*, 2016), an R- interface for the nest survival model (Dinsmore *et al.*, 2002) in the software program 'MARK' (Cooch and White, 2005). I then included concealment and height in a second and third model respectively to calculate if DSR varies with these covariates. Only those nests that reached the egg stage were included in analyses. As two pairs reached the egg stage twice, I first ran all models with the complete data set and then re-ran the models after excluding the second of each of these nests to assess the influence of repeated measures. Akaike's information criterion corrected for small sample sizes (AICc) was used for model selection (Shaffer, 2004). Means +/_ one standard deviation are presented throughout this paper.

Results

Twenty-eight nesting attempts by 13 pairs (26 individuals as all pairs remained unchanged through the study) were recorded during the 2015 breeding season. There was a mean of 2.2 +/- 1 nests per pair (range 1-4). Ten of these nest attempts, from seven different pairs, were subsequently filmed. Two nests that were filmed never received eggs, and two nests that reached the egg stage were not filmed. In total 10 nests reached at least the egg stage and were used in our DSR analysis.

Timing of breeding

The first pairing was confirmed on 1st July 2015, 78 days after the birds were released. At this date 78% of the released individuals (60 of 77 birds), and 86% of those known to be alive (60 of 69 birds) were being regularly sighted. Most pair bonds were confirmed during August (54%, 7/13). By the end of August almost half of all released birds (45%, 35/77) were no longer being detected in the area, most likely due to a combination of mortality and dispersal. For example by 31st August 2015 10 transmitters had been recovered in settings that indicated the focal bird had died (e.g. a mass of feathers and/or bones). Once a pair had secured a breeding territory, the male typically ceased to call and the pair became increasingly difficult to detect. We therefore assume breeding attempts from additional unmonitored pairs occurred. The first nest to reach the egg stage was recorded on 23rd August 2015, 131 days after birds were released.

Table 3: Information collected on the site characteristics of each nest attempt by all the 13 pairs followed through the breeding season. Native: locally native species or introduced exotics. Stage reached: N = nest, E = egg, C = Chick, F = fledged. Cause of failure: M = mammal, A = Avian, D = Died in nest, U = Unknown. (Inferred) meant we didnot capture it on footage, however (1) a broken egg was retrieved and (2) heavy rain was very likely the cause

Pair ID	Nest attempt	Native (0=no, 1=yes)	Tree species	Height (m)	Dis water (m)	% canopy cover	Clutch size	Stage reached	Cause of failure	Camera deployed
1	1st	1	*Eucalyptus macrorhyncha*	12	10	15	1	E	M (inferred)	No
	2nd	1	N/A (stump)	<1	8	0	2	1 fledged, 1 depredated	A	Yes
	3rd	1	*Melaleuca* sp.	6	30	75	3	C	D	Yes
2	1st	1	*E. sideroxylon*	8	17	50	n/a	N	U	No
3	1st	1	*E. macrorhyncha*	11	32	60	2	E	M	Yes
4	1st	1	*E. sideroxylon*/*E. albens* hybrid	12	75	25	1	E	M	Yes
	2nd	0	*Hedera* sp. (Ivy)	3	5	100	>1	C	D	No
5	1st	1	*E. sideroxylon*	16	53	50	n/a	N	U	No
6	1st	1	*E. blakelyi*	10	40	100	n/a	N	U	No
	2nd	1	*E. sideroxylon*/*E. albens* hybrid (mistletoe)	14	150	25	n/a	N	U	No
	3rd	1	*E. polyanthemos*	12	100	20	n/a	N	U	No
7	1st	1	*E. macrorhyncha*	7	300	0	n/a	N	U	Yes
	2nd	0	*Acer negundo*	2	7	75	n/a	N	Heavy rain (inferred)	Yes
	3rd	0	*Quercus* sp.	10	10	75	n/a	N	U	No
8	1st	1	*E. macrorhyncha*	14	321	30	n/a	N	U	No
	2nd	0	N/A (hanging flower pot)	2	2	75	3	C	D	Yes
9	1st	1	*E. sideroxylon*	14	100	25	n/a	N	U	No
	2nd	1	*E. sideroxylon*	14	100	25	n/a	N	U	No
	3rd	1	*E. sideroxylon*	10	30	25	n/a	N	U	No
	4th	1	*E. polyanthemos*	5	33	0	n/a	N	U	No
10	1st	1	*E. sideroxylon*	8	50	50	n/a	N	U	No
	2nd	1	*E. macrorhyncha*	12	67	45	n/a	N	U	No
11	1st	1	*E. albens*	13	50	30	n/a	N	U	No
	2nd	0	*Cytisus proliferus*	3	20	100	1	E	A	Yes
	3rd	0	*C. proliferus*	2	50	100	n/a	N	U	No
12	1st	0	*Photinia robusta*	4	10	75	n/a	N	U	No
	2nd	0	*Cedrus* sp.	6	20	95	1	C	D	Yes
13	1st	0	Unknown (exotic sp.)	4	30	80	2	F	-	Yes

Characteristics of nesting sites

Nest building typically took 4-5 days, followed by a day with little activity before egg laying took place. The mean height of nests was 7.6 ± 4.8 m (range: 0.7 m in a dead stump to 16 m in a Mugga Ironbark). A total of 13 different tree and shrub species were used as nest sites (Table 3). The most commonly used tree species for nesting were Mugga Ironbark (7 of 28 nests) and Red Stringybark (5 of 28 nests). The mean distance of Regent Honeyeater nests from surface water was 61 ± 76 m.

Causes of nest failure

Of the 28 nest attempts monitored (both with and without video surveillance), 18 (64%) were abandoned before eggs were laid, four (14%) failed at the egg stage, four (14%) failed at the nestling stage, and two fledged young. For the 10 active nests (those that reached egg stage) the best supported DSR model was our null model, although a second model with concealment was also equally plausible (ΔAIC < 2; but less than half as well supported based on model weights (Table 4 & 5). Given a lack of influence from our predictor variables we calculated DSR from the null model as 0.95 ± 0.002, giving a nest survival probability over a 30 day nesting period of 0.21 (0.95^30= 0.21).

Table 4: Model selection results for captive bred regent honeyeaters in Chiltern Mt-Pilot National Park, 2015.

Model	Npar	AIC_c	ΔAIC_c	Wi	Deviance
S(Constant)	1	49.57	0.00	0.51	47.53
S(Concealment)	2	51.23	1.66	0.22	47.13
S(Height)	2	51.64	2.07	0.18	47.54
S(Concealment + Height)+Concealment*Height	4	53.16	3.59	0.08	44.80

Table 5: Beta coefficients of covariates included in regent honeyeater nest survival models. Data from Chiltern Mt-Pilot National Park in 2015. (N=8 nests)

Covariate	**Estimate (β)**	**SE**	**95% CI**	
			Lower	Upper
Constant	5.84	2.67	0.62	11.06
Concealment	-1.51	1.01	-3.49	0.46
Height	-0.48	0.58	-1.62	0.66
(Concealment + Height)+Concealment*Height	0.23	0.19	-0.14	0.61

Five of the pairs abandoned all nest attempts and were never observed to reach the egg stage, whereas eight pairs reached at least the egg stage before failure. Causes of failure or abandonment before an egg was laid could not be ascertained. Three categories of failure at the egg or nestling stage were identified; these were mammalian predation (3/10; (egg stage) Sugar Glider (*Petaurus breviceps*) and Squirrel Glider (*P. norfolcensis*)), avian predation (2/10; (nestling stage) Australian Magpie *(Cracticus tibicen*), (egg stage) House Sparrow (*Passer domesticus*)) (Figure 11) and unexplained nestling mortality (i.e. chicks found dead in the nest or ground: 4/10). (Note that one nest had one chick predated and one chick that escaped and fledged, hence nine nests were subject to predation and two nests that successfully fledged one or more young). The results of post mortems were inconclusive for chicks found dead in nests (Taronga Zoo, unpublished data).

Video monitoring captured important information that would likely have been otherwise missed and/or misinterpreted. Both females whose eggs were predated by nocturnal marsupials returned to the nest the following morning and continued to briefly adopt a sitting position within the empty nest that resembled incubation. Without the camera this behaviour would have inferred that nest failure occurred during subsequent daylight hours. A female Regent Honeyeater was also shown to defend her nest by continuing to incubate despite attempts by a Sugar Glider to get underneath her. The glider eventually abandoned its attempt, but the eggs were predated the following night by Squirrel Gliders.

Figure 10: Top panel: Selected video frames showing a Sugar Glider (*P. breviceps*) climbing on the branch where a Regent Honeyeater is incubating (circled), before flushing her off the nest and consuming the egg.
Bottom panel: Selected video frames showing an adult male Australian Magpie *(C. tibicen*) attacking two Regent Honeyeater nestlings. In the second frame a Regent Honeyeater can be seen defending the nestlings (circled).

Discussion

This study is the first to provide detail on the breeding outcomes of captive bred and released Regent Honeyeaters. Furthermore, I report the first records of native Sugar and Squirrel Gliders as nest predators of this species. Nest survival was worryingly low at 0.21 and 64% of nest attempts never reached the egg stage. Video footage has provided important information on the impact of predation on the nest success of released birds as well as documenting adult behaviour that would have otherwise been misinterpreted.

Nest abandonment prior to egg laying is not unique to Regent Honeyeaters, yet remains poorly understood (Beckmann and Martin 2016; Flegeltaub *et al.,* 2017). This may reflect a mix of inexperienced breeders in the captive-released cohort and the greater intensity of tracking individual captive-released pairs (aided by transmitters) through an entire breeding season. Furthermore, we know from observations in captivity that breeding pairs will frequently initiate multiple nests before settling and completing one (Taronga Zoo pers. comm.). Given that I did not film nests during nest building we are unable to offer additional insight into the drivers of this abandonment. Further monitoring to establish the causes of failure during the nest building phase may have merit.

Predation by native species was the principal cause of nest failure where eggs or nestlings were present. This is consistent with many previous studies that identify nest predation as a substantial threat to open-cup nesting bird species (Beckmann & McDonald 2016; Stojanovic *et al.*, 2014). A single nest failure was attributed to the activities of an introduced avian species when a House Sparrow was filmed destroying eggs. Sugar Gliders have previously been identified as a significant, novel threat to Swift Parrots *(Lathamus discolour)* in Tasmania, where the glider is an introduced species

(Stojanovic *et al.,* 2014). I am not aware of any observations where Sugar Gliders or Squirrel Gliders have previously been recorded depredating the eggs of bird species within their native range. Stojanovic *et al*. (2014) also reported that Sugar Gliders killed and ate incubating female Swift Parrots. Whilst no predation of adult Regent Honeyeaters by gliders was recorded in our study, video footage does document two instances where gliders lunge at an incubating female Regent Honeyeater, indicative of a possible predation attempt.

I also recorded nestlings found dead in four nests with no obvious sign of predation or predator-caused nest abandonment. This is common in birds and can be attributed to various factors such as exposure to extreme weather, disease and parasites (Smith *et al*., 1998) and limited food availability (Jovani and Tella 2016). In order to investigate if nestling mortality could be due to extreme weather conditions I retrieved the daily temperatures for the week prior to chicks dying in the nest. The mean maximum temperature in the week prior to chick death for the first two nests was 26.6^{0} C +/-3.7^{0} C, (max 32.4^{0} C), for the third nest it was 27^{0} C +/-2.9^{0} C (max 32.4^{0} C) and for the fourth nest it was 32.1^{0} C +/-3.3^{0} C (Max 36.6^{0} C) (BOM, 2017). None of these mean maximum temperatures were substantially higher than the means for their respective months (Oct 26.9^{0}-C and December 31.7^{0} C). In addition, whilst post mortems were inconclusive, no apparent sign of disease was detected (Taronga Zoo unpublished data). On this basis limited food availability was considered a more plausible driver of nestling mortality than either extreme temperatures or disease. Assessment of this food limitation hypothesis is the focus of on-going research.

Whilst I acknowledge the role that human disturbance may play in nest failure, we are confident that the presence of researchers and the placement of cameras was not a significant driver of nest abandonment or failure. Cameras were only placed near nests when nest building had been completed. The birds were of captive origin and habituated to the presence of humans since hatching, however we ensured that nest-building attempts were observed from a distance. In addition, there was no evidence from the video footage that visiting predators were aware of, or attracted specifically to, the cameras.

The seemingly poor nest success we have recorded in reintroduced Regent Honeyeaters is concerning. There are no known records of nest success rates prior to significant population declines; however, previous studies on the breeding biology of wild Regent Honeyeaters have reported much higher rates than found in this study: 46% (Geering and French 1998) and 38.3% (Oliver *et al.,* 1998). Crates *et al.* (2018) studied nest success in wild Regent Honeyeaters and compared the results to historical and contemporary accounts of not only Regent Honeyeaters, but other honeyeater species, concluding that current estimates for Regent Honeyeaters are comparatively lower. It may be that captive bred birds, with no prior breeding experience, are particularly naïve to nest building, nest defence, and feeding of young in wild settings. Surviving birds might therefore be expected to improve in future breeding attempts. However recent observations of wild breeding birds have also reported high rates of failure, seemingly due to predation (Crates *et al*., 2018). Taken together these observations suggest poor reproduction is a proximate limiting factor for the Regent Honeyeater population and not solely related to birds in our study being captive-bred and reproductively naïve.

Knowing that a low reproduction rate may be an important limiting factor in the recovery of Regent Honeyeaters means it can become the focus of management. My study offers critical insight into the causes of nest failure and suggests targets for possible intervention. We have highlighted two separate areas of concern. Firstly we provide direct evidence for predation by mammals and birds. Secondly, I document nestling mortality that did not appear to be related to disease or temperature extremes and may be due to starvation. Developing interventions and testing their utility requires care and needs to consider the objectives of management (Canessa *et al.,* 2016c). Furthermore, appropriate interventions need to consider other affected groups and species. For example, control of predators through culling or translocation is unlikely to be acceptable because most identified predators were native species, and some such as the Squirrel Glider are considered regionally threatened. Alternatively, management actions may consider strategies such as predator surveys in the planned release area to assess predation risk (Chalfoun and Martin 2009) or barriers at nests that prevent or reduce predator access (Homberger *et al.,* 2017).

Either way, managers of the Critically Endangered Regent Honeyeater are now better informed regarding the causes of poor reproductive success and are thus better positioned to develop, deploy and monitor an appropriate management strategy.

Chapter 5 ~ Making risky decisions in species recovery: what to do about poor nest success in Regent Honeyeaters

Abstract

Making transparent and rational decisions for managing threatened species in situations of high uncertainty is difficult. Managers must balance the optimism of successful intervention with the risk that intervention could make matters worse. I show how using a bespoke decision tree to represent expert opinions of uncertain outcomes can clarify the decision process and be used to target learning to improve predictions. I study the management of poor nest success in a reintroduced population of the critically endangered Regent Honeyeater *(A. phrygia)* in South-Eastern Australia and show how uncertainty hinders the choice of nest management (alternatives include anti-predator collars and cages or supplementary feeding). Initial expert opinions showed limited value in supplementary feeding but optimism in how management could reduce predation (expert opinion was that mean probability of predation would reduce from 0.64 to 0.49 using collars or from 0.64 to 0.13 using collars and cages). Combined with risks of desertion and other causes of mortality this led to nest collars being perceived as the best option under uncertainty (leading to an estimated 22% nest success on average; range 9.2-35.7% compared to a mean success rate of 15.6%; range 6.0-28.9% under no management). However, learning about the effectiveness of anti-predation actions using an artificial nest experiment reduced expert optimism of the effect of management (mean probability of predation increased from 0.49 to 0.53 using collars and 0.13 to 0.38 using collars and cages). Although collars remained the preferred option, albeit with slightly reduced effectiveness (20.3% nest success on average; range

10.3-33.5%), our scenario analysis suggests that learning more about the perceived risk of nest desertion by breeding adults would provide the next biggest potential gain for nest management. Overall this approach provides transparency in decisions by clearly articulating confidence in the effectiveness of management options and shows the value of targeted learning, in this instance using artificial nests.

Introduction

Threatened species commonly require management to support their populations in the wild. This can be in the form of discrete actions such as the eradication of non-native predators from areas such as islands (Wilson *et al.,* 2017) , or providing on-going support for persistence of threatened populations. Examples of on-going management include predator control/exclusion (Major *et al.,* 2014; Maggs *et al.,* 2015; Wilson *et al.,* 2017), provision of breeding/nesting sites (Libois *et al.,* 2012), supplementary feeding (Ewen *et al.,* 2015) and nest protection (Homberger *et al.,* 2017).

Before these management actions are implemented, an *a priori* hypothesis of the factors limiting population growth should have been developed, ideally based on previous monitoring of the system and species. However, actively providing supportive management to threatened species can be difficult as knowing what needs to be addressed does not equate to knowing how best to address it. Making important management choices has to balance the optimism that intervention can improve the situation with the risk of making matters worse. Faced with uncertain outcomes conservation practitioners are typically risk averse, preferring options that minimise poor outcomes over seeking options that may provide excellent gains (Tulloch *et al.,* 2015). Conservation practitioners frequently engage an expert to implement what they believe is the best management solution, based on intuition (Armstrong and Ewen 2001; Gregory and Failing 2002). Whilst this approach can achieve positive on-ground outcomes, management based on intuition means our response to uncertain and risky outcomes is poorly defined, lacks transparency, and is not structured to allow monitoring of uncertainties to improve our choices over time.

Articulating uncertainty as quantitative distributions of our recovery objectives allows for transparent assessment of management alternatives (Canessa *et al*., 2016a) and the development of strategic monitoring to best inform our future choices (Ewen and Armstrong 2007). Furthermore it removes confusion arising from qualitative descriptions of risks and benefits (linguistic uncertainty; Kujala *et al.,* 2013) or a false sense of accuracy as provided by expert intuition (precise but often inaccurate expert opinions; (Englich and Soder 2009; Burgman 2015). Experts provide critical sources of information used to produce quantitative summaries of uncertain outcomes, but only when best practice approaches are used (Hoey *et al.,* 2016). Experts can also suggest valuable monitoring methods to reduce uncertainties that hinder our management choice (Canessa *et al.,* 2016b).

An important target of conservation is the attempt to improve poor breeding success in threatened bird species. Poor breeding success is often linked to predation (Que *et al.,* 2015; Guppy *et al.,* 2017; Taylor *et al.,* 2018), naivety of individual breeders following reintroduction (Moseby *et al.,* 2015), lack of food (Rooney *et al.,* 2015; Doerr *et al.,* 2017), and the associated consequences of genetic drift and inbreeding (Caughley 1994; Stephens and Sutherland 1999). It follows that there may be opportunities to assist populations through the breeding season with management aimed at reducing one or more of these threats. For example, successful protection of eggs and nestlings from predators using a type of exclusion barrier has been achieved in passerines (Major *et al.,* 2014), Giant Ibis (*Thaumatibis gigantea)* (Keo *et al.,* 2009) and shorebirds (Isaksson *et al.,* 2007). However unexpected negative outcomes have also been reported, such as increased predation on nesting adults (Anderson *et al.,* 2003; Isaksson *et al.,* 2007) and potential nest abandonment (Cohen *et al.,* 2016). The potential for either positive or negative outcomes from nest protection can therefore make it difficult to choose the best management alternative and the reason we explore and trial these options before implementing them. In the field experiments reported in this study, I use two types of nest protection methods; nest cages and 'possum' collars. As previously mentioned, nest cages/exclusion barriers have been used before to protect nest, more commonly ground nesting birds. Our cages were designed to be attached to branches (see methods). Possum collars are commonly used throughout Australia to prevent possums and other arboreal marsupials from accessing the tree canopy. There are frequently seen in towns and cities to manage urban populations and are commercially available, usually made from thick plastic or metal bands which are wrapped around the tree trunks. There are no studies to date which have trialled and quantified the effectiveness of these types of nest protection methods.

In this study I show how a quantitative representation of uncertainty can clarify expert opinion on the efficacy of nest management alternatives and direct strategic field monitoring to improve these expert opinions. I study a reintroduced population of the critically endangered Regent Honeyeater in South-eastern Australia. My previous work has highlighted that breeding success in this population is extremely poor, with predation by native mammals and birds being a key factor (Taylor *et al.,* 2018). Here I worked with experts to propose a range of possible management solutions. Using a bespoke decision tree to represent our uncertainty in management outcome this study demonstrates: (i) how experts believe management alternatives will influence nest success, (ii) how uncertainty and different scenarios create sensitivity in the optimal management choice, (iii) where learning will provide the largest possible improvement in nest success, and (iv) the application of field trials that target learning on these key uncertainties.

Methods

Framing the nest management decision problem

Detailed information about the Regent Honeyeater status and recovery program is provided in Chapters 1, 3 & 4. Here, I focus on the key objective of the Regent Honeyeater Recovery Team, who aim to ensure that the reintroduced population at Chiltern Mt-Pilot National Park successfully establishes. Whilst post-release survival, at least in the short term, has been high (Chapter 3), Regent Honeyeater breeding success has been very poor (Taylor *et al.,* 2018; Chapter 4). Taylor *et al.* (2018) monitored breeding success over one year and found high levels of nest predation by avian and mammalian species (including threatened natives such as Squirrel Gliders, *Petaurus norfolcensis*) and nestling death for unknown reasons. Causes of poor nest success mirror results in the remnant wild population in New South Wales (Crates *et al.,* 2018*)* and confirm the concerns of the recovery team. As a result, the recovery team is considering how best to improve nest success whilst not compromising the status of other threatened species in the park.

Using experts to develop a range of alternative management actions

A summary of all available data on captive bred and released Regent Honeyeater nest success (opportunistic reports and Taylor *et al*., 2018) was presented to seven species experts and additional experts including past and current members of the recovery team and those who work closely with the team from Australian National University, NSW government (Office of Environment and Heritage) and The University of New England.

Experts were chosen based on their extensive research, knowledge and publications in the field of woodland bird conservation in Australia. I asked the experts to consider information on the causes of nest failure and to think about answers to the following questions: (1) *What management could we do to reduce mammal predation of eggs and nestlings?* (2) *What management could we do to reduce bird predation of eggs and nestlings?* (3) *If it is true that nestlings are starving, then what management could we do?* And (4) *If it is not starvation that is causing nestlings to die then what do you think it is AND what management could we do?*

Elicitation of expert opinions

With limited study of nest success (Taylor *et al*., 2018; Chapter 4) and no available study of the effectiveness of suggested management solutions, I parameterised the decision tree by formalising current expert opinion. I used the modified Delphi method over two rounds to obtain expert opinion (from the same seven experts as above) for each chance node of the decision tree (Figure 12) (following methods detailed in Davies *et al.,* 2018). The modified Delphi method is a structured communication technique presented to experts in which they can answer questions in two or more rounds and update their answers after being presented with an anonymous summary of the replies. Before eliciting opinions, I provided the feedback received during development of the possible management actions to the experts to ensure they had the same set of background information. For each chance node, I elicited a set of three values from each expert: the most likely value and the lowest and highest values such that the expert would be 100% confident that the true value would be found between those bounds. Estimates by the seven experts were then averaged to obtain a unique set of values for each chance node to reflect uncertainty, defined as a mean most likely value, mean lowest and mean highest bounds (Table 6).

Combining objectives, alternatives and uncertainty in a decision tree

These distinct alternative management actions are best represented as a decision tree (Figure 13). Decision trees provide a graphical representation of a decision process (Behn and Vaupel 1982; Canessa *et al.,* 2016a). In this case the decision tree starts with a decision node with four main branches: the first three correspond to actions suggested by managers (protect nests with a collar and cage, protect nests with only a collar, supplementary feed breeding pairs) and a branch under which no nest management is carried out (used as a reference). Each decision branch is followed by the same series of chance nodes, each associated with a given probability of a negative outcome: probability that parents desert the nest, probability that nest will be predated, probability that nestlings would starve, probability that nestlings would die for other reasons. Nests that survive all these chance events are considered a success. For simplicity, here I do not include a combined strategy of feeding and nest protection.

Data analysis: Solving the decision tree with expert opinion

I created a decision tree using the program DecisionTools Suite 7 (Palisade, US), an add-in set of tools for MS Excel®. For each chance node, I used the mean lowest, highest and most likely values to fit a beta-PERT distribution (a continuous probability distribution which is defined by minimum, most likely and maximum values that a variable can take) to the estimates, a distribution specifically developed for the treatment of expert-elicited information (Vose 1996). I then drew a random value from each of these distributions and calculated the discrete outcome for each branch of the tree (nest success), given by; $NS_a = p_{1\text{-}d}(a) \cdot p_{1\text{-}p}(a) \cdot p_{1\text{-}s}(a) \cdot p_{1\text{-}0}(a)$

Where NS_a is the probability of nest success of action a and is the product of four probabilities, $p_{1\text{-}d}(a)$ is the probability that the nest will not be deserted under action a, $p_{1\text{-}p}(a)$ is the probability that the nest will not be predated under action a, $p_{1\text{-}s}(a)$ is the probability that the nestlings will not starve under action a, $p_{1\text{-}o}(a)$ is the probability that the nestlings will not die for other reasons under action a. This was repeated 10,000 times, using the Monte Carlo simulation function within DecisionTools Suite 7, with each iteration drawing a random set of values from the chance node distributions, and generating a distribution of nest success outcomes for each management action taking full account of uncertainty.

In addition to this numerical treatment of the uncertainty of expert-elicited values, I carried out a more general sensitivity analysis in two ways. First, I explored a “status quo” scenario where I assessed how nest survival with no management changed across a range of values from our chance node inputs. Those input nodes having a greater influence on variation in nest success are those most important to learn about (i.e. reducing uncertainty in the most influential chance node rewards us with the largest reduction in uncertainty about nest success). Second, I ran two additional scenarios by repeating the simulation run above with slightly different sets of input values for our chance nodes. In scenario two (‘no *additional* desertion risk scenario’) I simulated the possibility that cages did not affect nest desertion rates, by replacing the probability of desertion in the cage branch with the corresponding value used in the no management action branch (Table 6). In scenario three (‘predation risk not reduced’) I simulated a scenario in which neither cages nor collars would reduce the chance of predation, by replacing the probability of predation in the cage and collar branches with the corresponding value used in the no management action branch (Table 6). These three scenarios allowed me to explore the predicted nest success outcomes, respectively (1) without a desertion risk but with reduced predation and (2) with desertion risk and no reduction in predation.

Table 6: Mean ‘low’, ‘best’ and ‘high’ expert opinion values for each chance node in the decision tree for each of the four decision tree scenarios. Expert opinions were obtained from seven experts using a modified Delphi approach. Input values for the decision tree that change from the initial run are highlighted in bold.

	Collar	Collar + Cage	Feeding	Status quo
Expert opinion on outcome of management				
Desertion	0.24 (0.14-0.35)	0.61 (0.44-0.81)	0.24 (0.14-0.35)	0.24 (0.14-0.35)
Predation	0.49 (0.31-0.75)	0.13 (0.07-0.38)	0.64 (0.48-0.84)	0.64 (0.48-0.84)
Starvation	0.28 (0.16-0.41)	0.28 (0.16-0.41)	0.17 (0.06-0.27)	0.28 (0.16-0.41)
Chicks die	0.18 (0.09-0.32)	0.18 (0.09-0.32)	0.18 (0.09-0.32)	0.18 (0.09-0.32)
Scenario: Desertion no problem				
Desertion	0.24 (0.14-0.35)	**0.24 (0.14-0.35)**	0.24 (0.14-0.35)	0.24 (0.14-0.35)
Predation	0.49 (0.31-0.75)	0.13 (0.07-0.38)	0.64 (0.48-0.84)	0.64 (0.48-0.84)
Starvation	0.28 (0.16-0.41)	0.28 (0.16-0.41)	0.17 (0.06-0.27)	0.28 (0.16-0.41)
Chicks die	0.18 (0.09-0.32)	0.18 (0.09-0.32)	0.18 (0.09-0.32)	0.18 (0.09-0.32)
Scenario: Predation not improved				
Desertion	0.24 (0.14-0.35)	0.61 (0.44-0.81)	0.24 (0.14-0.35)	0.24 (0.14-0.35)
Predation	**0.64 (0.48-0.84)**	**0.64 (0.48-0.84)**	0.64 (0.48-0.84)	0.64 (0.48-0.84)
Starvation	0.28 (0.16-0.41)	0.28 (0.16-0.41)	0.17 (0.06-0.27)	0.28 (0.16-0.41)
Chicks die	0.18 (0.09-0.32)	0.18 (0.09-0.32)	0.18 (0.09-0.32)	0.18 (0.09-0.32)
Updated opinions on outcome of management after nest experiment				
Desertion	0.24 (0.14-0.35)	0.61 (0.44-0.81)	0.24 (0.14-0.35)	0.24 (0.14-0.35)
Predation	**0.53 (0.39-0.74)**	**0.38 (0.24-0.58)**	0.64 (0.48-0.84)	0.64 (0.48-0.84)
Starvation	0.28 (0.16-0.41)	0.28 (0.16-0.41)	0.17 (0.06-0.27)	0.28 (0.16-0.41)
Chicks die	0.18 (0.09-0.32)	0.18 (0.09-0.32)	0.18 (0.09-0.32)	0.18 (0.09-0.32)

Validating expert opinion through field experiments

Our initial exploration of the decision tree indicated the decision largely depended on the trade-off between the effectiveness of collars and cages in preventing predation, and the risk that they might increase the rates of nest desertion. Therefore, there was value in learning more about these two parameters. However, the recovery team considered that an experimental study involving live birds would run the risk of desertion, and this would not be acceptable unless a potential reduction in predation could be demonstrated beforehand. Therefore, artificial nests with eggs were used to experimentally quantify any changes in predation rates that would occur by using a cage or collar. Experiments were run within Chiltern Mt-Pilot National Park using the same methods but done at different times. In each experiment, artificial nests (N=40 in cage experiment; N=48 in collar experiment) consisting of wicker canary nests lined with coconut fibre on the inside and sphagnum moss on the outside were used. Each nest contained three eggs; two commercially available quail eggs (of a suitable size to simulate Regent Honeyeater eggs) and one hand moulded plasticine egg. Plasticine eggs were used as teeth and beak marks would be imprinted in the eggs and would help identify the predators as at least a mammal or avian. Half of the nests in each experiment were assigned as treatment nests, and the other as control nests and these were paired. The nests were attached to trees using fine tie wire in locations similar to those typically used by Regent Honeyeaters. Nest heights varied but were restricted to a maximum of 3.5 m given safe working heights of available ladders. Each member of a nest pair was placed at a similar height and within a similar setting (e.g. fork of a branch, within epicormic growth or on a horizontal branch) and was approximately 50-100 m from the other nest.

In the nest cage experiment the cages were fitted to the treatment nest of each pair, Table 7 shows details of trees used for cage experiment. The nest cages were constructed of 25x25 mm wire mesh made into a cylindrical shape with a diameter of 50 cm and height of 55 cm (Figure 12A). To accommodate for the potential of a nest predator entering the nest cage and becoming stressed and/or trapped, the base and 'lid' were attached in a fashion that allowed applied pressure to open them from the inside but not from the outside (essentially acting as a one-way door). The cage was fitted to the branch by cutting the cage walls upwards in a 'zipper' fashion that would allow them to straddle the branch. Once the walls sufficiently straddled the branch the bottom was attached and the cage secured in a manner that would withstand the elements. Once secured to the tree, three 10x10cm holes were cut into the wire wall to (hypothetically) allow a Regent Honeyeater to enter and exit the cage (Figure 12A). These holes were strategically placed so as to not readily facilitate the entry of potential nest predators along branches. A subsample of nests (N=9 treatment; N=9 control) were fitted with a Reconyx Hyperfire HC600 camera trap in order to identify the predator and any attempts to get past the cage. The trial lasted 14 days in total to replicate the Regent Honeyeater incubation period. During this time the nests were checked daily for signs of predation, which was defined as any event which caused one of the quail eggs to become unviable (i.e. was broken and/or removed from the nest). Individual nests were removed from the study once nest predation was detected.

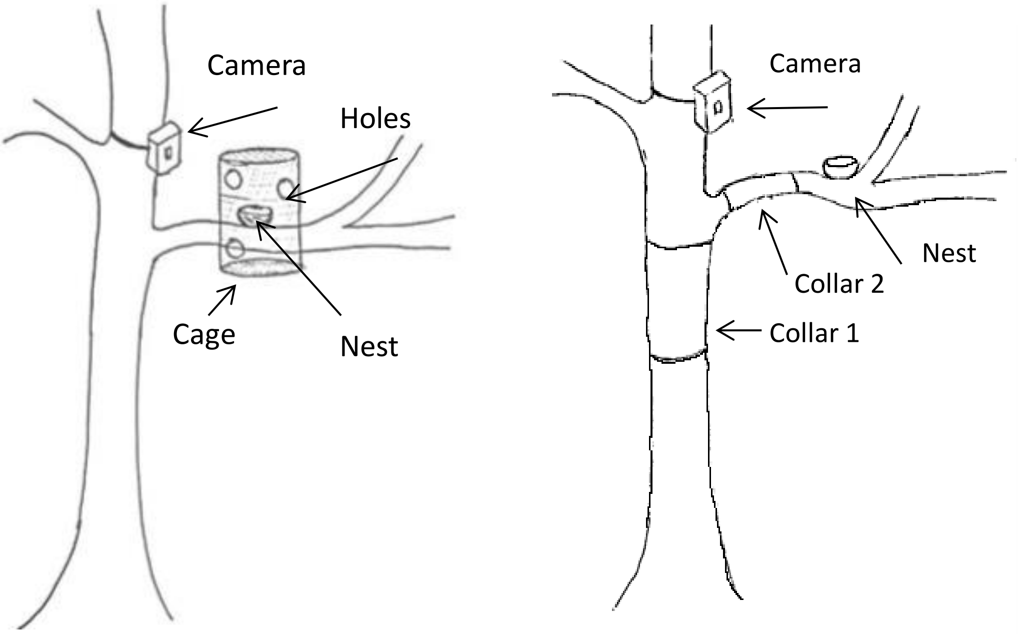

Figure 12: Diagrams showing set up of the (A) cages and (B) collars on the artificial nests. For simplicity the natural foliage giving concealment has not be shown and the distance from camera will vary.

In the nest collar experiment the collars were a sleeve shaped ‘possum collar’ made from commercially available polycarbonate sheets that we fitted around the tree trunk (Figure 12B). Our choice of collar was based on observation that this hindered Sugar Gliders more than conical type designs (personal observation). On treatment nests two collars were attached, one on the branch between the artificial nest and the trunk of the tree, and a second around the trunk of the tree before the primary union with the branch holding the artificial nest, table 8 gives details of trees used in collar experiment. The polycarbonate collar was cut to the size of the trunk and branch then secured in place by folding around on itself and attached using strong Velcro. All nests in the treatment group were fitted with Reconyx Hyperfire HC600 camera traps. All control nests were checked the day after they were deployed, by climbing the tree and any that showed signs of predation were removed. Any that were still active were subsequently checked using an extendable pole and mobile phone to capture a short video recording. Treatment nests were monitored passively for two weeks using camera traps.

Table 7: Details of trees used in cage experiment. T= Treatment (cage) and C=Control (no cage)

Tree #	Pain #	Nest height (m)	Tree (non specific)
1	T1	1.1	Small Eucalypt
2	C1	1.1	Small Eucalypt
3	T2	3.07	Small Eucalypt
4	C2	3.1	Small Eucalypt
5	T3	2.2	Small Eucalypt
6	C3	2.6	Small Eucalypt
7	T4	1.4	Small Eucalypt
8	C4	1.6	Small Eucalypt
9	T5	3.2	Small Eucalypt
10	C5	3.2	Small Eucalypt
11	T6	2.7	Small Eucalypt
12	C6	2.5	Small Eucalypt
13	T7	3.1	Small Eucalypt
14	C7	3.2	Small Eucalypt
15	T8	1.7	Exocarpus
16	C	1.7	Exocarpus
17	T9	3.3	Medium Eucalypt
18	C9	3.3	Small Eucalypt
19	T10	2.2	Small Eucalypt
20	C10	2.2	Small Eucalypt
21	T11	1.4	Small Eucalypt
22	C11	1.4	Small Eucalypt
23	T12	3	Small Eucalypt
24	C12	2.2	Small Eucalypt
25	T13	2.1	Small Eucalypt
26	C13	3.2	Small Eucalypt
27	T14	1	Exocarpus
28	C14	1	Exocarpus
29	T15	1.7	Small Eucalypt
30	C15	1.7	Small Eucalypt
31	T16	2.4	Small Eucalypt
32	C16	2.4	Small Eucalypt
33	T17	3	Small Eucalypt
34	C17	3.1	Small Eucalypt
35	T18	1.7	Small Eucalypt
36	C18	1.9	Small Eucalypt
37	T19	2.8	Small Eucalypt
38	C19	2.7	Small Eucalypt
39	T20	3	Medium/Large Eucalypt
40	C20	3.1	Medium Eucalypt

Table 8: Details of trees used in collar experiment. T= Treatment (collar) and C=Control (no collar)

Tree #	**Pair #**	**nest Height (m)**	**Species**
1	C1	15	Eucalyptus blakelyi
2	T1	8	E. blakelyi
3	C2	16	Eucalyptus albens
4	T2	14	E. albens
5	C3	10	E. blakelyi
6	T3	9	E. blakelyi
7	C4	6	E. polyanthemos
8	T4	6	E. polyanthemos
9	C5	10	Eucalyptus macrorhyncha
10	T5	13	box hybrid
11	C6	12	E. albens
12	T6	10	E. albens
13	C7	15	E.macrorhyncha
14	T7	10	E.macrorhyncha
15	C8	8	E.macrorhyncha
16	T8	8	E.macrorhyncha
17	C9	8	box hybrid
18	T9	8	E.macrorhyncha
19	C10	14	Eucalyptus sideroxylon
20	T10	14	E. sideroxylon
21	C11	11	box hybrid
22	T11	12	E. polyanthemos
23	C12	11	E. albens
24	T12	11	E. albens
25	C13	12	E. sideroxylon
26	T13	14	E. sideroxylon
27	C14	10	E. polyanthemos
28	T14	13	box hybrid
29	C15	13	box hybrid
30	T15	10	box hybrid
31	C16	12	E. albens
32	T16	10	E. albens
33	C17	14	box hybrid
34	T17	14	E.macrorhyncha
35	C18	9	E. albens
36	T18	11	box hybrid
37	C19	6	box hybrid
38	T19	7	box hybrid
39	C20	7	E. albens
40	T20	9	E. polyanthemos
41	C21	13	E. sideroxylon
42	T21	14	E. sideroxylon
43	C22	5.5	E. sideroxylon
44	T22	6	E. sideroxylon
45	C23	6.5	E. polyanthemos
46	T23	8	E. polyanthemos
47	C24	5	E. polyanthemos
48	T24	6	E. polyanthemos
49	C25	5	E. polyanthemos
50	T25	7	E. polyanthemos

Data analysis: Daily nest survival

Daily nest survival (DNS) of artificial nests was modelled using the nest survival option in the program MARK (Cooch and White 2005). In each experiment nests were assigned to groups (treatment or control) with height of nest as a covariate following methods described in Dinsmore and Dinsmore (2007). Other covariates such as tree species, percentage foliage and flowering were not modelled as these were not paired between the treatments. Four candidate models were compared in each experiment and model averaging was used in both cases to determine DNS for control and treatment nests given more than one model was <2 AICc. DNS was converted to a probability that a nest would survive the 14 day incubation period of Regent Honeyeaters.

Results

Experts suggested a range of alternative management actions to overcome poor Regent Honeyeater nest success, reflecting the threats that management was struggling with, including the attraction of predators to nests and/or nest abandonment caused by interventions. Initial consideration of removing predators was dismissed given they are native species and some are also threatened (e.g. Squirrel Gliders). The remaining actions included the use of tree collars, nest cages and supplementary feeding.

The results of the Monte Carlo simulation of the decision tree suggested that using branch collars would provide the best option under current uncertainty (providing a mean nest success rate of 22%; range 9.2-35.7% compared to a mean success rate of 15.6%; range 6.0-28.9% under no management), although the distribution of outcomes shows substantial overlap between alternative actions (Figure 14A). Sensitivity analysis identified that a combination of optimism in the effectiveness of collars to deter predators and a fear that nest cages would cause nest desertion was driving this result. Uncertainty in predation rate had the biggest influence on the expected outcome: nest success ranged from 10.4% to 20.6% across the range of inputs for predation whereas all other inputs led to less marked variations in outcome (13.9%-17.2%; Figure 14). In scenario two, where management could reduce predation without increasing desertion rates, then predator exclusion using a combination of branch collars and nest cage is the favoured management choice, potentially providing a substantial improvement in nest success (mean nest success 37.1%; range 23.4-53.1%) (Figure 14B). In scenario three, where using collars or cages would not reduce predation, the predicted nest success changed markedly and supplementary feeding became the most effective management action to take, although its expected benefits were small (mean nest success 18.1%; range 7.8-30.1%)(Figure 14C). The uncertainty surrounding the effectiveness of managing predation, and the associated risk of nest desertion, means that the best management action is sensitive to uncertainty and there is value in learning more about it.

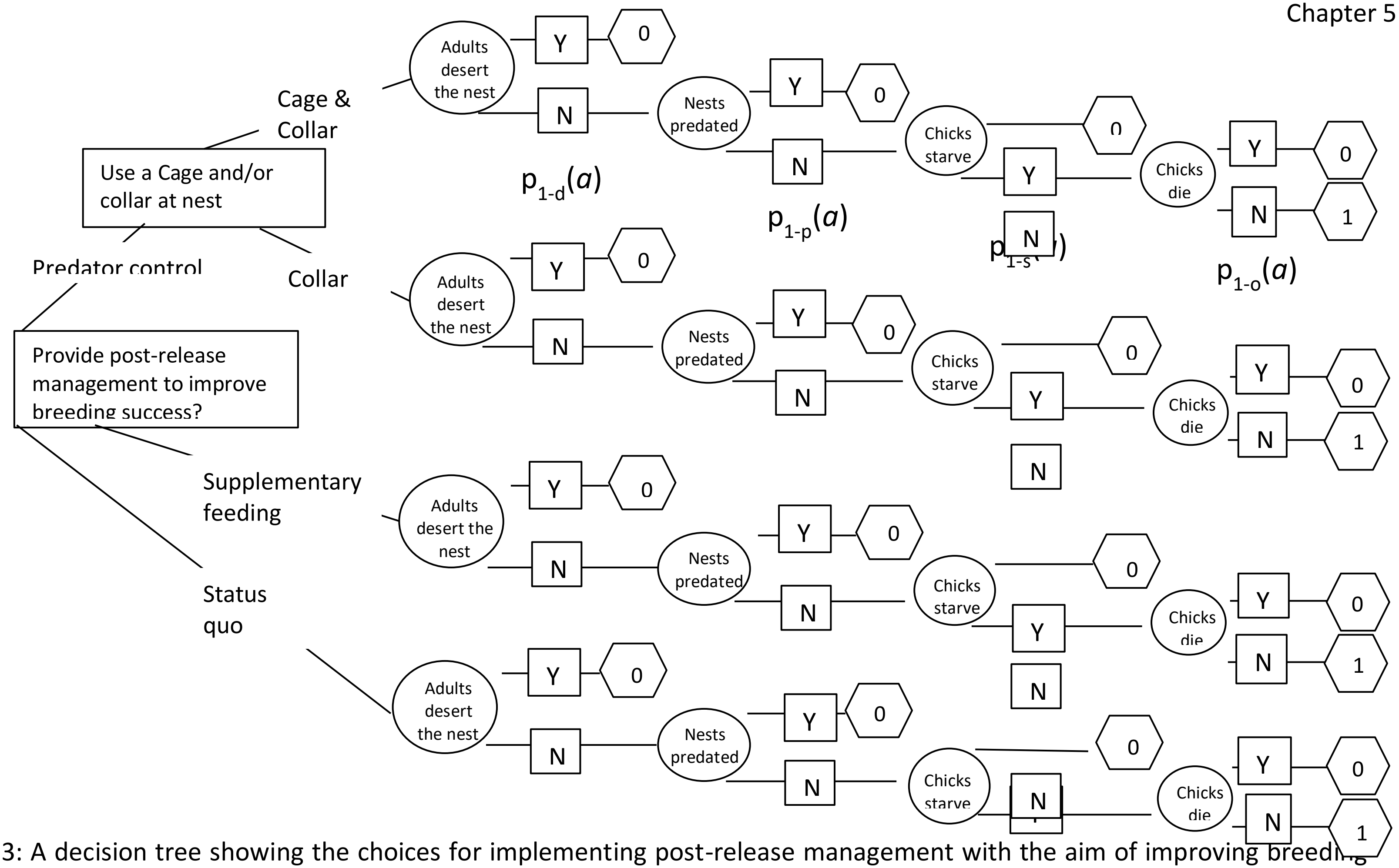

Figure 13: A decision tree showing the choices for implementing post-release management with the aim of improving breeding success. Decision nodes are represented as rectangles. Outcomes are represented by circles and show possible outcomes. Outcomes are represented by hexagons and discrete binary values of 0 for nest failure and 1 for successful fledging. Values from these distributions were used to solve the discrete outcome for each branch of the tree (nest success), given by; $\mathbf{NS_{a} = p_{1\text{-}d}(a) \cdot p_{1\text{-}p}(a) \cdot p_{1\text{-}s}(a) \cdot p_{1\text{-}0}(a)}$. $\mathbf{NS_{a}}$ is nest success of action *a* and is the product of four independent probabilities, $\mathbf{p_{1\text{-}d}(a)}$ is the probability that the nest will not be deserted under action a, $\mathbf{p_{1\text{-}p}(a)}$ is the probability that the nest will not be predated under action *a*, $\mathbf{p_{1\text{-}s}(a)}$ is the probability that the nestlings will not starve under action *a*, $\mathbf{p_{1\text{-}o}(a)}$ is the probability that the nestlings will not die for other reasons under action *a*

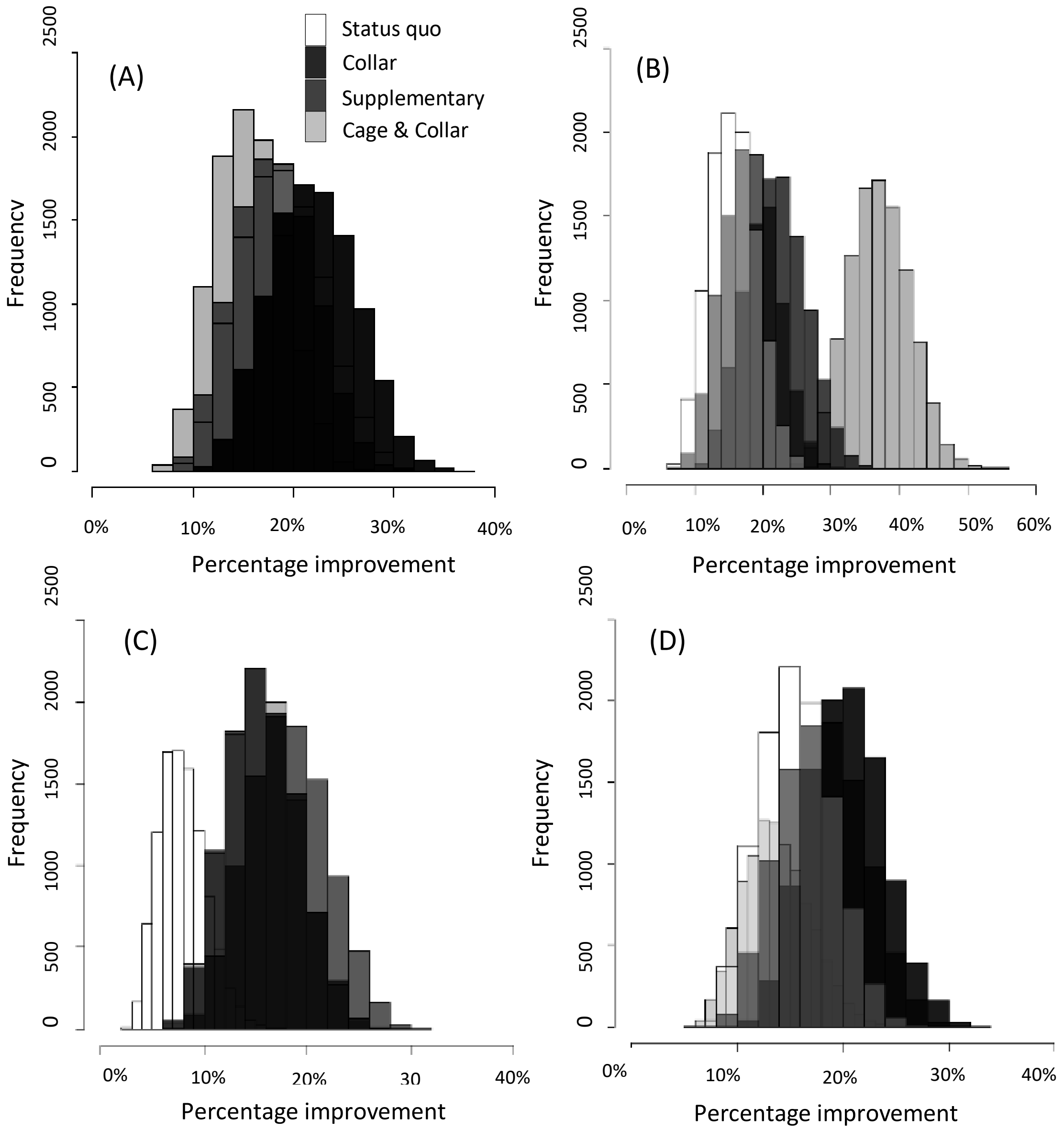

Figure 14: Frequency distribution of probability of nest success from 10,000 iterations of the decision tree using Monte Carlo simulation function built in DecisionTools Suite 7 under four different scenarios; (A) under current uncertainty based on expert opinion, (B) a 'no *additional* desertion risk scenario' the possibility that cages did not affect nest desertion rates was simulated, (C) a 'predation risk not reduced' where a scenario in which neither cages nor collars would reduce the chance of predation was simulated, and (D) an updated decision tree following validation of expert opinion by field experiments.

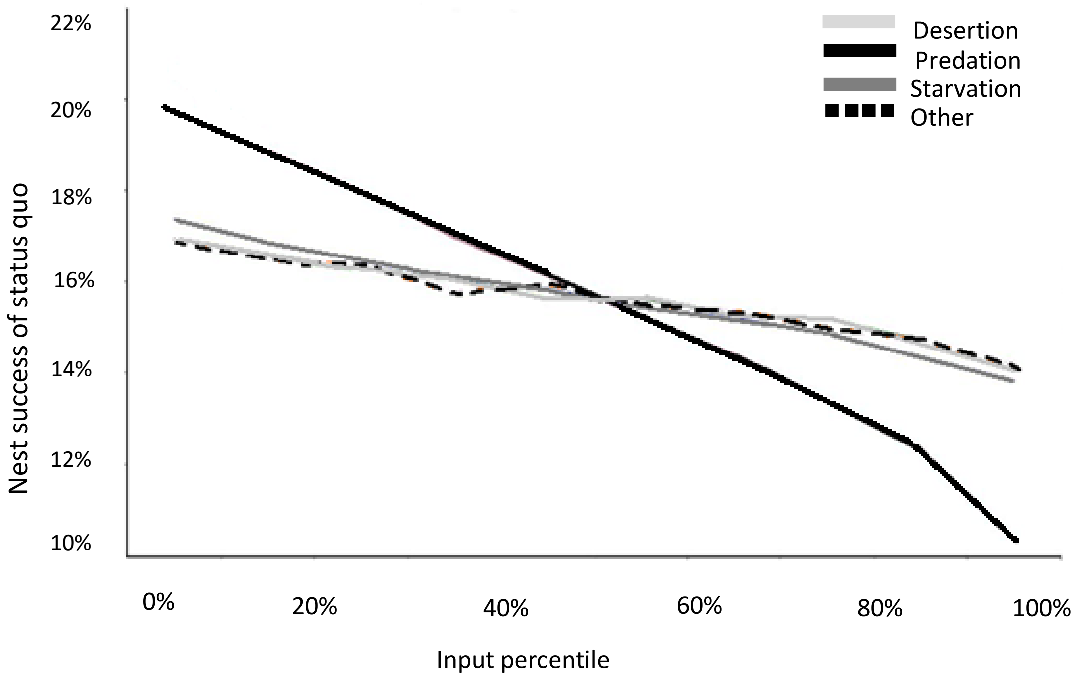

Figure 15: Spider graph showing the relative influence of uncertainty in each chance node of the decision tree for the *status quo* scenario (where there is no nest management) on probability of nest success. The full range of uncertainty in each chance node is divided into input percentiles. The steeper the slope the greater the contribution of uncertainty in input values on uncertainty of nest success. The four chance nodes of the decision tree are probability of desertion (light grey line), probability of nest predation (solid black line), probability of starvation (dark grey line) and probability of other causes of nest failure (dashed line).

Half of all nests (10 of 20) protected with cages survived (compared to 6 of 20 control nests) and two of 25 nests survived that were protected with collars (compared to 0 of 25 control nests). Camera trap images confirmed a range of nest predators whilst characteristic indentations in plasticine eggs allowed less refined classification of nest predators. Predators of caged nests were categorised as mammal (17%), mammal or large bird (54%) and unidentified (29%). A similar range of predators were captured on cameras of collared nests; bird (65%), mammal (26%), and unidentified (9%). One camera failed to work. Images show that marsupials such as Squirrel Gliders could navigate around the collars and that a Grey Shrike-Thrush *(Colluricincla harmonica)* was able to enter a cage (Figure 16).

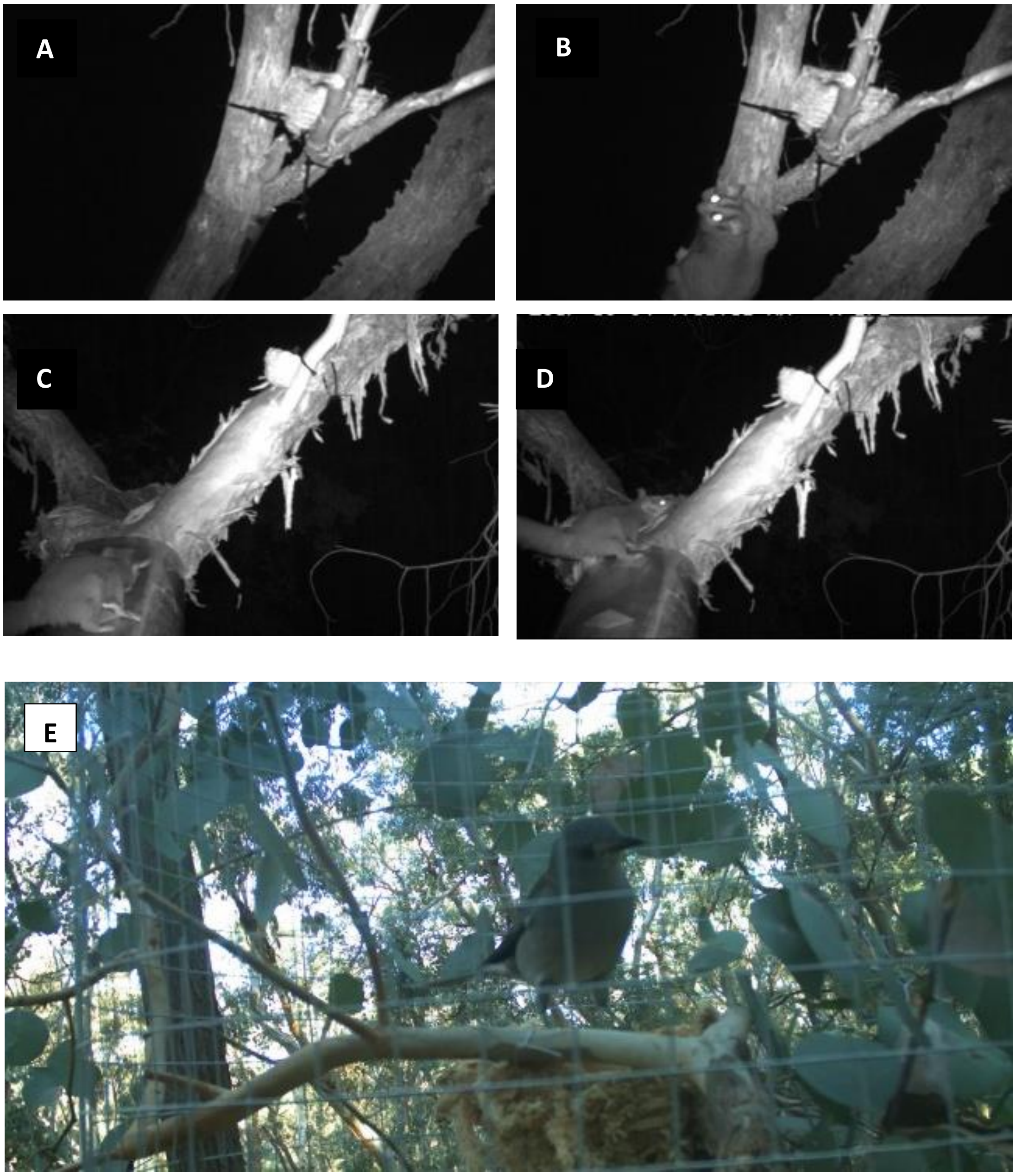

Figure 16: Photos of marsupials managing to climb over the polycarb collars. (A) shows one treatment tree with the nest, a Feather Tail Glider (*Acrobates pygmaeus*) with the collar just below the animal. (B) shows the same nest with a Squirrel Glider (*P. norfolcensis)* holding onto the collar. (C) shows a Squirrel Glider balancing on a collar and (D) making it to the nest branch. (E) A Grey Shrike Thrush inside a cage.

In the cage experiment, model comparison showed that there was little influence of either treatment or height, with the null model only slightly outside a delta AIC threshold of 2 (Table 9). Model averaged predictions for daily nest survival for treatment nests (those with a cage) were 0.94, 95% CI: 0.89-0.97, giving a probability of surviving the entire incubation period of 42% (20%-65%), whereas DNS for control nests (those without a cage) were 0.91, 95% CI: 0.85 – 0.95, giving a probability of surviving the entire incubation period of 27% (10%-49%) (Table 10). Model comparison in the nest collar experiment also showed very little influence of either treatment or height on DNS with the null model <2 AICc of the top ranked model (Table 9). In this case model averaged predictions for DNS of treatment nests (those with a collar) were 0.61, 95% CI: 0.5 – 0.71, giving a probability of surviving the entire incubation period of 0.001% (0% - 0.008%), which was very similar to that of control nests (those without a collar) where DNS was 0.58, 95% CI: 0.46- 0.69 giving a probability of surviving the entire incubation periods of 0.0005% (0% - 0.006%) (Table 10).

Table 9: Results of model comparison for daily nest survival for cage and collar treatments using artificial nests in Chiltern-Mt Pilot National Park, Victoria, Australia. Daily nest survival was modelled using the nest survival function in Program MARK (Cooch and White 1999) as constant (.) or as a function of treatment ("Treatment"; management using cage or collar versus not using cage or collar) and/or nest height ("Height"). Models were ranked by Akaike Information Criterion (AIC), the model with the lowest AIC receiving the greatest support, and models within two AIC points considered to receive similar support. K indicates the number of parameters, weight indicates the normalised AIC weight, and deviance indicates model deviance.

Nest Survival

Model	**K**	**AICc**	**ΔQAICc**	**Weight**	**Deviance**
Cage (n=40)					
DNS (Treatment+Height)	3	172.44	0	0.38	166.37
DNS (Height)	2	172.84	0.4	0.31	168.81
DNS (Treatment)	2	173.94	1.5	0.18	169.9
DNS (.)	1	174.48	2.04	0.14	172.47
Collar (n=50)					
DNS (Height)	2	156.97	0	0.4	152.87
DNS (.)	1	157.84	0.87	0.26	155.81
DNS (Treatment+Height)	3	157.96	0.99	0.24	151.74
DNS (Treatment)	2	159.71	2.73	0.1	155.6

Table 10: Model averaging results for cage and collar experiments treatments using artificial nests in Chiltern-Mt Pilot National Park, Victoria, Australia giving weighted averages of daily nest survival and 95% confidence intervals for managed nests (those with a cage or collar) and control nests (those without cage or collar) for each experiment.

	Daily Nest Survival		
	Weighted average	Lower 95% CI	Upper 95% CI
	Cage experiment		
Cage	0.94	0.89	0.97
No Cage	0.91	0.85	0.95
	Collar experiment		
Collar	0.61	0.5	0.71
No Collar	0.58	0.46	0.69

Updated decision tree

The outcomes of the artifical nest experiments were provided to all seven experts and they were provided with an opportunity to update their previous opinions on the probability of predation for actual Regent Honeyeater nests. This was required given the field experiments only considered incubation whereas management was proposed to improve nest success to the point of fledging. Learning allowed experts to better understand how their proposed actions would reduce predation of nests. The largest change in expert opinion occurred for use of a nest cage with a substantial increase in the risk of predation on pre-experiment opinions, whereas there was only a slight increase assigned to the perceived risk of predation when using only a collar (Table 6). Interestingly, experts still believed there was benefit in both collars and cages over status quo (i.e. believing that predation would be reduced) (Table 6). A final simulation of the updated decision tree again showed that tree collars are the favoured management action, albeit with a slightly reduced overall improvement in nest success compared to earlier opinions (20.3% nest success on average; range 10.3-33.5%), and nest cages became by far the worst (13.7% nest success on average; range 5.4-26.1%) (Figure 14D).

Discussion

My study shows the reality of decision making in response to a conservation threat for a critically endangered species. I show how managers have to deal with limited information to predict management outcomes, balancing the optimism from benefits of management with the risks that management makes matters worse. Poor nest success in Regent Honeyeaters led the recovery team to consider what to do. I worked with Regent Honeyeater experts to approach this problem in a structured way. Experts suggested a number of alternative management actions aimed at reducing nest predation and nestling starvation, but these interventions also risked increased nest desertion. Reducing uncertainty in the opinions of experts about the risks of predation and desertion was likely to clarify the best management response. The recovery team, however, was not willing to learn about desertion risk until it was sure that management would reduce predation. Interestingly, despite field experiments demonstrating that proposed management had limited ability to reduce predation, expert opinion did not shift substantially and a belief remained that these actions could still be effective. Therefore under reduced uncertainty the use of tree collars remained the most effective management response to poor nest success, but with potential for much greater improvement in nest success if cages could be used without causing nest desertion.

When faced with a complex problem in scenarios with limited knowledge, conservation managers often fall back to requesting more research in the hope this will clarify a best course of action (Canessa *et al.,* 2015), or follow the advice of an experts' intuition (Moseby *et al.,* 2014). The former may be valid but should be embedded within a decision making framework to ensure research provides useful information (Taylor *et al.,* 2017; Bower *et al.,* 2017). The latter often reflects an alternatives-focussed approach, which often fails to achieve management objectives, may neglect important considerations, and otherwise lacks transparency that facilitates group agreement and improvement (Reed 2008). I found that the Regent Honeyeater experts often had strong and diverse opinions on key uncertainties. The implementation of a structured approach to explore this management problem allowed these opinions to be expressed and combined in a highly transparent way to effectively capture the uncertainty in these opinions (both in what to do about poor nest success and how effective it would be). This contrasts to threatened species management solutions frequently applied by experts without critical evaluation of their utility, risking ineffective management (Armstrong and Seddon 2008; Taylor *et al.,* 2017) and reinforcement of conservation dogmas (Martínez-Abraín and Oro 2013).

The problem of Regent Honeyeater nest management was captured as a decision tree. Calculating the predicted outcomes of alternative management actions in this way allowed the identification of a best management alternative under current uncertainty (the use of tree collars to protect Regent Honeyeater nests). Decision trees remain underused but provide useful and clear summaries of conservation problems (e.g. Canessa *et al.,* 2016a). The decision tree was also used to show how uncertainty influenced which management action would be best by running a set of simple sensitivity tests (reducing uncertainty in the effectiveness of predator management and in desertion risk could change the preferred management strategy). The sensitivity analysis, therefore, captured both the experts' optimism that management solutions could effectively reduce predation but also the risk that this might cause substantial nest desertion. Linking how uncertainty hinders the selection of a best management alternative can help direct learning (strategic monitoring; Ewen & Armstrong 2007). In the current study it led to the use of artificial nests aimed at reducing uncertainty around how management would reduce predation. Initial exploration of management alternatives and uncertainty in this way helps select a best management response and provides predictions which can be tested, new information generated and existing knowledge updated.

The experimental studies on the effectiveness of collars or cages in reducing predation was necessarily done using artificial nests that focussed only on the egg stage. The results indicated that the initial opinions of the effectiveness of management might have been overly optimistic. Artificial nests were the preferred method for learning given the recovery team was averse to the risks of testing on actual Regent Honeyeater nesting attempts. Using artificial nests to measure nest predation rates is common (Anderson *et al.,* 2016), especially in habitats where real nests are hard to find (Chmel *et al.,* 2018) or likely to be in low numbers. However, artificial nest experiments are often found to report predation rates that differ from real nests (Zanette 2002; Batáry and Báldi 2005). Given predation rates may be different for artificial nests, and that this study only included the egg stage, it was important to provide the experiment results back to experts to update their opinions of nest predation. Results from our study show a large difference in the survival rates of nests at egg stage between the experiments, cages and collars. Those nests which were treated with collars saw a much reduced survival rate than those with cages. There could be a number of reasons for this; firstly, the collars may offer no protection from avian predators and without parents to protect the nest, they are fully exposed during the day to be predated by other birds. Secondly, the collars may attract inquisitive birds initially, who then find the nests and opportunistically predate the eggs. The plastic of the collars may reflect light and provide a shiny attractant for many species. Lastly, for logistical reasons, the experiments were conducted two months apart and although it was not predicted that this could be a confounding variable, time of year and the types of predators present may have varied. This would need to be controlled for if future trials were to be conducted.

Interestingly, the experts' updated opinions differed only marginally from their original values. Experts still believed that using cages and/or collars would reduce predation, substantially so in the case of cages. Continuing to believe that predation would be reduced, albeit to a lesser extent than prior to the experiment, suggested that experts either consider artificial nests poor proxies for understanding predation rates or otherwise remain strongly anchored on their prior beliefs. The behaviour of experts when confronted with new information may reflect a confirmation bias (e.g. here expressed as discounting evidence that does not support their prior belief). Alternatively, the perceived value of information (Canessa *et al.* 2015) when developing the experiment was not realised when implementing it. Importantly, however, experts remain confident that cages could substantially reduce predation and the reason why this management is not favoured hinges on the risk of desertion. A recommended next step in Regent Honeyeater management is therefore to trial cages on real Regent Honeyeater nests, perhaps initially using captive birds.

Treating expert knowledge thoroughly by both asking the available pool of experts and clearly articulating uncertainty quantitatively is regarded as best practice (Runge *et al.,* 2011; McBride *et al.,* 2012). However, it remains rare in management of threatened species where the primary decision maker is often a single manager who relies on experience (Pullin and Knight 2003; Pullin *et al.,* 2004). Reliance on intuitive responses to personal experience by experts generates two linked problems. First, management recommendations become very difficult for others to engage with and learn from. Second, intuitive management can lead to more dogmatic attitudes in management which are rarely critically reviewed, with no predictions against which to compare even if they were (Possingham *et al.,* 2002; Martínez-Abraín and Oro 2013). A better approach is to estimate the consequences of a range of management alternatives (either via monitoring, expert opinion, or both) on a set of objectives and select the best one (Taylor *et al*., 2017; Chapter 2). My study showcases how expert opinion can be applied to the management of a critically endangered woodland bird species in Australia, as part of a structured process aimed to improve breeding outcomes.

Chapter 6~ Thesis discussion

Advances in knowledge

My PhD research aimed to improve conservation efforts for threatened species by showcasing the application of decision tools to the management of a Critically Endangered bird in Australia. In this discussion chapter I outline the key findings of my research and critically evaluate my work, highlighting areas which require further attention and directions for future research for both the management of the Regent Honeyeater and for the wider reintroduction biology community.

Reintroduction Biology as an effective applied science

In order for applied science to directly benefit conservation managers, the science needs to assess the efficacy of alternative management actions at achieving management objectives. Over the last two decades there have been repeated calls to improve conservation outcomes through evidence based management (Sutherland *et al.,* 2004; Pullin and Knight 2009; Sutherland and Wordley 2017). In chapter 2, I evaluated 20 years of scientific literature, with a focus on reintroduction biology, to quantify how well applied science in this field was supporting decision makers. Although I found an increase in reintroduction science literature over the two decades and a marginal increase in those that stated *a priori* hypotheses, overall reintroduction biology has not been providing the evidence based support that is most helpful to conservation managers, by comparing two or more management actions against recovery objectives. These data are important, not only to those involved in reintroduction biology, but to anyone working in conservation science supporting species recovery. Included in this chapter I give examples of how applied science can be most useful to managers and explain why simply having a reflective ‘management implications’ section without explicitly testing for the optimal management action will, at best, be an inefficient approach to improving conservation outcomes. This chapter was published in the journal ‘*Trends in Ecology and Evolution’* and I hope that future studies aimed at learning more about species reintroduction will take note of the recommendations and plan their research in a way which is more useful for decision makers; that is, truly evidence-based conservation.

With this primary chapter I documented the current standing of the field of reintroduction biology and highlight the need to better integrate decision making into evidence based conservation. Following from this, I then show how this approach can be achieved using the case study of captive bred and release Regent Honeyeaters in north east Victoria, Australia.

Short Term survival

To showcase the methods I promote I first analysed survival data from four different release events of Regent Honeyeaters to answer *a priori* hypotheses about the factors affecting short term post-release survival in the wild. Previous studies on a range of taxa have found that the age, sex, weight and fitting transmitters on animals can all significantly influence how well they survive post-release (Masuda and Jamieson, 2012; Sarrazin *et al.,* 2000).

The recovery team specifically wanted to know if these variables also influence the survival of the captive bred and released Regent Honeyeaters. For example, if older birds had higher post release survival then managers could hold birds longer in captivity before release. The decision is difficult because holding birds for release for longer periods would require substantial additional investment in captive facilities as there is currently no capacity to do this at Taronga Zoo. I found that age, sex and weight had no significant effect on the survival of the birds once they were released. In previous years a mix of ages and an even as possible a ratio of males to females have been released. Younger birds and females tend to be lighter in weight than older birds and males. My findings are important for the recovery team as it answers the question '*should we release younger or older birds to improve survival?'* They can now be confident that the captive breeding component of the reintroduction has not negatively affected the birds' survival and major changes are not required. An important outcome from this analysis was that the year of release had the greatest impact on survival, and even more so if the birds are wearing a transmitter. Our results here were not exactly as predicted; I hypothesised that birds wearing a transmitter would have lower survival than those without, however this was only the case for the 2017 release. Unfortunately there are no quantitative data on environmental conditions for each year of release. From the first cohort of 50 birds to be released in 2017, 12 were found dead within the first 2 days. These birds were all wearing transmitters and all banded only birds were observed alive. The retrieved dead birds had lost substantial weight and the management team hypothesised that there was not enough food available and that the added weight of the transmitters was causing the mortalities. What this does show is that being released into a new environment is challenging and that the conditions at the time of release will play a critical role in immediate post release survival.

Improving breeding outcomes

Released birds need to not only survive in the wild, but to also successfully breed in order to reverse the decline of the population. In chapter 4 I undertook the first intensive study of captive bred Regent Honeyeater breeding success. This was a key uncertainty of the recovery team, and so monitoring was planned specifically to capture discrete events at the nests, using modified surveillance cameras. I found that the majority of nests were predated by native species, both mammals and birds and that a significant proportion of nests failed due to chick mortality. This latter cause of nest failure remains a knowledge gap and further experimental monitoring is required to understand if lack of food, poor parental provisioning, weather and other variables caused the chicks to die in the nest. It is important to note that poor breeding success is also observed in wild Regent Honeyeaters which have a nest survival rate of 0.32 (Crates *et al.,* 2018b). This is slightly higher than the nest survival rate of captive bred and released Regent Honeyeaters reported in chapter 4 (0.21), yet it confirms poor reproductive success it is not unique to them.

These worrying results caused the recovery team to urgently consider management actions aimed at improving breeding success after future releases. Therefore in chapter 5, I showcased how to formalise the decision process to take full account of the substantial uncertainties using a mix of expert elicitation and developing a bespoke decision tree, and conducting targeting learning via artificial nest experiments. Firstly I captured the opinions of experts working with the species and their concerns regarding management alternatives aimed at improving breeding outcomes. I was then able to represent these opinions using a decision tree which enabled me to quantify the uncertainty involved and also target learning to reduce this uncertainty using experimental trials of nest protection methods. The use of decision trees in conservation science is still uncommon, yet they are a tool which effectively articulates uncertainty, providing a more transparent assessment of management alternatives. This specific study showed that expert opinion can be over optimistic about the efficacy of certain management actions, in this case tree collars. It also highlighted that further learning is required on potential desertion of nests by adults with the use of nest cages, as this alternative could be the most effective in improving nest success of Regent Honeyeater. I hope that this chapter encourages further use of the suite of decision support tools available in the future and flags that despite the obvious value in expert knowledge, in rapidly changing systems adaptive management and targeted learning are essential when making risky decisions.

Implications for Regent Honeyeater Management

My results have helped answer some of the key questions of the recovery team regarding Regent Honeyeater survival and can provide some guidance to the recovery team for addressing future management decisions. Results from chapter 3 show that released birds have high short term survival and can transition from a captive setting to the wild well. I document that in 2017 there were higher than expected mortalities in the first 2 days when 13 birds wearing transmitters were found dead. It was hypothesised that environmental conditions were suboptimal and the added weight of the transmitter put too much stress on the birds. This appears to correlate with the survival analysis in which the top model suggested that the year of release and interaction with wearing a transmitter had the biggest effect on survival, with 2017 being the only year that birds wearing transmitters had lower survival than those that were not. This hypothesis needs to be explicitly tested to gain clarity on which environmental variables contribute to immediate survival post release. When deciding on the future release sites, especially as these are proposed to be in New South Wales from now on, it will be important to talk to the local community including apiarists, farmers, Aboriginal land owners, wildlife groups and other community organisations that are involved with the local environment to gain a better understanding of the area, then develop hypotheses which can be explicitly tested.

Monitoring of breeding attempts in Regent Honeyeaters show that they are able to form pair bonds, choose nests sites, build nests and produce eggs, however, predation and chick mortality are limiting factors for reproductive success. New experiments need to be planned to test the effectiveness of different nest protection designs which do not risk desertion and also control of predators for which collars will not deter such as other birds. This can be localised *in situ* in the case of predator management and in captivity at Taronga zoo with regards to cage design and familiarising the birds with a cage to reduce the potential of desertion.

The expert elicitation process highlighted the difficulties often faced when management are reluctant to engage and welcome novel ways of making decisions. Experiential knowledge is invaluable and can not only complement scientific research, but also guide it (Fazey *et al.,* 2006). There is increasing attention in the literature for applied science to be integrated into and to help inform management decisions (Cook *et al.,* 2010; Milner-Gulland and Shea 2017; Cook and Sgrò 2018). However, this will only be possible if those making decisions are open to the different methods to achieving this. If not, management is at risk of being based on dogmatic approaches to decision making and learning and effective outcomes will be hindered and at worst completely fail.

One area of management which could greatly be improved for this project is the inclusion and collaboration with local Aboriginal communities. Australian Aboriginal culture is thought to be the oldest continuous culture (Nagle *et al.,* 2017) and inherently contains detailed knowledge of the land it has evolved on. The inclusion of Aboriginal knowledge into environmental management has been advocated for decades in Australia (Ens *et al.,* 2014), and although progress is being made, Aboriginal Australians are still struggling to gain sovereignty over their land rights. The Australian government have a number of policies mandating working and collaborating with local Aboriginal Communities. In Victoria, The Department of Land, Water and Planning (DEWLP), the primary partner in the Regent Honeyeater reintroduction project have management plans which state their commitment to working with local communities: 'Munganin – Gadhaba – Achieve Together: DELWP Aboriginal Inclusion Plan 2016-2010' and 'Protecting Victoria's Environment – Biodiversity 2037' (State of Victoria 2015; DELWP 2017).

Although there is currently dispute over who has native title rights in Chiltern and the surrounding area, there are the Aboriginal communities of the Yorta Yorta and the Dhudhuroa peoples in Victoria, (Blake and Reid 2002; Ens *et al.,* 2014) and the Wiradjuri of New South Wales, which is only 40km away. The Murray River, which separates the two states and is the longest river in Australia, was and still is highly significant for Aboriginal people in the area (Humphries 2007). In the Hume-riverine region of North East Victoria and New South Wales, where Chiltern Mt-Pilot National

Park is situated, the Murray River was an important meeting place for the different tribes and clans in the area and for travelling and hunting groups who would be passing through. It is beyond the scope of this thesis to give a full account of the Aboriginal history here and the importance of the concept of Caring for Country, but it is important to understand that there is still significant connection to country in the area including and surrounding Chiltern - Mt Pilot National Park. Caring for Country is the term used to describe the relationship and holistic custodial management approach to the land, water and wildlife of the Aboriginal People of Australia. Aboriginal culture all over Australia is founded on stories, songs and art which create a spiritual and physical bond to their ancestors and the creation of their land. Aboriginal people in Australia are the most socio-economically disadvantaged group in the country and research has long shown that the loss of this connection has detrimental impacts of the health and mental wellbeing of Aboriginal peoples (Adams 2008; 'Yotti' Kingsley *et al.,* 2009). Aboriginal communities living on their traditional lands have lower levels of disease and death rates than those living in forced centralised communities (Mcdermott *et al.,* 1998). *''I need to convince people when we talk of our Country as being central to our existence we are not engaging in a flowery word; we really are talking of life and death''* (Flick, 1998 in 'Yotti' Kingsley *et al.,* 2009). With this deep connection to Country, it is irrefutable that their knowledge about land management is invaluable and essential.

The Regent Honeyeater project relies on volunteers from all walks of life, from local residents, university students, novice and experienced birders, and anyone who wants to get out into the forest and help collect the data. The recovery team provide training on radio tracking and in doing so they learn about other bird species and all the native vegetation. This is an opportunity to collaborate with the local Aboriginal community as young adults could gain valuable field skills and local elders could come out and talk to volunteers and teach about the forest. The Burraja Indigenous Cultural and Environmental Discovery centre in Wodonga provides classes for local children who identify as Aboriginal. These classes include day trips into Chiltern Mt-Pilot National Park to visit the traditional rock art sites and scarred trees; however, there is no collaboration between this organisation and the Regent Honeyeater project, despite the fact the offices are 6km away from each other. From personal communication with the Aboriginal Elders in this community, they would love the chance to engage with the project and provide an opportunity for the children and young adults to learn from the project. The next release in 2019 is planned for New South Wales, and the Aboriginal peoples of the land which is the remaining stronghold for the wild Regent Honeyeaters in the Capertee Valley are also the Wiradjuri.

There are examples in the literature of indigenous values being incorporated into conservation and natural resource management decisions (Failing *et al.,* 2007; Harmsworth *et al.,* 2016). This can be easily facilitated throughout the process using structured decision making and can be particularly valuable when developing alternatives.

Implications for reintroduction biology and threatened species management

My results are also relevant to the wider reintroduction community and the methods used in this thesis can help guide the approaches taken in other species reintroduction projects. I start this thesis with a review that indicates that reintroduction biology can still improve its application to practical conservation. I show that an increase in the literature does not directly relate to its usefulness to conservation managers, and applied scientists working within this field have the opportunity to expand their research and focus on metapopulation and ecosystem level studies. The running theme and message I have been advocating through this thesis is that alternative management actions should be compared against recovery objectives in order to select the optimal action and I proceed to demonstrate how this can be achieved.

The application of decision tools can be useful in all areas of threatened species management. Although my research did not explicitly deal with the importance of stakeholder diversity in the decision process, the inclusion of multiple stakeholders and other interested parties such as local apiarists and nature groups in this instance could have provided extra knowledge on environmental conditions that were not captured by anyone else. If there is a knowledge gap that has been identified when making decisions about any step of the reintroduction process, simply collecting more data might not be the most effective strategy. Expanding the collective knowledge of those involved in the project may help develop hypotheses and alternatives to be tested that would otherwise be overlooked and therefore I recommend other projects be more open to public participation and traditional knowledge through workshops to help inform decision making.

My thesis has identified and filled knowledge gaps for both Regent Honeyeater conservation and reintroduction biology. I have demonstrated how decision analytic tools can facilitate what the literature has been recommending for a while: the need for *a priori* hypothesis testing and a structured approach to monitoring to inform management.

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