Planning Principles and Tools for the Conservation of Threatened Species

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ABSTRACT

Biodiversity is being lost at unprecedented rates. Area and threat-based conservation measures act too slowly or inadequately for species already threatened with extinction and more urgent, intensive action is often required. Science-based conservation planning for these species emerges as a critical tool to 'bend the curve' of biodiversity loss. Thousands of species need this attention, but few receive it. A massive global upscaling of planning effort is imperative. Governments and organisations need clarity on how to plan effectively, while funders need evidence that good planning leads to positive outcomes for species. First, I reviewed the case for species conservation planning to clarify why it is important, where it sits in the broader landscape of biodiversity planning, and how many species are likely to need it. Second, I comprehensively reviewed planning advice from different countries, organisations and academic studies. This yielded consensus on crucial elements of a species conservation plan, a common acknowledgement of under-resourcing, a shared impetus for more efficient planning methods and a trend towards greater stakeholder involvement. Third, I investigated whether species conservation prospects are measurably better with planning than without it. For 35 planning projects completed in 23 countries over 13 years, IUCN Red List assessments showed aggregate decline to extinction slowed after planning and was reversed within 15 years. Conversely, counterfactual simulations projected around eight extinctions without planning, evidencing planning's positive impact on species status. Fourth, I explored the utility of Population Viability Analyses (PVA), through in-depth analysis of New Zealand's critically endangered kākāpō. These analyses suggested that more regular supplies of preferred foods and larger carrying capacities may lead to management independence. PVA can therefore play a pivotal role in planning for species with small or highly fragmented populations. Fifth, I demonstrated integration of PVA, Disease Risk Analysis and *in situ* and *ex situ* consideration into stakeholder-inclusive planning for Tasmanian devils, showing the tangible, immediate conservation progress that resulted.

My research offers a roadmap for effective conservation planning, underscores the value of integrating PVA into planning for specific species and provides compelling evidence that good planning benefits species conservation outcomes.

To my Mom and Dad

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CHAPTER 1. THE CASE FOR SPECIES CONSERVATION PLANNING

“The first rule of intelligent tinkering, is to save all the pieces.” Aldo Leopold (Leopold 1953).

1.1 THESIS ROADMAP

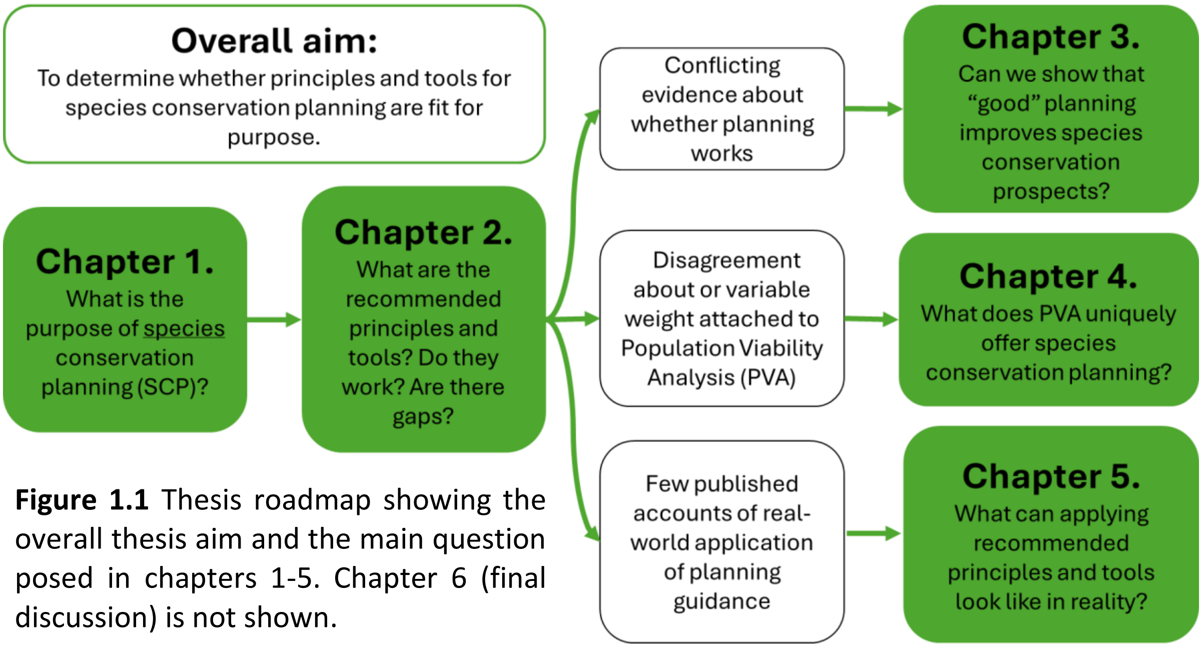

Figure 1.1 Thesis roadmap showing the overall thesis aim and the main question posed in chapters 1-5. Chapter 6 (final discussion) is not shown.

This thesis includes six chapters, five of which are described in the form of a roadmap in **Figure 1.1**. An introductory chapter (Chapter 1. The Case for Species Conservation Planning) is a literature review that sets the topic of species conservation planning in the wider context of biodiversity conservation planning. A second literature review (Chapter 2. Species Conservation Planning Approaches and Lessons to Date) identifies three notable gaps or ambiguities in the literature which are then pursued in the following chapters as: a published article in the journal *Biological Conservation* (Chapter 3. Science-based, Stakeholder-inclusive and Participatory Conservation Planning Helps Reverse the Decline of Threatened Species); a chapter showing the unique value of population viability analysis to species conservation planning (Chapter 4. Population Viability Analysis Provides Insights into the Potential for Conservation Independence in the New Zealand Kākāpō); a published book chapter (Chapter 5. Population and Habitat Viability Assessment: a One Plan Approach to Saving the Devil) which describes in detail a practical application of the tools and approaches analysed in the previous chapters. A sixth chapter provides discussion and recommendations for future work

(Chapter 6. General Discussion). The peer-reviewed manuscript includes a stand-alone abstract and the same format has been followed with the other chapters for consistency. The two original publications have been reformatted to meet university guidelines with the previously attached reference lists integrated into a single master reference section. However, the personal pronoun "we" has been retained as per the original publications rather than standardising the whole thesis to "I". All supplementary material for the published chapter and for the unpublished PVA chapter can be found in the appendices. Throughout the thesis, I employ the pronoun 'we' instead of 'I' whenever the research is based on collaborative efforts. Finally, we make use of the names New Zealand and Aotearoa interchangeably.

1.2 Abstract

Biodiversity is in precipitous decline with current rates of extinction set to increase. Beyond ethical considerations this presents severe challenges to human health and well-being. Nations are responding with their own internal measures and in response to external encouragement and support from international agreements. Of particular relevance is the Convention on Biological Diversity (CBD) Post-2020 Framework which carries renewed calls for drastic action to "bend the curve" of biodiversity decline by 2030. Within this framework, there are calls for protecting more terrestrial and marine areas for species, for further investment in mitigating key threats such as over-exploitation, invasive species and pollutants and for urgent action to conserve and recover species already threatened with extinction. There are many different approaches to planning biodiversity conservation. Some focus on the designation of areas where biodiversity will be protected, others on how best to manage it at the habitat or ecosystem scale, and others focus on mitigating specific, pervasive threats. Applied at scale these can all play a significant role in slowing or preventing the decline of common species. For threatened species, many of which may not respond quickly enough to these approaches, species-focused conservation planning can be an effective way to identify and drive well-targeted action. Estimates suggest many thousands of species require this level of attention, but few receive it.

1.3 THE GLOBAL STATE OF BIODIVERSITY AND THE IMPLICATIONS FOR PEOPLE

"Biodiversity conservation is more than an ethical commitment for humanity: it is a non-negotiable and strategic investment to preserve our health, wealth and security." (WWF 2020).

According to the Living Planet Report, until 1970, humanity's ecological footprint was smaller than the Earth's rate of regeneration. Since then, the human population has doubled and to feed and fuel our lifestyles, we are now overusing the Earth's biocapacity by at least 56% (WWF 2020). The result is unprecedented declines in global biodiversity and a shifting climate (Newbold et al. 2016; Mace et al. 2018; Foden et al. 2013, 2019). The Living Planet Index, a measure of the state of the world's biological diversity based on trends in vertebrate populations (Ledger et al. 2023) recorded an average 68% fall in monitored (non-human) vertebrate species populations between 1970-2016, while the 2019 assessment from the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) estimated that the average abundance of native species in most major land-based habitats has fallen by at least 20% compared to pre-industrial levels (WWF 2020; IPBES 2019). Globally, wetlands are vanishing three times faster than forests, and freshwater vertebrate populations have fallen more than twice as steeply as terrestrial or marine populations (Tickner et al. 2020). The IUCN Red List reports more than 42,100 species assessed as threatened with extinction (roughly 28% of the total number assessed) (IUCN 2023a). Those threatened include: 41% of all amphibian species, 27% of mammals, 13% of birds, 21% of reptiles, 37% of sharks and rays, 36% of reef forming corals, 34% of conifers and 69% of cycads. In addition, 10% of invertebrates have been tentatively estimated to be under extinction threat (IPBES 2019). Further, based on an analysis of extinction trajectories, the rate of extinctions is predicted to increase in future (Monroe et al. 2019).

Beyond the ethical considerations of how humans interact and behave towards other species, this loss is understood to have significant implications for human health and well-being, with negative impacts projected for: agriculture and food security; livelihoods; protection against a shifting climate; access to water and basic materials; disease outbreaks and access to medicines; as well negative implications for non-material benefits to physical, social and spiritual well-being (Díaz et al. 2006, 2019; Pilling et al. 2020; Bawa et al. 2020; Methorst et al. 2021; Schmeller et al. 2020).

1.4 A GLOBAL RESPONSE

"Conservation needs to move beyond random acts of kindness and instead mobilise strategic, coordinated action to save the planet." (Sally Jewel, former US Secretary of the Interior, IUCN World Conservation Congress 2016).

Local, isolated efforts to conserve nature date back hundreds of years (e.g. Evelyn, 1664). However, global, systematic efforts escalated during the 1900s as the limits of the earth's natural resources were increasingly tested (Groombridge & Jenkins 2002; Hall et al. 2009). Beginning with the International Whaling Commission (IWC) in 1946, a series of international agreements were forged towards more sustainable treatment of biodiversity. The most comprehensive of these with respect to biodiversity coverage and implications for species, emerged from the 1992 United Nations (UN) Rio Summit, which galvanised the world's governments behind a single treaty initiative, the Convention on Biological Diversity (CBD), aimed at conserving nature across the globe. The CBD is an international legally binding treaty that embeds responsibility and accountability for nature conservation within national government structures. The treaty has three main goals: conservation of biodiversity; sustainable use of biodiversity; and fair and equitable sharing of the benefits arising from the use of genetic resources (CBD 2022). In 2023, the Convention has been ratified by 196 nations. The CBD, which also encompasses the Nagoya Protocol on, "*Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilisation*", and the Cartagena Protocol aimed at protecting biodiversity from the international movements of living modified organisms arising from biotechnology, works in concert with a range of other international agreements relevant to conserving species (Rogalla von Bieberstein 2019; Kreienkamp 2022) (see **Figure 1.2**).

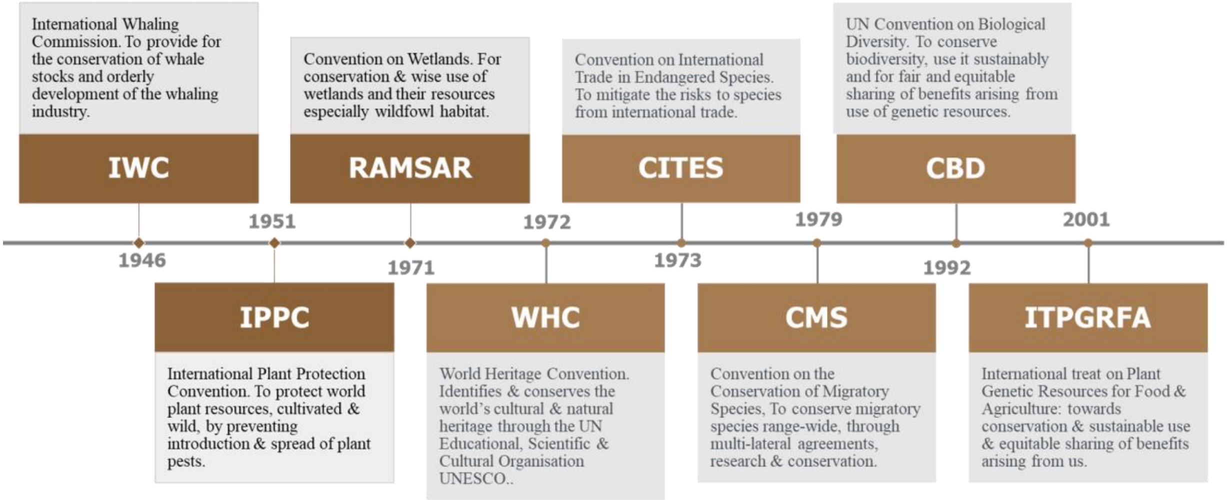

Figure 1.2 Timeline showing the main international agreements relevant to conserving species (information from Rogalla von Bieberstein 2019 and https://www.cbd.int/brc/).

Nations that are signatories to these conventions enact their commitments through national regulatory frameworks (which in some countries pre-date the CBD). Of particular relevance, and an obligation and reporting requirement of the CBD, are the National Biodiversity Strategy and Action Plans (NBSAPs) which support the mainstreaming of biodiversity into the policies of key economic sectors, such as agriculture, forestry and fisheries (Whitehorn et al. 2019).

Despite failures to date, the CBD continues to be one of the most significant hopes for global-scale movement on biodiversity conservation. The new Kunming-Montreal CBD Post-2020 Global Biodiversity Framework includes four long-term goals to 2050 (**Box 1.1**), each with associated milestones and action-oriented targets to 2030. National commitments to these will form the backdrop to the next ten years of conservation planning and action. Target 4 of Goal A is the most directly relevant to threatened species (**Box 1.1**), though significant movement on all 23 targets will be needed to ensure that progress is possible, sustainable and welcomed and that many more common species do not find their way onto threatened species lists (CBD 2022).

1.5 APPROACHES TO PLANNING THE CONSERVATION OF BIODIVERSITY

"Conservation planning uses the best scientific information to ensure that natural systems are conserved as human-induced change takes place." (Craighead & Convis, 2013).

Box 1.1 Kunming-Montreal Global Biodiversity Framework goals for 2050 and Target 4 (adopted 22/12/2022; CBD 2022)

Goal A: The integrity, connectivity and resilience of all ecosystems are maintained, enhanced, or restored, substantially increasing the area of natural ecosystems by 2050; human induced extinction of known threatened species is halted, and, by 2050, the extinction rate and risk of all species are reduced ten-fold and the abundance of native wild species is increased to healthy and resilient levels; the genetic diversity within populations of wild and domesticated species, is maintained, safeguarding their adaptive potential.

TARGET 4 (relevant to threatened species)

Ensure urgent management actions to halt human induced extinction of known threatened species and for the recovery and conservation of species, in particular threatened species, to significantly reduce extinction risk, as well as to maintain and restore the genetic diversity within and between populations of native, wild and domesticated species to maintain their adaptive potential, including through *in situ* and *ex situ* conservation and sustainable management practices, and effectively manage human-wildlife interactions to minimize human-wildlife conflict for coexistence.

Goal B: Biodiversity is sustainably used and managed and nature's contributions to people, including ecosystem functions and services, are valued, maintained and enhanced, with those currently in decline being restored, supporting the achievement of sustainable development for the benefit of present and future generations by 2050.

Goal C: The monetary and non-monetary benefits from the utilization of genetic resources and digital sequence information on genetic resources, and of traditional knowledge associated with genetic resources, as applicable, are shared fairly and equitably, including, as appropriate with indigenous peoples and local communities, and substantially increased by 2050, while ensuring traditional knowledge associated with genetic resources is appropriately protected, thereby contributing to the conservation and sustainable use of biodiversity, in accordance with internationally agreed access and benefit-sharing instruments.

Goal D: Adequate means of implementation, including financial resources, capacity-building, technical and scientific cooperation, and access to and transfer of technology to fully implement the Kunming-Montreal Global Biodiversity Framework are secured and equitably accessible to all Parties, especially developing country Parties, in particular the least developed countries and small island developing States, as well as countries with economies in transition, progressively closing the biodiversity finance gap of $700 billion per year, and aligning financial flows with the Kunming-Montreal Global Biodiversity Framework and the 2050 Vision for biodiversity.

Biodiversity refers to the variety of all living things and their interactions and includes ecosystems, species and genetic diversity. Conserving biodiversity involves taking action to sustain biodiversity content (what ecosystems, species and genes exist), patterns (the variations in this content that occur in different places or situations) and processes (the natural spatial, physical and evolutionary factors that influence changes in content and pattern over time) (Kreienkamp 2022; CBD 2022).

Though this study focusses on planning for species identified as threatened with extinction, ensuring conservation of biodiversity requires planning for all species, to prevent declines that could lead to them becoming threatened in future and to maintain them in sufficient abundance to fulfil their ecological roles (Monroe et al. 2019).

In-line with CBD targets, biodiversity conservation should include: setting aside large areas of the planet where nature can remain relatively intact; implementing rules, incentives and deterrents that will protect it from human activities and influences within and outside those areas; and restoring it to areas from which it has been lost. To enable and sustain this, biodiversity benefits to people must be recognised, realised and shared equitably, and the tools, solutions and resources needed to achieve all of this must be readily available to all relevant societal sectors (CBD 2022).

Planning for this is difficult and complex and is made more so by the many, often apparently competing, approaches available (Margules & Pressey 2009; Groves and Game 2016). For a given area, country, region or continent, biodiversity conservation planning is likely to benefit from a multi-layered approach involving several different but complementary planning methods. Groves & Game (2016) identify and describe eleven methods that can work together in this way. For simplicity here these are collapsed into the following four broad categories of biodiversity conservation planning:

1. **Spatial planning for biodiversity:** planning where nature will be conserved as a priority, on land or in the sea.
2. **Management planning for biodiversity:** planning how biodiversity and ecosystem services will be managed, restored or protected inside and outside formally protected areas.

3. **Systemic threat reduction planning:** planning the mitigation of widespread, high impact threats that may require multi-disciplinary approaches.
4. **Species conservation planning:** *in situ* and/or *ex situ* planning for species declining or failing to recover despite or in absence of other biodiversity conservation efforts.

The fourth of these is the main subject of this study. However, the first three have the potential for the greatest impact on the greatest number of species and it is past or current failures in these that give rise to the need for species-specific planning and action, which can be an intensive and expensive way to protect biodiversity at scale (Martin et al. 2012; Woellner & Wagner 2019; Gordon et al. 2020). To explain the case for species-focussed attention, this section begins with a brief description and examples of the first three, and some of the reasons why they are unable to drive effective action for all the species that need it.

1.5.1 Biodiversity spatial planning

About three-quarters of the Earth's land surface has been altered by humans within the last millennium and only 13% of the oceans now meet the definition of marine wilderness (Winkler et al. 2021; Jones et al. 2018). Conversion of natural areas to other purposes, in particular agriculture, is the biggest threat to terrestrial biodiversity (WWF 2020). Setting aside areas on land and in the oceans where nature can persist relatively intact or at least be under increased protection from human activities can make a dramatic contribution to conservation provided the areas selected are in the right places, are large enough, and are sufficiently inter-connected (Margules & Pressey 2000; Alexander 2013; Harris et al. 2019).

The CBD 2030 global target is to protect and restore 30% of land and marine areas (CBD 2022). To date, coverage extends to around 17.2% of the global land area and 8.26% of the oceans but these figures are increasing (UNEP-WCMC & IUCN 2023).

As of 2023, 244 countries have protected areas of one kind or another (UNEP-WCMC & IUCN 2023). Setting aside marine areas dates back at least to the early part of the 20th Century and terrestrial examples such as the royal hunting forests of Europe and the sacred groves of Africa and Asia date back hundreds of years (Humphreys & Clark 2020; Samojlik et al. 2020; Sheridan, 2009; Chadrashekura & Sankar, 1998). For historical, cultural, practical and economic reasons existing protected areas will inevitably form the backbone of any future national nature networks, regardless of their biodiversity properties. Planning must account for this when identifying new areas or re-drawing existing ones, to ensure optimal biodiversity outcomes for any combined network (Groves & Game, 2016). Though national efforts are key, the identification of areas for nature conservation can be done at many scales. These range from local nature reserves through national and regional networks such as the EU Natura 2000 network, to Global initiatives such as the IUCN's Key Biodiversity Areas (KBA) project which aims to ensure that networks of globally important sites, identified based on their vulnerability and irreplaceability with respect to the globally important species populations they support, are properly safeguarded (Eken et al. 2004; Langhammer et al. 2007).

Multiple planning approaches have emerged to help tackle the challenge of optimising the location, size and connectivity of area or ecological networks to maximise biodiversity conservation alongside other benefits and their similarities and differences are reviewed elsewhere (Pressey & Bottrill 2009; Groves & Game, 2016). The most documented approach (referred to as Systematic Conservation Planning) is illustrated in **Box 1.2**. Though often described using terrestrial examples, Systematic Conservation Planning is equally relevant to planning marine reserve networks (e.g. Green et al. 2009; Kirkman et al. 2019; Harris et al. 2019).

Box 1.2 An example of a landscape and seascape planning approach: Systematic Conservation Planning (from Margules & Pressey 2000).

This approach assumes reserve networks should represent the biodiversity of a region and separate it from threatening processes. It assumes existing networks will contain a biased sample of biodiversity (usually that of remote places and areas unsuited for commercial activities). The planning stages involved are:

Stage 1. Compile biodiversity data: including on the locations of rare or threatened species.

Stage 2. Define conservation goals for the region: identify surrogates to represent biodiversity across the planning region and set goals for them (e.g. at least three occurrences of each species, 1,500 ha of each vegetation type) as well as goals for minimum size, connectivity or other design criteria.

Stage 3. Review existing areas: measure the extent to which existing protected areas achieve the targets specified.

Stage 4. Select additional areas that will better support agreed goals: supported by reserve selection algorithms and decision support software (e.g. Marxan (Ball et al. 2009)) able to factor in constraints such as existing reserves, budget and opportunity costs for other land-uses.

Stages 5. Determine conservation actions needed: where these prove too onerous, reconsider inclusion and return to Stage 4.

1.5.2 Biodiversity management planning

The USA Wilderness Act (1964), which has led to the establishment of more than 800 federally designated wilderness areas defines wilderness as, "An area where the earth and its community of life are untrammelled by man, where man himself is a visitor who does not remain". Such large, strictly protected areas where nature and natural processes can remain intact are rare and areas designated for nature, including many protected areas, often accommodate multiple uses. The IUCN recognises six categories of protected area from those that are strictly protected for biodiversity values to others that integrate human activities to a greater or lesser extent, including forestry, agriculture, fisheries and human habitation (Dudley, 2008). Wherever human pressures are present, restoration, management and monitoring are likely to be required to ensure biodiversity potential is realised and maintained and that problems are caught and addressed early. This extends beyond formally protected areas to any and all areas where biodiversity is valued.

Approaches have been developed to deal with planning the specifics of managing and monitoring individual sites, areas, landscapes or seascapes, to ensure that biodiversity features (including habitat types, species and ecosystems) are preserved. These result in action plans for management towards identified goals, that can be regularly reviewed and revised based on monitoring data. Typical steps in biodiversity conservation management planning are described in **Box 1.3**, for the Conservation Action Planning (CAP) method practised by The Nature Conservancy, a front runner in this field (Poiani et al. 1998; TNC 2007; Carr et al. 2017).

Other, closely aligned methods in this category include the Conservation Measures Partnership's Open Standards for the Practice of Conservation (Schwartz et al. 2012) and the Wildlife Conservation Society's Landscape Species Approach (Sanderson et al. 2002b; Didier et al. 2009).

Also relevant is the IUCN's Green List Standard which describes a set of seventeen criteria and 50 indicators for successful conservation in protected and conserved areas, including provisions for species conservation, providing an international benchmark to support good, protected area planning (IUCN-WCPA 2017).

Box 1.3 A representative framework for site-based biodiversity management planning (from Poiani et al. 1998)

Steps and relevant questions asked, for site conservation planning in The Nature Conservancy.

1. **Assemble teams:** Who should be included in the planning process and implementation of the plan?
2. **Agree targets and goals:** What are the significant conservation targets (e.g. ecosystems and their services, natural features, habitats, species) and long-term goals for those targets?
3. **Gather ecological information:** What biotic and abiotic attributes maintain the targets over the long term?
4. **Gather human context information:** What are the basic characteristics of the human communities at the site?
5. **Analyse threats:** What current and potential activities interfere with the survival of the conservation targets and the maintenance of ecological processes?
6. **Identify stakeholders:** Who are the organized groups and influential individuals at the site, what impacts might we have on them, and how might they help or hinder us in achieving our goals?
7. **Develop conservation strategies:** What can we do to prevent or mitigate threatening activities, and how can we influence important stakeholders?
8. **Identify conservation zones:** What are the areas on the ground where we need to act?
9. **Define and describe actions:** What kinds of actions are necessary to accomplish our goals, who will do them, how long will they take, how much will they cost?
10. **Assess feasibility:** Can we succeed in our goals, based on assessment of ecological and human context concerns and programmatic resources?
11. **Agree measures of progress:** How will we know if we are making progress toward our goals and if our actions are bringing about desired results?

1.5.3 Systemic threat-reduction planning

Threat identification and mitigation is a normal part of biodiversity conservation management planning for specific sites or areas. However, for some threats, site-based action is not enough, and more systemic planning and action is needed. Examples are those threats that travel with species or across multiple sites or areas, or that require the input of multiple disciplines for effective mitigation. These threats include illegal trade in high-value animal or plant parts, whose effective mitigation may require a combination of spatially explicit site-based protections, disruption of trade routes and campaigns to change consumer behaviour. Other complex, multi-dimensional challenges include invasive species, dispersed pollutants (including pesticides and nitrates), some diseases (such as amphibian chytridiomycosis) and climate change impacts (Challender et al 2015; Jain et al. 2018; Hoffman & Challender 2020; Palmer & Mclauchlan 2023; Skerratt et al. 2007; IUCN SSC HSG/CPSG 2022 pp: 61-66).

Hutchings et al. (2004) discuss the development of threat abatement plans as a statutory requirement under Australia's Threatened Species Conservation Act 1996 (see **Box 1.4**). This provides a framework for prioritising the allocation of limited resources to reduce the adverse impacts of key threatening processes of which there are 22 currently listed (DCCEEW 2022).

Box 1.4 Example: Australia's Threat Abatement Planning Framework for Invasive Species (from Hutchins et al. 2004).

Objectives:

- Target abatement where the impacts of specific pests are likely to be greatest;
- Develop best practice guidelines that maximise the effectiveness of control programs
- Minimising non-target impacts;
- Establish monitoring programs to demonstrate impacts and to measure the effectiveness of the resulting control programs;
- Identify knowledge gaps and develop research proposals where information is lacking;
- Increase community education and involvement.

Other examples of planning processes or systems designed to mitigate single threats to species are harvest management plans, which support the regulation of harvests taken from wildlife populations to prevent over-exploitation, guided by first principles or by harvest models developed for a specific system (Sutherland 2001). Examples are Canadian management plans for grizzly bears (e.g. Nagy & Branigan 1998) and Alaskan plans for caribou (e.g. Harvest Management Coalition 2019). Sustainable harvesting is also a key component of Fisheries Management Plans which are required by law in some countries and states (e.g. Queensland, Australia and the USA). Fisheries Management Plans typically include: a description of the fishery especially its status and any established user rights; the management objectives; how these objectives are to be achieved; how the plan is to be reviewed and/or appealed; and the consultation process for review and appeal (Die, 2002).

Also relevant here is the IUCN's Species Threat Abatement and Restoration (STAR) metric, designed to help countries identify systemic threat reduction opportunities using IUCN Red List data, and to plan policy measures that link action on these threats to global species targets (Mair et al. 2023).

1.5.4 Shortcomings in the designation and management of protected areas

The well-chosen designation and then effective management of networks of terrestrial and marine areas have the potential to deliver huge benefits to species (Margules & Pressey 2000; Alexander et al. 2013; Watson et al. 2014). However, the needs of many species will not be met through this approach alone. Constraints related to cost, land-use history and competing interests, perpetuate a bias in protected area designation towards: areas of low commercial value under relatively little threat of land conversion; those that serve larger bodied, well studied taxa; and those where the threats are too difficult to mitigate to realise the intended protection (Margules & Pressey 2000; Joppa & Pfaff 2009; Kuempel et al. 2019). Area designations may be too small to accommodate minimum viable population sizes for some species. For mobile or migratory species, throughout-life needs may not be met as life-stage or temporal patterns of movement may not be fully encompassed by the size of the areas designated, area boundaries may interfere with important meta-population or source-sink dynamics and species' long-term requirements for habitat heterogeneity and succession may not be met where sites are relatively isolated (Harcourt 2002; Hansen 2011; Ivanova & Cook 2023). Lack of information on which to base decisions, lack of funding, and inadequate political support, are also regularly cited barriers to protected areas networks achieving their potential (Margules & Pressey 2000; Groves & Game 2016; Giehl et al. 2017; Roberts et al. 2019; Watson et al. 2014).

Management plans developed for sites can result in benefits to species that are an explicit focus but may miss the needs of others for which data are less available and needs underestimated (Wiens et al. 2008; Jones et al. 2016). Further, addressing threats (such as pollutants) arriving from adjacent areas may fall outside the purview of area managers. At a more basic level, due to lack of resources or capacity, management plans do not exist at all for many protected areas, where they do exist they may not be sufficiently resourced for implementation, and information on management effectiveness is available for less than 1% of the World's 230,000 protected areas (Kendall et al. 2015; UNEP-WCMC 2018).

In the context of protecting species, the gap between theory and practice in the designation of protected area networks in Australia is highlighted by Watson et al. (2011b) and reinforced by Kearney et al. (2020), see **Box 1.5**. Similarly, Fonseca & Venticinque (2018) highlight shortcomings in Brazil's protected area network and McIntosh et al. (2017) provide a broad-scale systematic review of the current state of knowledge about the outcomes of Systematic Conservation Planning projects, identifying opportunities to improve evaluation and information sharing across the discipline.

Box 1.5 Protected area network designation and management alone will not conserve and recover threatened species: lessons from Australia.

Watson et al. (2011) assessed Australia's terrestrial protected area system and found that 166 (12.6%) of species were not covered at all and 259 (19.6%) were covered inadequately. Optimally shifting the amount of area protected at the time (11.6% of the area of Australia) would have resulted in meeting targets of 1272 (93.3%) of threatened species. Extending the area optimally across a total of 17.8% of Australia's landmass would have adequately protected all threatened species. However, pre-existing land uses and available resources would render these options impracticable. They advocated instead a mix of new, well-positioned and managed protected areas in combination with conservation management for species inside and outside the protected areas network.

Kearney et al. (2020) reinforced these findings, in a study of the pressures facing Australia's threatened species, finding that 52% of species faced one or more threats that could not be mitigated by protected area management actions alone, emphasising again the importance of investing in coordinated management for species both inside and outside the protected area networks.

1.5.5 Shortcomings of systemic threat reduction planning

High impact threats cannot always be mitigated fast enough, or to the extent needed. In Australia, the eradication of invasive species over large areas is rarely if ever possible (Hutchings et al. 2004; Emery et al. 2021). The impacts of chytridiomycosis, though the primary cause of dramatic global declines and extinctions of hundreds of amphibian species since 1980 and despite much focused attention by conservationists, remain unable to be mitigated in the wild (Skerratt et al. 2007; Gerber et al. 2023).

The species-specific impacts and tolerances of known threats are not always well understood and so can be difficult to plan for. This can be a particular problem for threatened species, for which observing or studying threat impacts can be especially difficult due to scarcity. In Europe this has resulted in, for example, allowable pesticide and other toxin limits being calibrated to the needs of more resilient species within a taxon due to their greater availability for testing, leaving more susceptible species highly exposed (e.g. the use of the relatively resilient *Episyrphus balteatus* as an indicator species for hoverfly toxin impacts (IUCN SSC HSG/CPSG 2022 pp: 61-66)).

Regarding marine and terrestrial over-harvesting, Sutherland (2001) points to multiple reasons why, even where science-based planning efforts are in use, over-exploitation remains common. These include: the difficulties of quantifying density-dependence and of accurately measuring population growth (which are required to establish sustainable offtake); the fact that it is better to monitor the population than the harvest, though easier to monitor the latter; and that increasing harvest effort is easier that reducing it.

Issues of ethics and social licence are another barrier to good planning and action on systemic threat reduction. For example, when applied at the right scale and frequency, aerial baiting with 1080 poison has been shown to be an effective strategy for protecting native birds from introduced mammalian predators in New Zealand (Griffiths et al*.* 2015; Robertson et al. 2019) but animal welfare and other concerns make its use controversial (Green & Rohan 2011).

1.5.6 Issues of small population size

Conservation measures, even where effective, may not work quickly enough. Many species now persist as small or highly fragmented populations. Once species abundance has deteriorated in this way, area protection and threat-abatement may not be sufficient. Stochastic effects may continue the extinction process even once external threats are removed (Goodman 1987). Such species may require more urgent, intensive and concerted intervention *in situ* and sometimes also *ex situ*, to prevent extinction and drive recovery (Foose et al. 1995; Frankham et al. 2017; Heywood et al. 2018).

1.6 The role of species conservation planning

As described above, current combinations of landscape and seascape, ecosystem, and threat-based biodiversity conservation planning continue to leave many species exposed to extinction risk or failing to recover. These species may be subject to multiple, poorly known or intractable threats, have extensive or complex spatial requirements, be closely connected to politically charged issues or competing human interests, or be simply too small to recover without intensive intervention. Such species may all benefit from plans that are developed to address their needs more specifically and comprehensively. Species conservation planning aims to fill this need and promote action that is swift enough to halt and reverse declines before it is too late (Byers et al. 2022; Rossi et al. 2016). **Figure 1.3** illustrates the role species conservation planning plays in relation to other approaches.

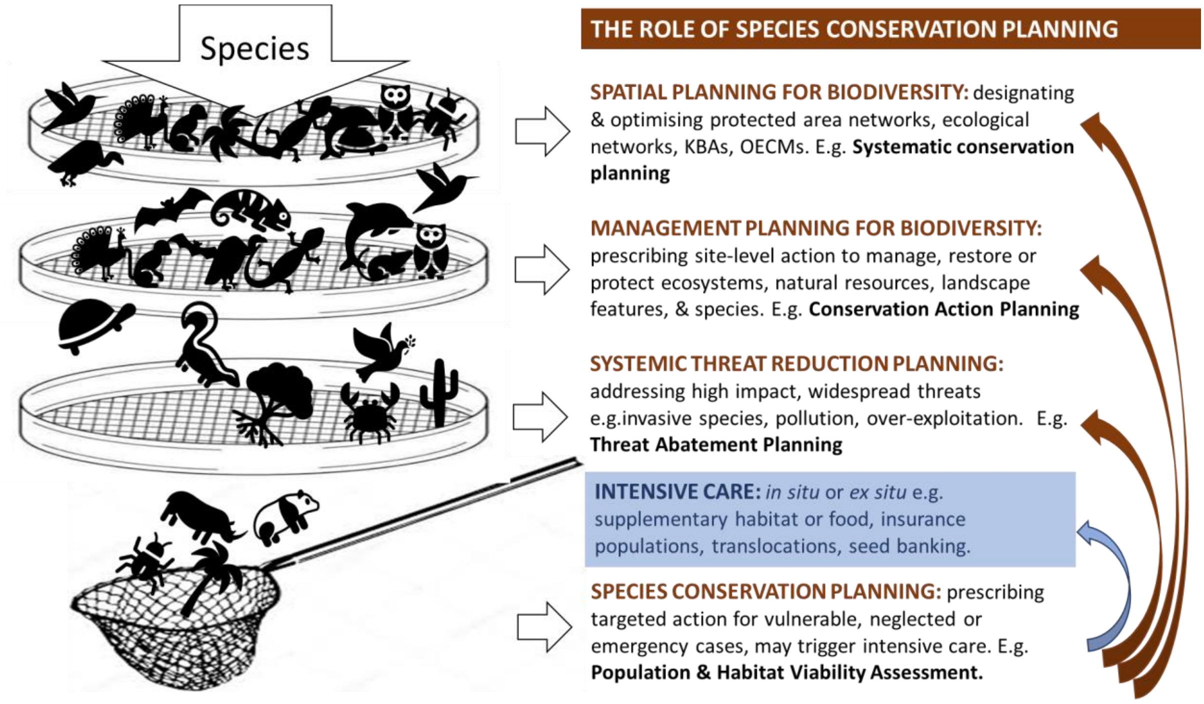

Figure 1.3 Illustration of the way species conservation planning supports biodiversity conservation by focusing on species whose needs are unlikely to be met by other approaches. Species conservation planning takes a deep dive into why species are declining or failing to recover, and evaluates a broad array of potential solutions, to recommend priority conservation actions. In many cases, some form of intensive care *in situ* or *ex situ* is prescribed (blue arrow and box). Such intervention may be critically important to prevent extinction and drive recovery but is unlikely to emerge from other planning approaches. Recommendations from species conservation planning may be usefully integrated into spatial, site-level biodiversity management, or threat abatement planning (brown arrows). KBAs=Key Biodiversity Areas; OECMs=Other Effective Area-based Conservation Measures. **Bold font** indicates example planning methods.

1.7 How many species need a conservation plan?

As a rough guide, the 42,100 species currently categorised in the IUCN Red List as globally threatened can be considered indicative of the number of species currently known to be falling through gaps in other measures and in need of targeted planning and action (Saiz et al. 2003; Rossi et al. 2016).

This is likely to reflect a minimum number as it is based on results from an assessment of only 150,300 of the ~1.9 million species so far described, out of an estimated ~8.7 million species planet-wide (Roskov et al. 2000; Mora et al. 2011; IUCN 2023a).

Though this seems an onerous task, plans do not have to be detailed documents. For many species, relatively simple action prescriptions integrated into other plans could be sufficient. Nevertheless, for many other species, detailed planning for intensive action *in situ* and sometimes *ex situ* is needed. Based on an analysis of IUCN Global Red List data, Bolam et al. (2023) estimate the latter to be 4035 species or a median of 54 species per country.

The total number of globally threatened species already covered by a plan of some kind is unknown. However, as shortfalls are reported in well-resourced countries (Kraus et al. 2021; Vercillo 2023; DCCEEW 2022) and as in most places there is little provision for species conservation planning (Heywood et al. 2018), we can assume that the number covered is relatively small.

1.8 Summary

The current biodiversity challenge is large and species conservation planning has an important role to play in keeping thousands of species from extinction. Globally, the gap between the species that need targeted planning and those receiving it is large. To begin to address that gap we need to understand why it exists. One possible reason is that too little information is available about how to plan for species effectively. Another is that planning for thousands of species is too onerous a task and national or global mechanisms for planning cannot cope with the challenge.

In Chapter 2, I review literature from governments in different countries, from NGOs working in planning and from academic studies, to understand what advice there is about species conservation planning and how much agreement exists on what constitutes good practice. Next, I review information from a small number of countries that are planning on a large scale, to identify common features or tools that could be considered minimum requirements for a planning framework able to generate and implement plans effectively, for the species that need them.

CHAPTER 2. SPECIES CONSERVATION PLANNING APPROACHES AND LESSONS TO DATE

"Species conservation planning is a systematic process focused on identifying and developing implementable actions to conserve species, the ecological processes that sustain them and the ecological services that are provided by them." Adapted from Groves & Game (2016).

2.1 ABSTRACT

Thousands of species conservation plans for threatened species have been produced but many more are needed. They are usually developed through national or regional government agencies or by international or national non-governmental organisations. Though there is much peer-reviewed literature about species conservation in general, the subset relevant to species conservation planning is more limited, has sometimes provided conflicting conclusions, and emanates from relatively few countries and organisations. Nevertheless, over time, what exists has converged on: a) what good content looks like for species conservation plans; b) how that content should be developed and c) how large numbers of plans can be supported through to successful implementation. Provided there is adequate resourcing, these three elements, delivered both nationally and internationally, would drive significant progress towards Target 4 of the Post-2020 Global Biodiversity Framework.

2.2 INTRODUCTION

Biodiversity conservation planning operates at different levels, from spatial planning through to species conservation planning (see Chapter 1 for detailed discussion). Here I discuss species conservation plans as a crucial tool for recovery of the world's most threatened species.

The aim of this review was to gain an understanding of how much is known about how to plan species conservation effectively and at scale. Rather than describe in detail all the material found, the review focused on how thinking in this area has evolved over the past 30 years, the findings of major reviews and the advice emerging from them, and any remaining knowledge gaps.

Different organisations have different naming conventions. Documents that lay out what needs to be done to conserve a species may be referred to variously as species conservation plans, species action plans, species conservation action plans, species conservation strategies and action plans, species recovery plans and conservation advices.

Here, “species recovery plan” is used to describe government-led plans and “species conservation plan” is used as a general term to encompass all documents that outline plans for species conservation. Where other terms are used their points of difference are explained.

The review identified three main categories of material: species conservation planning guidance documents designed to support different audiences to plan effectively; multi-plan reviews of plan characteristics or effectiveness; and other studies providing commentary and insights into species conservation planning. The guidance documents are listed in **Table 2.1** showing their origins and scope.

Table 2.1. Details of species conservation planning guidance documents consulted in this review.

Title	Scope	Source
Essentials of a good recovery plan.	Australia all species	Burbidge 1996
Guidelines for action plans for animal species: planning recovery (Vol. 92). Council of Europe.	European nations all species	Machado 1997
Action Plans for the Conservation of Globally Threatened Birds in Africa: Species Action Plan Development Manual.	African birds.	Sande et al. 2005
Interim Endangered and Threatened Species Recovery Planning Guidance Ver.1.3	USA all species	NMFS-FWS 2010
Recovery Planning Handbook. Version 1.0.	USA all species	NMFS 2020
Strategic Planning for Species Conservation: A Handbook	General	IUCN SSC 2008
Guidelines for Species Conservation Planning. Version 1.0	General, with sections for plants, fungi, invertebrates, herpetofauna, marine fishes & invertebrates, freshwater systems	IUCN SSC Species Conservation Planning Sub-Committee (2017)
BGCI and IABG’s Species Recovery Manual	Global, for plants	Heywood et al. 2018
Species Conservation Planning Principles & Steps, Ver. 1.0	General	CPSG 2020

The main multi-plan reviews were: for Australia, by Moore & Waller (2004), Bottrill et al. (2011), Watson et al. (2011a) and Walsh et al. (2013); for Brazil, by Baptista at al. (2019) and

Vercillo et al. (2023); for Canada, by Mooers et al. (2007, 2010) and Kraus et al. 2021); for the UK, by Laycock et al. (2009, 2012); for New Zealand, by Cullen et al.(2005) and Seabrook-Davison et al. (2010); and for the USA by Hoekstra et al. (2002). The most comprehensive of these reviews was that undertaken in the USA which involved a large collaboration between government and universities and covered all aspects of recovery planning (Hoekstra et al. 2002; Clark et al. 2002; Clark & Harvey 2002). The Australian and Canadian conservation planning systems are largely based on the USA one and one review discusses these together (Kraus et al. 2021).

Other material emanates from China and Europe and there are occasional insights into planning in additional countries, from individual plans and from reviews by non-government organisations including Botanic Gardens International (BGCI) and the International Union for Conservation of Nature (IUCN) (IUCN SSC 2002; Fuller et al. 2003; Heywood et al. 2018). In addition, reviews of plans across individual taxa provided useful information, for example, Roberts and Hamann (2016) for marine turtles, Reuter et al. (2022) for primates and Rossi et al. (2016) for plants.

2.3 WHAT SHOULD GO INTO SPECIES CONSERVATION PLANS?

According to the sources consulted, the elements that typically make up a species conservation plan are as follows:

- a **definition of success** for the species' conservation;
- a description of the **obstacles to success**
- a set of **objectives** to address those obstacles over the life of the plan;
- **actions** to describe who will do what, where and when;
- a **description of intended implementation**: how the planned conservation effort will be organised, monitored for impact, and adapted as needed.

Each of these elements is covered below and, in addition, a section is included that describes common methods used by groups to develop this content collaboratively.

2.3.1 A DEFINITION OF SUCCESS

Note that here a "definition of success" refers to a preferred future state or conservation end point for a species and not to shorter-term changes in the threats or challenges facing its conservation, which are a means to that end (discussed later under "objectives").

Restoring species to a state unaffected by human pressures is rarely possible. Even defining that state often presents a challenge. Nevertheless, those responsible for the recovery and conservation of species must make decisions about how much conservation is enough. Though pivotal to directing action and frequently called for, a description of clear and appropriate end goals within species plans is often lacking (Tear et al. 1995, 2005; McNeely 2000; Lundquist et al. 2002; Scott 2005; Redford et al. 2011; Akçakaya et al. 2020).

In the USA and Canada, past reviews of government species recovery plans found goals for success set at or below existing population sizes or below the IUCN Red List thresholds for the Endangered category. In some instances, political, social, or economic considerations were found to have influenced recovery goals, in some cases reducing them below those prescribed strictly on biological viability grounds (Tear 1993; Mooers 2010).

Following these early findings, emphasis was placed on encouraging more quantitative and consistent goals for recovery, in creating clearer and more consistent links with species' biological status, and in increasing the integration of population viability analyses (PVA) for assessing extinction risk (Morris et al. 2002; Gerber & Hatch 2002; Tear et al. 2005; Schemske et al. 1994; Neel, 2012; Himes Boor 2014).

This emphasis on quantitative approaches was challenged by other authors on the basis that PVA is too data-intensive to be possible or reliable for many listed species. In addition, they pointed out that over-emphasis on minimum viable population sizes would not lead to sufficiently ambitious or ecologically relevant recovery planning as species can be safe from near-future extinction at considerably smaller population sizes, in fewer populations, at lower densities and at smaller range sizes, than previously or currently exist (Soulé et al. 2003; Redford 2011; Wolf et al. 2015).

At the same time, early expectations that much threatened species recovery work would involve a finite period of intensive action followed by a return to fully recovered self-sufficient status were exposed as unrealistic for many species. Species' continued failure to recover despite decades of protection and planning made it clear that ongoing conservation management dependence would play a large and increasing role in species conservation requiring better accommodation within definitions of success (Goble 2009; Scott 2005; Sanderson et al. 2008; Redford 2011). Around the world, practitioners often advocate a two-tier definition of success encapsulating both an aspirational, descriptive "vision" of an ideal future state, along with a set of goals that describe this vision in operational terms that can be measured (e.g. Sanderson et al. 2008; NMFS 2020; IUCN SSC 2008; CPSG 2020). See **Box 2.1** for an example of a vision statement used as a basis for conservation planning for the North American Bison, *Bison bison*. While the operational goals are the essential component, the visioning piece can be a useful tool to help practitioners think beyond concepts of minimum population viability and short-term threat reduction and to explore more ambitious themes of ecological restoration and cultural connection (IUCN SSC 2008).

Box 2.1. Extract from a vision statement used to define successful recovery of the North American Bison, *Bison bison* (Sanderson et al. 2008)

"Over the next century, the ecological recovery of the North American bison will occur when multiple large herds move freely across extensive landscapes within all major habitats of their historic range, interacting in ecologically significant ways with the fullest possible set of other native species, and inspiring, sustaining and connecting human cultures."

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The recently established IUCN *Green Status of Species* standard has attempted to bring together these diverse perspectives into a single universal definition of a fully recovered species as one that, "*Is present, viable and ecologically functional in all parts of its indigenous range.*" The standard is designed to be applicable across taxa and spatial scales, accounts for conservation dependence, and provides for quantitative comparisons (Akçakaya et al. 2018, 2020; Grace et al. 2021). Though there remain challenges to its universal application it is anticipated this will be a valuable support tool for species conservation planners. Advice on defining success in plans is summarised in **Box 2.2.**

Box 2.2: Summary of advice on defining success for threatened species conservation planning (from Morris et al. 2002; Scott et al 2005; IUCN 2008; Redford et al. 2011; Mooers et al. 2010; Roberts & Hamann 2018, Akçakaya et al. 2018, 2020).

Species-specific definitions of successful recovery should:

- describe a desired, long-term future state for the species;
- consider representation, viability and ecological function across the indigenous range;
- be aspirational as well as objective and measurable;
- account for conservation dependence including, where needed, different *in situ* and *ex situ* management systems;
- connect to the well-being of stakeholders;
- where feasible and appropriate, be informed by quantitative, probabilistic tools such as population viability analysis;
- make a clear separation between scientifically derived biological requirements and factors related to feasibility, politics, and socio-economic factors;
- be documented clearly and periodically adapted based on the best available information.

2.3.2 Obstacles to success

Globally, in order of importance, the biggest threats to species are: changes in land and sea use; direct exploitation; climate change; pollution and invasive alien species (IPBES 2019). This pattern is often mirrored in national assessments (e.g. Lawler et al. 2002). However, species' declines or failure to recover are more often the result of multiple factors interacting in complex ways than by single factor cause-and-effect. Different species respond differently to the same threats, and threats may operate differently in different circumstances. Further, social, political, legal, technical or economic issues may be barriers to addressing direct threats even when they are well understood. Understanding these factors and their inter-relationships is important to planning effective action (e.g. NMFS 2020; IUCN SSC 2008).

To cope with this complexity various approaches are advocated for the elicitation, organization, visualisation and analysis of relevant information. These are variously referred to as threat, problem or situation analyses and generally include some or all the elements described in **Box 2.3.**

Box 2.3: Synthesis of advice on what to include in a problem or threat analyses in species conservation plans (from Sande, 2005; TNC 2007; NMFS-FWS 2010; IUCN SSC 2008; Roberts & Harmann 2016; Heywood 2018; CMP 2020; CPSG 2020).

Threats analyses should:

- document past, present and potential threats to the species, their drivers and interactions;
- clarify the impact of each threat on the target species including on different life-stages, in different areas and at different times of year;
- where possible, describe the severity, scope, reversibility, frequency of impact and any trends in threats;
- be explicit about the attributes of the species that make it vulnerable to identified threats or that would support it to recover;
- describe barriers to taking effective action to address the threats described, including social, political, economic, legal or technical factors;
- for threat-related information include explicit clarification of what is fact (with supporting evidence), what can be assumed (and on what basis) and key information gaps that hinder effective action;
- consider using quantitative methods to describe and compare threats;
- clarify relevant political, economic, social, technological, or legal barriers and opportunities for addressing threats;
- prioritise threats using transparent criteria related to (for example) impact on the species and/or feasibility of addressing them under prevailing conditions.

Further, to illustrate the known or hypothesized interactions between threats they are commonly illustrated in plans using diagramming tools such as causal flow diagrams, problem trees or mind-maps (IUCN SSC 2008, 2017; CMP 2020; CPSG 2020). See **Figure 2.1** for an example.

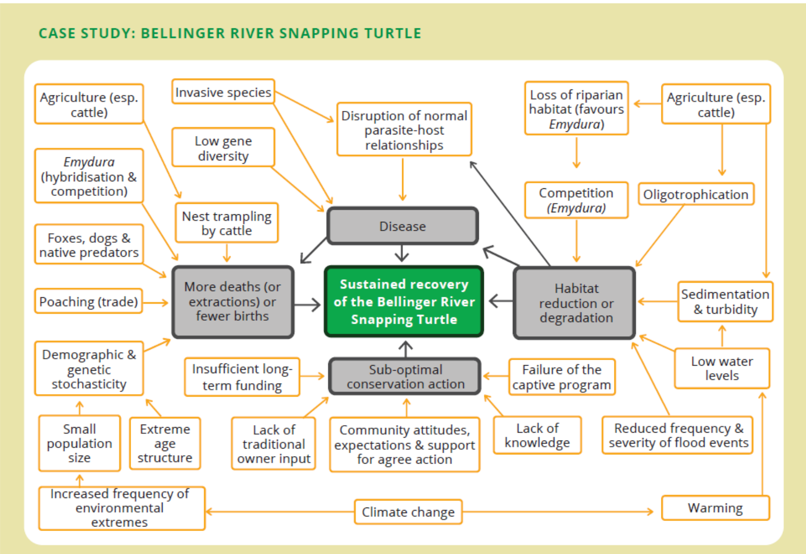

Figure 2.1. Illustrated list of current and potential risks to the Bellinger River Snapping Turtle *Myuchelys georgesi*, and their perceived inter-relationships (Jakob-Hoff et al. 2017)

While analysis and prioritisation of threats is commonly approached qualitatively, Runge et al. (2007) describe a quantitative threats analysis approach for the Florida manatee, *Trichechus manatus latirostris*, using comparative population viability analyses in which the population of interest is projected forwards under different scenarios regarding the presence, absence or partial absence of five specified threats. This allows evaluation of their relative importance, or of the amount of mitigation required to achieve specific viability thresholds (**Box 2.4**).

Box 2.4: Example of the outputs of a quantitative threats analysis for the Florida manatee (*Trichechus manatus latirostris*) (Runge et al. 2007)

- The estimated probability of the manatee population falling to less than 250 adults on either the Atlantic or Gulf coasts (from a current statewide population size of near 3300) within 100 years is 8.6%.
- Complete removal of the watercraft threat alone would reduce this risk to 0.4%.
- Complete removal of the warm-water threat would reduce risk to 4.2%.
- Removal of both threats would reduce the risk to 0.1%.


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The modelling approach taken also allows consideration of partial removal of threats, as well as removal of multiple threats simultaneously.

Another innovation in this area has been the development of a threats lexicon by Salafsky et al. (2008) to promote standardization and improved communication among practitioners and to enable review and analysis across projects. The descriptive hierarchy of threats provided is in wide use and an updated version currently underpins the IUCN Red List threat categorization standard, enabling standardized analysis and prioritization of threats across thousands of species (IUCN 2023b).

An initial review of the best available information about the species and its circumstances is recommended by all guidance documents, as a starting point to understanding all the obstacles to success, as well as the opportunities for remedial action. Key topics recommended include: the species' biology and ecology; population size, distribution and trends; critical habitat; human-mediated threats and challenges to addressing those threats; and past and current conservation action and its outcomes (e.g. IUCN SSC 2008; IUCN SSC Species Conservation Planning Sub-committee 2017; NMFS 2020). Drawing from a range of sources is advised. Depending on the species these might include peer-reviewed scientific literature and other published sources of evidence, and knowledge and insights of species specialists. The incorporation of local and Traditional Knowledge from communities with a long association with the species or its habitat is encouraged. The latter has proven valuable in terrestrial, freshwater and marine biodiversity conservation, providing not only information about traditional use but also historic and contemporary baseline information for species that does not otherwise exist (e.g. Fraser et al. 2006; Thornton & Sheer 2012; Biró et al. 2019; Wehi et al. 2019).

2.3.3 OBJECTIVES

Plans often operate over a 5–10-year timeframe whereas making significant progress on species recovery can take much longer (Abbitt & Scott 2001). A distinction is usually made between the long-term goals that define ultimate success for the species (see previous section on defining success) and the shorter-term objectives related to addressing the obstacles to that success over the life of the plan (NMFS 2020).

Well-formed objectives are essential to providing direction and tracking progress in plan implementation. Criticism in early reviews pointed to lack of clarity, absence of clear success measures and poor links with species' threats (Schemske et al. 1994; Gerber & Hatch 2002; Tear et al. 2005). Advice provided by Doran (1981), that objectives should be Specific, Measurable, Assignable, Realistic and Time-related (S.M.A.R.T.) has persisted over time and continues to be widely advocated across planning frameworks and in different countries. Its

Box 2.5. Summary of advice on setting objectives (from Tear et al. 2005; NMFS-FWS, 2010).

- **State objectives clearly:** Well-defined, unambiguous statements that are brief, specific and make clear what threat or challenge they will address.
- **Define measurable objectives:** Measurable by some standard scale (e.g., number or percent) over time (e.g, months or years) and space (e.g., for a political or ecological region like a state or ecoregion).
- **Include both near and long-term objectives:** i.e. those that prevent extinction and those that deliver long-term recovery.
- **Separate science from feasibility:** Science alone must drive the process for setting objectives. Once set, feasibility may then be considered to evaluate the likelihood of achieving the stated objectives.
- **Anticipate change:** expect objectives to change as knowledge and science change and employ the concepts of adaptive management.

use has been correlated with a higher likelihood of species undergoing recovery (Watson et al. 2011a). See **Box 2.5** for advice on formulating objectives.

2.3.4 Actions

An action can be defined as, *"any activity which will, directly or indirectly, contribute to improving the conservation status of the species involved*." (IUCN SSC 2008). Guidance generally advises actions should specify who will do what, when, where, and what indicators will be used to measure progress and completion. Some frameworks may require additional information such as an assessment of cost or feasibility (NMFS-FWS, 2010; Roberts & Hamann 2016; Heywood et al. 2018; IUCN SSC 2008).

Multi-plan reviews of recommended actions have criticised: under emphasis on development and policy issues; a failure to address the threats described; and an over-emphasis on research; with the latter linked to poorer recovery outcomes (McNeely 2000; Lawlor et al. 2002; Buxton et al., 2020). The need to limit research actions to those essential to achieving objectives is emphasised (NMFS-FWS, 2010).

A quantitative approach developed to help address this, called the Value of Information (VoI), helps identify the data uncertainties that are most and least important to resolve in terms of their expected impact on conservation outcomes (Canessa et al. 2015). It has been used successfully in planning for whooping cranes, *Grus americana*, Tasmanian devils, *Sarcophilus harrisii* and for amphibians at risk to chytridiomycosis (Runge et al. 2011; McDonald-Madden et al. 2010; Gerber et al. 2023). Other authors recommend the use of "results chains", which are designed to support planners to make the assumptions underpinning recommended actions more explicit and to facilitate their evaluation (Margoluis et al. 2013; IUCN SSC 2017). In other advice the importance of ensuring that the actions recommended in a plan are both necessary and sufficient to achieve objectives is emphasised (IUCN SSC 2008).

Prioritising actions based on their potential to contribute to recovery objectives, as well as on their likelihood of successful implementation, is recommended and there are specific recommendations to prioritise: actions necessary to prevent extinction; those needed to avoid significant further declines; followed by other activities necessary to achieve recovery

(Machado 1997; NMFS 2020). **Box 2.6** summarises advice on developing and documenting actions.

Box 2.6. Summary of advice on specifying actions (from McNeely 2000; Sande et al. 2005; Margoluis et al. 2013; IUCN 2008; Machado 1997; NMFS-FWS, 2010; NMFS 2020; CPSG 2020).

- Ensure recovery actions are discrete and action oriented, and their descriptions concise.
- Ensure documented actions record, as a minimum, who will do what, where, when, and how progress and success will be measured.
- Recovery actions that are dependent on the outcome of earlier actions should be indicated as such.
- Avoid over-emphasis on actions for research and monitoring and include only those essential to achieving stated goals and objectives.
- Check for a logical pathway from each action to the relevant objectives.
- Ensure the sum of recommended actions is necessary and sufficient to achieve planned objectives.
- Prioritise actions based on the opportunity to maximise recovery efforts and the feasibility of successful implementation.

2.3.5 Implementation, monitoring, adapting

This study is about planning rather than implementation. However, plans are built to be implemented and it is important that they are developed and documented in ways that maximise the likelihood of effective implementation. Here the focus is on individual planning projects. However, plan implementation is influenced by wider conservation culture and frameworks and this is discussed further in **Section 2.5.4.3.**

Though Abbitt and Scott (2001) provide evidence that the extent of plan implementation has a direct bearing on recovery progress, few studies report on this metric and the factors that influence it. Two reviews across multiple government-led plans in the USA (n=135) and UK (n=164) reported means of 70.3 – 78.2% partial or complete implementation of actions, though the range was 0-100% in both cases (Lundquist et al. 2002; Laycock et al. 2009). A recent study of 38 plans from Brazil covering 303 species found 58.12% of 2,044 actions proposed were partially or fully completed while 41.88% had not been started. Of the 874 completed actions, 39% were focused on generating knowledge, 13% on raising awareness and only 17% on controlling human activities that are damaging the species (Vercillo 2023).

Implementation rate statistics alone may not be useful indicators of progress towards success as some actions and objectives are more important than others. The projects used in the UK study reporting 78.2% completion rate were also surveyed for a government report, which recorded a lower rate of 39%. Laycock and colleagues attribute this to either the more complete response rate to the government study, or to a difference in accounting method. Theirs had first excluded, on advice from experts, actions deemed unnecessary.

Researchers across all three studies found higher implementation rates: for older rather than newer plans (indicating rates may improve over time) and for species with restricted ranges. In addition: higher implementation rates were found for single- rather than multi-species plans; for species with greater public profiles; for plans reviewed at least once; for animal rather than plant plans; for plans targeting shorter-lived species; and for plans characterised by simple operational frameworks including clear roles and responsibilities for implementation (Lunquist et al. 2002; Laycock 2009). The type of threats involved was also a predictor of implementation (Vercillo 2023).

Ensuring that every plan is embedded within a project cycle that: includes monitoring, regular review and adaptation, and is driven by a dedicated coordinator or other well-functioning administrative body, is widely recommended (Battisti, 2018; Heywood et al. 2018; IUCN SSC 2017; Lundquist 2002; Laycock 2009; NMFS 2020). The Conservation Measures Partnership (CMP) provides materials and tools to help practitioners integrate monitoring and evaluation provisions into the plan during its development (Salafsky et al. 2002; Margoluis et al. 2013; CMP 2020).

Implementation usually relies on multiple stakeholders (see section 2.7.1 below). The way in which stakeholder collaboration, communication and coordination is intended to work should be clear in the plan or in associated documentation. Further, the plan should enable all stakeholders, whether they were involved in developing the plan or not, to understand the rationale behind the plan and to recognize their role in its implementation (NMFS 2020; Natural England 2022).

Simple administrative structures with clear roles identified from the outset, and led effectively, are expected to lead to good results (Laycock 2009; Crees 2016). Authors caution against a one-size-fits all approach to managing implementation and recommend tailoring administrative structures to the needs of the plan (NMFS 2020).

In addition to the items specified above, the intended schedule for implementation and the resources required, should be agreed and described either in the plan itself or in an associated document (Roberts & Hamann 2018; NMFS 2020). Lack of resources and highly skewed allocation of limited resources is a widely cited and major limitation both to the development of plans and to their subsequent implementation (Cullen et al. 2005; Watson et al. 2011a; Laycock et al. 2012). Developing a resourcing strategy alongside the plan is recommended by Roberts & Hamann (2016). Also, plans themselves can be effective fund-raising tools (Reuter et al. 2022) and if that is an intended purpose it should inform the design of any outputs.

A summary of advice on how to promote the likelihood of effective implementation through plan content, is summarised in **Box 2.7.**

Box 2.7. Summary of advice on how plan content can improve effective implementation. (From Laycock et al. 2009; Roberts & Hamann 2018; Heywood et al. 2018; NMFS 2020)

- Ensure stakeholders will understand the rationale behind the plan and will recognize their role in its implementation.
- Include an implementation strategy that:
    - clarifies what structure or body will be responsible for plan implementation and how coordination and communication among stakeholders will be achieved;
    - describes how plan implementation will be monitored, reviewed and adapted;
    - is as simple as possible given the needs of the situation, with clear roles and responsibilities identified.
    - lays out a schedule for implementation and the resources required for it.
- Includes or references a strategy for resourcing implementation.

2.4 PLAN DEVELOPMENT

"Too many Action Plans over the years have been shelved because they were written by international species specialists with little or no input from other stakeholders, particularly range State government stakeholders whose authority is critical for implementation." IUCN Species Survival Commission Species Conservation Planning Taskforce (2008).

In addition to the content of plans, the way that plans are developed is cited as an important contributor to success. Key themes include stakeholder inclusivity and the use of specialised tools.

Those developing plans, whether NGOs or governments, rarely have the resources or the authority to directly implement all, or even most, recommended actions. Multiple and often diverse stakeholders are usually needed (NMFS 2020; Ruddock et al. 2007; Natural England 2022).

Stakeholders are defined as individuals or organisations who play a role in recovery activities or are affected by them, as well as those who demonstrate some combination of concern about planning outcomes, bring expertise, or have influence over plan implementation or acceptance (IUCN SSC 2008; USFWS-NMFS 2010, 2020).

There is evidence that including stakeholders from a range of disciplines and backgrounds from the planning stage can lead to more successful outcomes, with benefits to both conservation practitioners and to species (Boersma et al. 2001; Clark et al. 2002; Vredenburg & Westley 2003; Knight et al. 2006). Inclusion of stakeholders is advocated in all the planning guidance consulted but there are different ideas about what this means in practice. For example, guidance to USA recovery teams describes a cautious approach involving: transmitting information; receiving and acting on feedback; and encouraging and facilitating the uptake by stakeholders of planned conservation activities. Limiting representation of stakeholders on the recovery team to those bringing relevant expertise is also recommended and it cautions that too much involvement of stakeholders can slow recovery efforts (NMFS-FWS 2010, NMFS 2020). At the other end of the spectrum, the IUCN advocates the co-creation of species conservation plans by large, multi-stakeholder groups in facilitated workshops, to reveal and resolve issues and build early acceptance of planning outcomes (IUCN 2008; 2017; CPSG 2020).

In relation to the inclusion of stakeholders in planning, four commonly used formats are described below. Each has a sequence of steps and associated tools.

Population and Habitat Viability Assessment (Westley & Miller, 2003): this format, first developed by the IUCN SSC Conservation Planning Specialist Group (CPSG) in the 1980s, combines social science tools for promoting inter-organisational collaboration with Population Viability Analyses (PVA). This format is designed specifically for species conservation planning and is particularly valued for situations in which the species is reduced to small numbers, stakeholders are diverse and both *in situ* and *ex situ* management systems are needed (Westley and Vredenburg 1997; Byers et al. 2013; Lees et al. 2019). Examples of its application can be found at http://cpsg.org

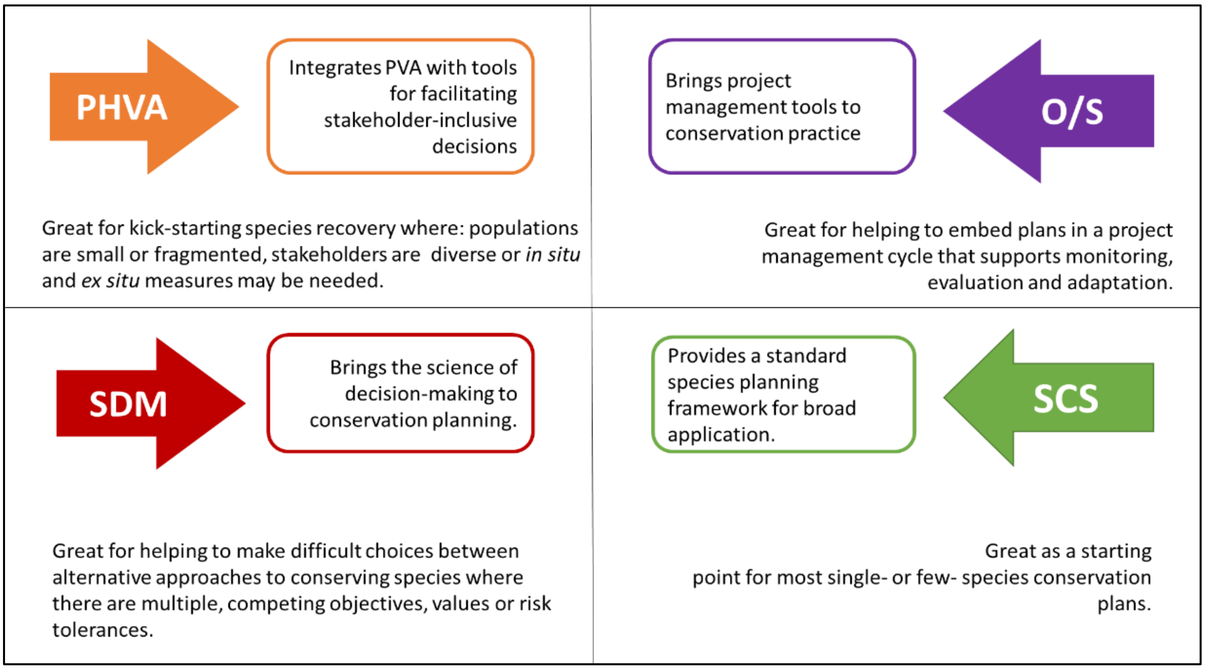
Species Conservation Strategy (IUCN SSC 2008): this format was first developed by a task force of the IUCN SSC following a review of the strengths and weaknesses of its previous species action planning initiatives (IUCN SSC 2002). Drawing from other planning approaches and advice it provides a general sequence of species conservation planning steps and guidance on how to complete them with stakeholders. The 2017 update provides specific advice and tools for planning for different types of taxa (e.g. invertebrates, felids, plants) and the 2020 update adds further practical emphasis.

Open Standards for the Practice of Conservation (CMP 2020): this format and its associated tools were first developed by the Conservation Measures Partnership coalition in 2002 for general nature conservation planning and are applicable to planning for species. The Open Standards (O/S) brings project management tools to conservation planning, helping to embed the finished plan in a project management cycle, promoting standards and measures for conservation practice and shared learning among projects. Examples of active projects can be found at: https://conservationstandards.org.

Structured Decision Making (Gregory et al. 2012): this format provides a generalised approach to evaluating alternative courses of action where there are multiple, competing, fundamental goals. It is used in a range of disciplines and can be applied to species conservation planning problems including cross-cultural ones where a values-based approach can be especially useful (e.g. McMurdo Hamilton 2016).

These approaches and their associated tools are not mutually exclusive and can be used in combination. **Figure 2.2** illustrates the potential strengths of each for species applications.

Figure 2.2. Four examples of step-wise planning formats used to develop species conservation plans, highlighting the particular strengths of each. Notes that these can be used together, or tools from some can be integrated into others where useful. PHVA=Population and Habitat Viability Assessment; O/S= Open Standards for the Practice of Conservation; SDM=Structured Decision Making; SCS=Species Conservation Strategy.



2.5 LARGE-SCALE SPECIES CONSERVATION PLANNING FRAMEWORKS

The previous section was about developing and writing individual species conservation plans. This section considers the elements involved in scaling-up planning so that large numbers of plans can be created and implemented quickly and effectively. Much can be learned about this from countries with an existing institutionalised practice of species recovery planning and from NGOs involved in developing plans.

Histories, reviews or accounts of current government planning frameworks or elements of them were found for Australia, Brazil, Canada, China, Italy, New Zealand, Spain, the USA and the UK (including separate information for Scotland and England) (Male 1995; Saiz 2003; Watson et al. 2011a; Baptista 2019; Vercillo 2023; Mooers et al. 2010; Kraus et al. 2021; Yang et al. 2015; Saiz et al. 2003; Rossi et al. 2016; Towns et al; 2019; NMFS-FWS 2010; NMFS 2020; Laycock 2009; Gaywood et al. 2016; Natural England 2022). Anecdotal information and insights about programs in additional countries were inferred from individual plans and overview documents. Machado (1997) reported on national recovery planning in the

Netherlands and Sweden but no recent records of these national plans were found and it is possible these programs are discontinued.

The information found is organized under the following themes:

- **Clarifying the purpose:** of the plans generated.
- **Defining planning pathways:** a set of planning options species can be directed to.
  - Multi-species versus single-species planning
  - Lower-intensity approaches
- **Prioritisation:** a way of deciding which species receive planning attention.
- **Enabling conditions:** elements supporting development of plans at scale, integrating or harmonising plans, and implementation.
- **Program-wide tracking and evaluation:** a central resource for plan-related information and approaches to measuring and reporting program-wide impact, to support learning and adaptation.

2.5.1 CLARIFYING THE PURPOSE

In some countries (e.g. Australia, Brazil, Canada, China, Japan, New Zealand, Spain, UK, USA) species conservation planning is government-led. Government bodies implement and release resources for, or at least are required to act in accordance with, the resulting plans (Male 1995; UK Government 2023; Kraus 2021). In these cases, there are often recovery groups or teams to promote, review and report on implementation progress and in some cases there is specific legislation to regulate it (Ewen 2013; DCCEEW 2017; Kraus et al, 2021). Though there are many reasons why these systems work imperfectly, they provide both a clear mandate for planning, a clear endpoint (often de-listing) and a pathway through which change for the species on the ground will occur and be monitored (e.g. Male 1995; Kraus et al. 2021; NMFS-FWS, 2010; Gaywood et al. 2016; Natural England 2022; UK Government 2023).

Plans initiated outside these frameworks, such as those developed by NGOs, do not always have the same clarity of purpose and route to impact. For example, a review of IUCN SSC Action Plans, developed for hundreds of species throughout the 1980s and 1990s, attributed low implementation rates in part to: those generating the plans having few resources and, therefore, no means of mobilising a target audience willing and able to act on the recommendations; and also to having no clear link from the comprehensive status reviews included to on-ground action recommendations (from IUCN SSC 2002 as cited in IUCN SSC 2008).

How the envisaged plans will lead to action on the ground should be clear from the outset as this plays an important role in determining how plans are developed, who is involved, and how success is later evaluated. For example, plans developed primarily as fund-raising vehicles in the first instance should be evaluated accordingly (e.g. Reuter et al. 2022). Those building plans that are expected to influence on-ground activities in a particular country or region should first understand and connect with the national agencies responsible for species conservation policy and planning (e.g. national wildlife authorities) to secure the necessary support and involvement (Heywood et al 2018; CPSG 2020).

Without these elements, finished plans may lay dormant, failing to reach their audience and their potential (Roberts & Hamann 2016; Heywood et al. 2018; CPSG 2020). **Box 2.8** summarises advice on documenting a clear purpose and route to impact for plans that are not initiated by governments.

Box 2.8: Summary of advice on ensuring a clear purpose and route to on-ground action in plans that are not government-led (from Roberts & Hamann 2016; Heywood et al. 2018; CPSG 2020; Reuter et al. 2022).

- Clarify the purpose of the plan and how it is expected to lead to action on the ground.
- Gain an understanding of the policy context for the plan, how it will support or interacts with government-led initiatives and responsibilities;
- Secure the support of the appropriate government wildlife authority;

2.5.2 DEFINING PLANNING PATHWAYS

For some species, successful conservation requires complex, coordinated management activities across multiple areas, across *in situ* and *ex situ* management systems and involving multiple agencies. Creating plans for these species takes time and can be labour and resource-intensive (for illustration, see Tamaraw example in **Figure 2.4**). For other species, conservation action can be planned and documented more simply and quickly. It makes sense to avoid a one-size-fits-all approach to planning. Innovation in this area has been driven by the need to stretch limited resources across increasing numbers of species and some examples of it are discussed below.

2.5.2.1 Multi-species versus single-species planning

Multi-species planning is defined here as planning that aims to address the specific conservation needs of several species as part of the same planning project. Included in this category are area management plans, ecosystem plans and threat reduction plans, where they are of sufficient scope and specificity to meet the definition. Reports were found of the use of multi-species planning by governments in Australia, Brazil, China, the EU and the USA; as well as by the IUCN SSC CPSG and BGCI (see **Table 2.2**).

Planning for multiple species as part of the same process is often advocated to reduce duplication and streamline consultation processes. Grouping criteria have been proposed by several authors (see **Box 2.9**). Despite the assumption of greater efficiency, studies to date have failed to find evidence of it (Cullen et al. 2005; Baptista et al. 2019). Multi-species planning can itself be complex, time-consuming and expensive and multi-species plans have been criticised for reflecting a poorer understanding of species-specific ecology and biology, having fewer actions implemented, for being revised less frequently and for being associated with poorer outcomes for species than single-species plans (Boersma et al. 2001; Clarke & Harvey, 2002; Lundquist et al. 2002; Heywood et al. 2018). Conversely, Moore & Wooller (2004) identified multi-species plans as those best able to address adaptive management, especially where there is little information about threats and their effects, and for facilitating the application of lessons learned for one species across others in the group. Baptista et al. (2019) found no difference between single- and multi-species plans in threat reduction effectiveness.

Despite this, single-species planning at the scale required has been shown to be too slow, and too expensive, and can result in duplicated effort (Kraus et al. 2021). Therefore, some form of multi-species planning will need to be part of the solution. Studies suggest its future lies in improving the methods used to group species to ensure closer overlaps in conservation need, and increasing the attention to individual species' requirements in these projects (NMFS-FWS 2010; NMFS 2020; Wiens et al. 2008). See **Box 2.9** for a summary of current advice.

Single-species planning continues to be advocated for: species with complex needs that do not align well with other species; those on a steeper trajectory of decline than their habitats or sympatric species; those reduced to very small or highly fragmented populations for which area- or threat-based planning will not drive fast enough or intensive enough action to avert irreversible damage; those whose cultural or economic significance can draw important interest or resources; and those for which lack of coordination of existing action, or conflicts among conservation agents are recognised barriers to progress (Foose et al. 1995; Moore & Wooller, 2004; Sanderson et al., 2002a; Lees et al. 2021).

Planning for “umbrella” or “surrogate” species, whose conservation needs coincide with those of many others can be viewed as a hybrid between single- and multi-species planning and studies show merit to this approach provided species are carefully chosen (Wiens et al. 2008; Branton & Richardson 2011; Ward et al. 2019).

Box 2.9 Summary of advice on multi- versus single species planning (from Foose et al. 1995; Baptista 2019; Clark and Harvey 2002; Brown et al. 1996; Burbidge 1996; Machado, 1997; Foin et al. 1998; Jewell 2000; Wiens et al. 2008).

- In selecting candidates for multi-species planning, consider:
    - species subjected to similar threats within a specified geographical area;
    - species with urgent management requirements that coincide (e.g. *ex situ* needs);
    - species reliant on protection or restoration of the site, or ecosystem.
- Multi-species approaches to planning and action may be better:
    - for stretching limited resources across more species
    - for understanding and addressing common threats;
    - for allowing lessons learned from one species to be applied to others in the same plan.
- Single-species planning may be better:
    - for species in small numbers and close to extinction;
    - for high-profile or politically charged species
    - for species with complex needs that do not overlap with those of other species;
    - for species where poor coordination or conflict among stakeholders is a recognised barrier to effective conservation.
- Carefully chosen "umbrella" species whose needs, if met, will concurrently meet the needs of many others may be a useful hybrid approach.

To get the most from multi-species planning projects (modified from NMFS-FWS 2010)

- Each species in the plan should be fully addressed in terms of status, threats, and biological needs and constraints (this does not mean that these items need be addressed for each species separately but that a reader should be able to discern each species' status, threats, etc., easily from the information provided).
- Objective, measurable recovery criteria must be developed for each species, although the same criteria can apply to more than one species where the threats are identical.
- Recovery actions should be consolidated for multiple species whenever possible, to maximise effectiveness, but should indicate which species will be affected.
- As part of a cycle of review and revision, plans can be updated for changes relevant to individual or subsets of species (i.e. where it will create delays, waiting to update all should not be considered essential).
- Prepare for the fact that in general, multiple-species plans will be more expansive documents, and means for keeping them updated and current will be more complex.

2.5.2.2 Low-intensity approaches

Due to statutory consultation and other administrative requirements, government recovery plans can take years to finalise. For example, though meant to take 2.5 years to complete, a USA study found plant plans take more than 4 years and animal plans more than 11 years (Malcolm & Li 2018). This can delay the implementation of urgent action. In some countries, interim documents have been developed either to bridge the gap or, where considered sufficient, to replace more onerous recovery planning. Examples are the Australian

"Conservation Advices", which describe key threats and priority local and regional conservation actions for listed species or ecosystems, and the USA Recovery Outlines, described as strategic documents used to direct the recovery effort and maintain recovery options for a species, group of species, or ecosystem, pending an approved recovery plan (Heywood et al. 2018; DCCEEW-Department of Climate Change Energy Environment & Water 2023).

In the 2020 iteration of its recovery planning guidance, the USA National Marine Fisheries Service recommends breaking up recovery plans into three separate parts which can each be developed, approved and updated separately, creating a nimbler system and a shorter path to action (NMFS 2020), see **Figure 2.3**.

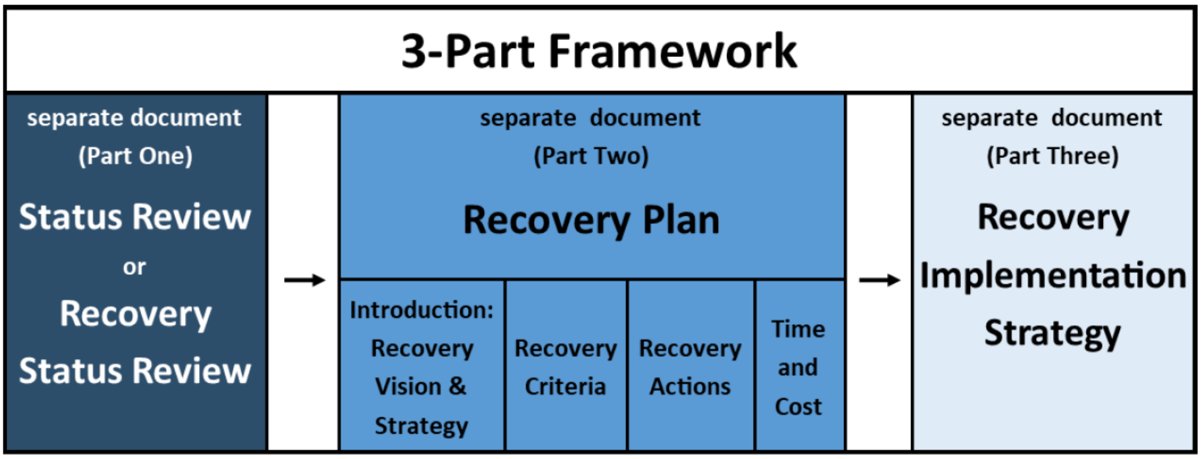


Figure 2.3. The optional separation of a USA National Marine Fisheries Service recovery plan into three separate documents to create a nimbler system of development and updates, and a faster route to action (from NMFS 2020).

Within the NGO sector, the IUCN SSC CPSG has run multi-stakeholder planning workshops in more than 65 countries at the invitation of governments. Reports from these workshops are generated rapidly (usually within 6-12 months) in the form of species conservation plans. In many cases these become official plans, but in countries with formal recovery planning frameworks they may operate for several years as interim guidance pending further rounds of consultation and amendment in-line with government requirements (Lees at al. 2021).

In New Zealand, the planning needs of many species are accommodated through spatially explicit species management "prescriptions" that describe and prioritise the action required for those species in defined spatial units managed by the government's Department of

Conservation. Detailed plans to restore and recover are reserved for a subset of species considered "iconic" or otherwise deserving of greater attention (New Zealand Department of Conservation 2023).

In some cases, effective planning coverage for a species may be achieved by formally connecting it to the plan of another species with the same requirements, or by embedding its needs within, for example, one or more protected area management plans, with associated performance indicators (NMFS 2020; Natural England 2022).

The IUCN SSC Conservation Planning Specialist Group applies a process called Assess-to-Plan (A2P) immediately following large IUCN Red List assessment projects. Assess-to-plan facilitators work with Red List assessors and local conservation actors to:

a) identify the broad conservation needs of each species categorised as threatened;
b) make recommendations for each species about either:
    i. grouping with others for planning,
    ii. housing under existing plans or initiatives, or
    iii. targeting for more intensive, single-species attention;
c) identify the stakeholders who could lead or collaborate on the recommended planning directions identified; and
d) agree the next steps towards mobilising these activities.

Assess-to-Plan aims to ensure all species identified as globally threatened move quickly from assessment, to planning and action. It is relatively new and has been used across a range of taxa and in several countries (e.g. Gibson et al. 2020; IUCN SSC HSG/CPSG 2022; IUCN/CPSG/CI/DENR-BMB 2022).

Given ongoing concerns about the ability to resource species conservation planning, more efficient, while still effective, planning approaches are needed. The "lighter" approaches described here are relatively recent and no studies were found evaluating their efficiency or effectiveness compared to longer-standing multi-and single-species approaches.

2.5.3 Prioritisation

There is no global standard for how species should be prioritised for planning. Formal assessments of rarity or vulnerability to extinction are recommended for signalling whether species are likely to be, or are already, falling through the gaps of biodiversity protection and management and this is generally used as a starting point (Rossi et al., 2016). The IUCN's

Global Red List Database is a well-used source of this information for organisations operating globally (such as the IUCN) and for countries with high levels of endemicity. However, as most planning and action takes place at the national or sub-national level, national assessments are often more relevant. As of 2023, at least 113 countries have national red lists, up from 76 in 2009 (Miller, 2009; ZSL and IUCN National Red List Working Group 2022).

The IUCN SSC CPSG advocates defining a pathway to planning and action for all globally threatened species (CPSG 2020). However, at the national level, once threatened species are identified (of which there can be thousands) there is generally a further layer of prioritisation. In Australia, Canada, the USA and UK a formal nomination process determines listing under the relevant legislation and, once listed, additional conservation measures are triggered, including planning (Kraus et al. 2021; BRIG 2007). Criteria used for this prioritisation step vary. In the USA the most important factors are the magnitude and immediacy of threats, and taxonomic distinctiveness (US Fish and Wildlife Service 2016). Listing decisions in the UK emphasise species of international importance, those in rapid decline and at high risk (BRIG 2007). For its 5000 or so threatened plant species (as an example), China has prioritised 120 for combined *in situ* and *ex situ* planning. Those selected fit the definition of Plant Species with Extremely Small Populations (PSESP) in numbering fewer than 5000 mature individuals in total and fewer than 500 individuals in each isolated population (Yang et al. 2020). In Europe, countries such as Italy have adopted mandated European-wide species priorities laid out in the EU Habitats Directive (a taxonomically biased and relatively inflexible list) for planning and action, rather than generate their own (Rossi et al. 2016). Canada also prioritises for planning those species considered at risk of becoming threatened in future and was the only country found with this provision (Creighton & Bennett, 2019). Likelihood of planned action being successful is also recommended (Mace et al. 2007; MAPISCo Project Team 2013).

Systemic biases create divides between the species that need targeted planning and those that receive it. In Australia, the UK and the USA, bird and mammal species are more likely to have recovery plans than herpetofauna and fish; and vertebrates are more likely to be planned for than invertebrates and plants (Tear et al. 1995; Metrick & Weitzman 1996; Laycock et al. 2009; Watson et al. 2011a). In Canada, arthropods and amphibians are less likely to have plans than other species, a situation that does not reflect assessed conservation status, with 56% of arthropods classified as endangered and only 7.9% with action plans

(Creighton & Bennet 2019). In addition, Canadian researchers report a bias against listing (and therefore against planning for and protecting) commercially valuable species, and those threatened by agriculture and residential and commercial development and advocate a more transparent distinction between science-based risk assessments and the politically influenced process that leads to listing decisions (Mooers et al. 2007; Findlay et al. 2009, McCune et al. 2013).

The species selected for red list assessments further exacerbates bias in planning coverage. To date, global assessments cover 74% of vertebrates, but only 2% of invertebrates, 12% of plants and 0.3% of fungi and protists (IUCN 2023a). National assessments follow similar patterns, with (for example) in New Zealand, only 35% of freshwater invertebrate species assessed compared to 100% of bird taxa (Drinan et al. 2020). These biases build on an even more fundamental one, through which societal preferences dictate that most species remain unknown or unstudied while relatively few dominate resources and interest (Troudet et al. 2017). **Box 2.10** summarises advice on common prioritisation pitfalls and how to avoid them.

BOX 2.10: Summary of advice on species prioritisation for targeted planning (from BRIG 2007; USFWS 2016 Mooers et al. 2007; Findlay et al. 2009, McCune et al. 2013; Rossi 2016; Mace et al. 2007; MAPISCo et al. 2013; US Fish and Wildlife Service 2016; Troudet et al. 2017; Drinan et al. 2020; Watson et al, 2011; CPSG 2020).

- Address taxonomic biases in the species prioritised for conservation status assessments and be taxonomically inclusive in selections for planning.
- As a minimum, aim to prescribe a pathway to conservation planning and action for all species considered vulnerable to extinction at the global level.
- In addition to extinction risk, consider factors such as global conservation significance, phylogenetic uniqueness, extremely small or fragmented populations, urgency, and umbrella species potential in determining criteria, as well as public preferences.
- Maintain a clear separation between science and politics in any prioritization process.
- Include consideration of likelihood of success.
- Wherever prioritisation is needed, ensure transparency.

2.5.4 Enabling conditions

This section looks at framework elements related to developing plans at scale, integrating or harmonising plans, and for enabling implementation.

2.5.4.1 Developing plans at scale

Studies indicate that many countries are struggling to deliver planning at the scale prescribed by their prioritisation systems. Studies for Canada, the USA and Brazil all report a shortfall of c. 25% (though note only animals are included Brazil's assessment). Further, many plans in these countries as well as in others such as Australia, are now old and need to be reviewed (Kraus et al. 2021; Vercillo 2023; DCCEEW 2022).

Other countries or organisations attempting to deliver planning at scale are likely to experience the same problems: many species needing urgent attention and limited ability to meet that urgency with the planning and implementation capacity available. As described above, in Australia, Brazil, Canada, the UK, as well as in the USA, practitioners are trialling multi-species planning, installing less labour-intensive "light" forms of plans and minimising red-tape, to create efficiencies that will lead to more plans being developed and implemented, more quickly.

To address current shortfalls in the USA, the US Fish and Wildlife Service (USFWS) has developed a set of national workplans for 2022-2025, reflecting a schedule for developing recovery plans and completing five-year reviews. These workplans aim to: provide greater clarity and predictability regarding the timing of recovery planning and species status reviews; strive for more timely completion of recovery plans and five-year reviews; and ultimately, expedite the implementation of those recovery plans with the highest likelihood of preventing extinction (US Fish and Wildlife Service 2023).

2.5.4.2 Plan integration

Within a given country there are likely to be many overlaps between species conservation plans in the kinds of action recommended, and additional overlaps between species conservation plans and ecosystem, protected area and systemic threat-abatement plans. In addition, for some species there will be plans for the coordinated storage or management of *ex situ* populations. Little information was found on how different countries or organisations keep track of and align these planned actions which, if not coordinated could lead to duplication, confusion and conflicting priorities, particularly where different groups of conservation actors and different sources of funding are involved.

Some of this streamlining can be done at the point of planning for individual species. For example, by ensuring that at the start of every new planning effort there is a scan for existing plans and initiatives relevant to the focal species and ensuring that the right connections are made to the relevant projects from the outset. This idea is encapsulated in the IUCN SSC's One Plan Approach which aims to engage or activate all relevant stakeholders and resources at the outset of planning (Byers et al. 2013). An example of an output from this is illustrated in **Figure 2.4** for the Tamaraw, *Bubalus mindorensis.*

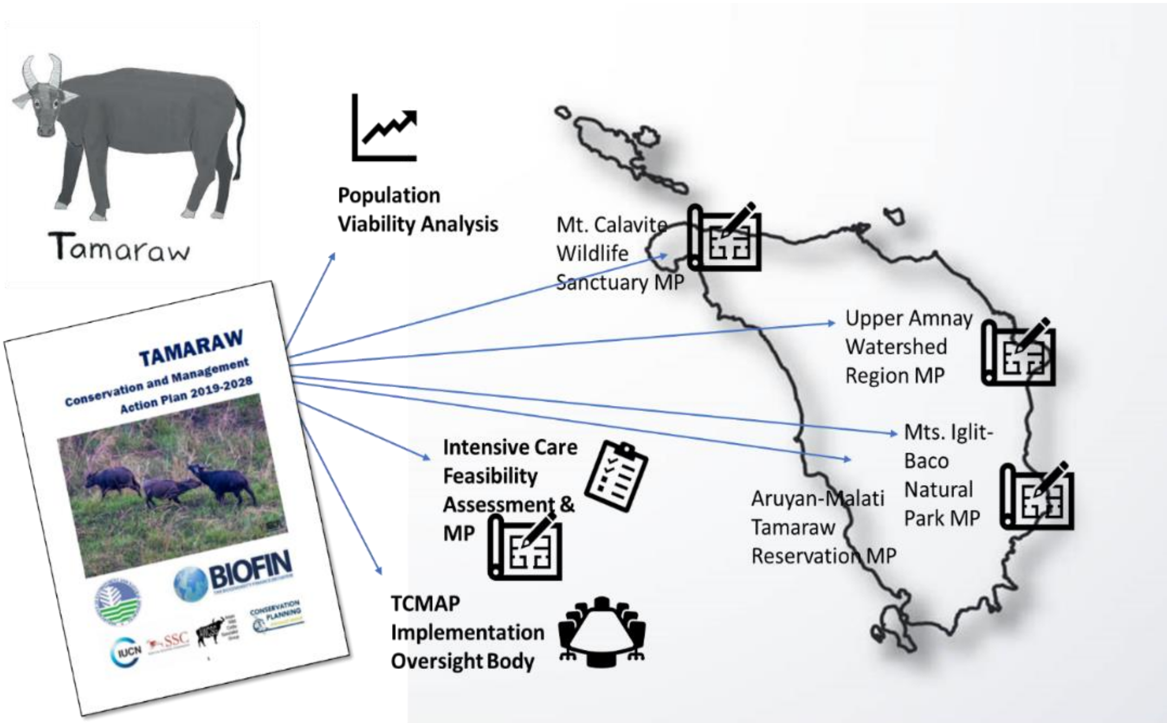


Figure 2.4. Illustrates the streamlining of multiple management plans (MP) relevant to the single-species conservation action plan for Tamaraw, *Bubalus mindorensis*, which is now assumed to occur at only four sites on the island of Mindoro in the Philippines. The Tamaraw Conservation Program has its own plan and associated staff who coordinate and drive action for the species. Stakeholders representing all four wild sites, as well as the *ex situ* community, took part in the development of the Tamaraw Conservation and Management Action Plan (TCMAP). Relevant recommended actions within it are now mirrored in area-based management plans for two of the four wild sites. Integration into the remaining two, as well as the *ex situ* intensive care feasibility study, are underway (DENR 2019).

Current USA National Marine Fisheries Service (NMFS) guidance recommends that if a newly listed species occupies the same habitat and has similar recovery needs as another species or group of species with an existing plan (either draft or approved), the newly listed species may be incorporated into that plan by adding in the new species' goals, actions, and time and cost estimates (NMFS 2020). Similarly, in the UK a recent review advocates embedding threatened species conservation into existing mechanisms wherever possible. For example, integrating species recovery actions into green infrastructure and natural area development plans, along with species-related performance indicators, to bring better outcomes for species for the investment provided (Natural England 2022).

A scalable system has been reported from New Zealand, where the Department of Conservation has developed a database of spatially explicit prescriptions for > 700 threatened species, overlaid on ecosystem management prescriptions for those same areas, mainly for public lands. Benefits of the system, which includes a prioritisation feature, are that it connects both local and national perspectives on what needs to be done, where, how much it will cost, and how limited public as well as private and corporate resources can be directed, to deliver the most benefit to the most species and ecosystems of concern (Joseph et al. 2008; 2009; 2011; Bennett et al. 2015).

These proposed and developing approaches are relatively recent and have yet to be evaluated for impact on species status.

2.5.4.3 Plan implementation

Information and advice on developing individual plans to maximise likelihood of implementation is covered in a previous section. This section looks at the steps national governments and NGOs are taking or are being encouraged to take, to create an enabling environment for successful implementation.

Governance and administration

Good coordination and administrative structures can improve implementation results (Laycock 2009; Crees 2016). Authors caution against a one-size-fits all approach and recommend tailoring these elements to the needs of the plan (NMFS 2020). Detailed advice is available on what to consider, on the potential pros and cons of establishing a dedicated and diverse recovery team, and on good practice recovery team governance (NMFS 2020; (DCCEEW Water 2017).

Public awareness

The way which nature is perceived and appreciated will determines to a large extent what kind of nature and how much of it will be the object of conservation efforts (Rientjes, 2002). Guidance and reviews from different countries emphasise the importance of raising public awareness of the importance of lesser-known species, of recovery plans, and of the role that wider society can play in ensuring their effective delivery (Troudet et al. 2017; Towns et al. 2019; NMFS 2020). Widening participation, ownership and support is one of five major themes of a recent review of species recovery in the UK, where it is recognised that with each generation the scale of past biodiversity losses is less well understood, at least through direct contact, and there is a real risk that loss is becoming either normalised as part of the human experience or is going unrecognised as a symptom of 'shifting baseline syndrome' (Natural England 2022).

Empowering Indigenous Peoples

Indigenous Peoples are critical partners in biodiversity conservation because of the knowledge they hold and their tenure rights over, and relationships to, over a quarter of the world's land surface (Garnett et al. 2018). Failures to recognize Indigenous Knowledge fully and to seek Indigenous cooperation lead to missed opportunities for partnerships important to implementing conservation (Cisternas 2019; Turcotte et al. 2021). A recent review from Canada showed that despite a legal requirement to consult Indigenous people to the extent possible, 52% of plans indicate no Indigenous involvement (Hill et al. 2019).

Ensuring that cross-cultural engagement is respectful, reciprocal and meaningful requires that appropriate principles, responsibilities and protections are in place from the outset. Promising examples are reported that are showing benefits to people as well as biodiversity (e.g. Moorcroft et al. 2012; Jasmine et al. 2016; Godden & Cowell 2016; Cisternas 2019; McAllister et al. 2023), however, there remain many challenges to getting this right and systemic support is needed (Wehi et al. 2019).

Harmonising government sectors

Conflicting imperatives or priorities within government can be a significant obstacle to implementation of recovery plans. Towns et al. (2019) describe the history of contradictory land and biodiversity management activities in New Zealand due to conflicting mandates and priorities of the three main government agencies responsible. The differences have been gradually harmonised since they were brought together under a single Department of Conservation in the 1980s.

Resourcing

Ensuring sufficient resources both for planning and for implementation is a recurring recommendation. For decades in Australia, Brazil, Canada, New Zealand, the USA and UK, studies have identified inadequate resourcing as a major obstacle to achieving recovery progress (Male 1995; Metrick & Weitzman 1996; Male & Bean 2005; Laycock et al. 2009; 2011; Seabrook-Davison et al. 2010; Watson 2011a; Wintle et al. 2019; Baptista et al. 2019). Wintle and colleagues report current expenditure in Australia is only 15% of what is needed to halt extinctions and recover threatened species. Further, studies have shown that not only are just a fraction of species' needs funded, but resource distribution is also skewed towards few species (Male and Bean 2005; Laycock et al. 2012). For example, in a study of 38 UK plans, Laycock and colleagues showed that 80% of expenditure was on the top five most expensive plans.

Though currently insufficient, formal funding mechanisms were found to be in place in several countries (Rossi et al. 2016; Wintle et al. 2019; Kraus et al. 2021; UK Government 2023). The sums required are large. Kraus et al. (2021) report annual spending on endangered species of 60 million, 92 million and 1.478 billion US dollars for Canada, Australia and USA respectively. As government funding is unlikely ever to be adequate (NMFS 2020), greater innovation in this area is recommended including partnerships with the private sector (e.g. Bennett et al. 2015).

Box 2.11 summarises available advice on supporting plan development, integration and implementation at scale.

Box 2.11. Advice on supporting plan development, integration and implementation at scale. (From NMFS 2020; Kraus et al. 2021; Natural England 2022; Towns et al. 2019; Laycock et al. 2012; Turcotte et al. 2021; Wehi 2019).

- Formalise a schedule for development of plans and communicate it in advance to stakeholders.
- Establish low-intensity planning approaches and measures that reduce red tape to create faster pathways to implementation wherever this can meet species' needs.
- Formalise measures to connect and integrate plans to create efficiencies and reduce duplication.
- Develop and promote good practice in leading, coordinating and communicating plan implementation.
- Invest in raising public awareness of the importance of lesser known species, of recovery plans, and of the role that wider society plays in implementing them.
- Advance the key role of Indigenous Peoples as conservation partners by providing systemic support to promote ongoing respectful, reciprocal and meaningful engagement.
- Include all relevant stakeholders in planning conversations.
- Harmonise government policies and inter-departmental priorities to smooth implementation.
- Prioritise available resources based on expected return on investment and work innovatively to open up new streams of resourcing including the private sector.

2.5.5 Program-wide tracking and evaluation of plans and their impact

Reviews of plans at the individual project scale ask questions about (for example) whether specific actions are being implemented, whether they are having the expected effect and, if not, what needs to be done to improve progress. Reviews across multiple plans enable an evidence-based approach to answering questions about what is and isn't working in general, and what program-wide changes could be made, or what additional guidance could be provided, to improve overall results. Most of the information and advice presented in this chapter is drawn from these reviews. They are also the basis for Chapter 3., which deals with measuring the impact of plans on species conservation status.

To support them, as well as general accountability reporting for larger planning frameworks, some countries and organisations have established lists or databases of plans produced. Publicly accessible ones are shown in **Table 2.2** though as plans are clearly being generated in many countries there are likely to be others. The publicly displayed information varies from a basic record of the existence of a plan, to detailed information on the actions prescribed for each species and their implementation status. Several allow plans to be downloaded.

Centralised database systems including this and related information on implementation and resources spent are advocated in several studies (e.g. Laycock et al 2009; Cullen et al. 2005).

Table 2.2. Information on the numbers of species covered by plans for countries and NGOs with publicly available records of this. (Botanic Gardens Conservation International (BGCI); International Union for Conservation of Nature Species Survival Commission (IUCN SSC).

Lead agency	Scope (state, national, regional)	Reported # species covered (in # plans)	Ref. Date	Sources
Government				
Australia	National	737 spp.	2023	Department of the Environment (2023). http://www.environment.gov.au/sprat.
Brazil	National	643 spp. (49 plans)	2017	Baptista et al. 2019
Canada	National	345 spp. with strategies	2013	Turcotte et al. 2021. https://species-registry.canada.ca
China	National	120 (1 plan, plants only)	2020	Yang et al. 2020
European Union	Regional	56 spp.(56 plans)	2018	https://ec.europa.eu/environment/nature/conservation/species/action_plans
New Zealand	National	73 spp.	2021	https://www.doc.govt.nz/about-us/science-publications/series/threatened-species-recovery-plans. Note also, prescribed actions identified for 700 taxa (Joseph et al. 2008)
United Kingdom	England, N. Ireland, Scotland, Wales	1150 spp.	2009	Note: since 2012, responsibility devolved to separate administrations for England, N. Ireland, Scotland and Wales. No longer a single UK list (Ruddock et al. 2007; Eaton et al. 2015)
	Scotland	32	2016	Gaywood et al. 2016
USA	National	1161 spp.	2020	Malcolm & Li (2018). https://ecos.fws.gov/ecp/report/species-with-recovery-plans
Non-government				
IUCN SSC CPSG	65 countries	> 500 spp. (322 plans)s	2023	http://cpsg.org
BGCI	2 countries	119 spp. (2 plans)	2023	Harvey-Brown & Shaw (2020); Harvey-Brown (2023).

2.5.6 OVERALL PLANNING FRAMEWORK REQUIREMENTS

The multi-plan reviews, recovery program histories and other information consulted converge on several basic elements of an effective planning framework able to operate on a large scale (see **Box 2.12**)

Box 2.12 Basic elements of an effective planning framework able to operate on a large scale (from Mooers et al. 2007; Kraus et al. 2021; Roberts & Hamann 2016; Towns et al, 2019; NMFS 2020; Natural England 2022)

- A standard, unbiased method to prioritise species for conservation planning attention.
- Guidance on: how to develop plans for the local context; planning options that can meet different species' needs efficiently; and on stakeholder engagement.
- An integration mechanism through which prescribed actions for species conservation can be integrated into or combined with, related activities for other species, habitats, ecosystems, landscapes or seascapes, and threats.
- Effective coordinating bodies to implement, monitor, adapt and advocate for plans.
- An adequate funding mechanism for plan development and implementation.
- Harmonised government policies and an aware, supportive, and engaged society.
- A central, curated and current record of what plans exist and their status;
- Standard reporting across plans to enable system-wide review and adaptation.
- A schedule for plan development communicated to stakeholders.

2.6 SUMMARY

In this Chapter I explored whether the large gap between the number of species that need planning and those that receive it could be due to lack of information about how to plan effectively for species, or to the inability of national or global planning frameworks to cope with the challenge.

I found that over the past few decades countries have developed national frameworks to support the generation of large numbers of species conservation plans. There is good agreement between countries on the challenges and essential components of such frameworks and regular evidence-based reviews support learning and adaptation. However, these frameworks and the planning projects that they enable remain chronically under-resourced and exist in only a few places around the world.

In addition, over time, lessons have been learned about what to include in plans and how to formulate this content, resulting in much convergence across planning guidelines from both government and non-government sectors. In particular, the integration of tools such as

Population Viability Analysis and the participation of stakeholders, is emphasized. Though clearly recognized as important, factoring these elements into planning adds further cost and complexity. Scaling-up this style of planning to prevent extinctions and drive recovery will require much greater investment, and prospective funders will need to be assured of a return on that investment.

Therefore:

- In Chapter 3, I evaluate a set of planning projects from a database held by the IUCN SSC Conservation Planning Specialist Group, which included both population viability analyses and the participation of stakeholders, to discern any measurable changes in downstream species conservation status that could be attributed to planning.
- In Chapters 4 and 5, I explore in more detail how and under what circumstances population viability analyses bring benefits to species conservation planning, using the New Zealand kākāpō, *Strigops habroptilus*, and Tasmanian devil, *Sarcophilus harrisii*, as case studies.
- In Chapter 5, using the Tasmanian devil example, I describe the integration of population viability analysis into a participatory stakeholder workshop setting and report on the resulting short-term conservation outcomes.

CHAPTER 3. SCIENCE-BASED, STAKEHOLDER-INCLUSIVE AND PARTICIPATORY CONSERVATION PLANNING HELPS REVERSE THE DECLINE OF THREATENED SPECIES

3.1 ABSTRACT

Reversing the decline of threatened species is a target for the Convention on Biological Diversity but current efforts are failing. An integrative, multi-stakeholder approach to species conservation planning, which includes population viability analyses and both *in situ* and *ex situ* management consideration, could improve outcomes for some of the most challenging cases. The IUCN Species Survival Commission (SSC) uses such a planning approach, however, evidence of improved outcomes for species has to date been anecdotal. To assess the impact of planning, we accessed 35 species conservation plans completed in 23 countries over 13 years from the IUCN SSC database and matched them with independently generated Red List assessments of extinction risk. We used the Red List Index and a counterfactual approach, comparing the overall predicted extinction trend without planning with the observed trend after planning. Post-planning, threatened species declines continued, but gradually slowed, and then reversed, with an upward trend of recovery within 15 years. No species became extinct. Simulated counterfactual projections indicated outcomes would have been worse without the planning intervention; around eight species would have become extinct over that timeframe. To date, this planning approach has been applied to relatively high-profile species facing multiple threats, and where conflicting views, uncertainty, or lack of coordination among stakeholders constrain action. Opportunities to broaden application to other taxa are discussed. Our study provides evidence that science-based, participatory approaches to planning can create a turning point for threatened species by supporting stakeholders to transition quickly to more effective ways of working together.

3.2 INTRODUCTION

Aichi Target 12 of the 2011-2020 Convention on Biological Diversity (CBD) calls on countries to prevent extinction and ensure sustained improvement in the conservation status of known threatened species (CBD 2010b). Despite this, reviews show little progress on slowing declines (WWF 2020; IPBES 2019), the IUCN Red List currently reports 37 480 threatened species (IUCN 2021), and future extinctions are predicted (Monroe et al. 2019).

Species conservation planning is one of a range of measures advocated to reverse extinction trends (Mace et al. 2018). Species conservation planning should aim to increase the effectiveness of conservation action, by ensuring that it is based on (*i*) relevant information for the species, (*ii*) well-defined goals, (*iii*) multiple perspectives, and (*iv*) agreement among those involved about what should be done (Boersma et al. 2001). Such planning, which ideally combines both social and analytical elements (Sande et al. 2005; Groves & Game 2016), takes time and resources and is currently applied to few of the species that need it (e.g. Brazill-Boast et al. 2018; Watson et al. 2011a). While recent studies provide compelling evidence that conservation action improves species status (Hoffmann et al. 2015; Butchart et al. 2006; Young et al. 2014), the way in which such successful action was planned, and whether planning supported outcomes, is rarely considered.

Evaluating the impact of planning on species is difficult, resulting in few attempts and conflicting conclusions. Although studies report that planning led to improved status of endangered species in the USA (Schultz & Gerber 2002; Taylor et al. 2005), a further study showed it to be detrimental if not combined with substantial government funds (Ferraro et al. 2007) and an Australian study showed no effect once biases associated with prioritising species for planning were removed (Bottrill et al. 2011). The challenges of evaluating the impact of planning include insufficient data, protracted implementation time of plans, the potentially long timescale over which species might be expected to show signs of recovery and the difficulty of disentangling planning effects from those of other influences (Bottrill & Pressey 2012; Watson et al. 2011a). Furthermore, attempts to overcome the latter by comparing taxa with plans, to those without, require strong assumptions about equivalence that are often confounded by variables such as phylogeny and geography (Fuller et al. 2003).

Finally, differences in purpose and approach complicate treatment of “planning” as a single type of intervention across multiple projects.

The Conservation Planning Specialist Group (CPSG) of the IUCN Species Survival Commission (IUCN SSC) supports diverse groups to develop species conservation plans collaboratively. Depending on project circumstances and emphases, the planning approach used is referred to variously as the “Population and Habitat Viability Assessment” (Miller & Lacy 2003) or as the “One Plan Approach” (Byers et al. 2013; Conde et al. 2015), but its underlying principles, key elements and format are consistent (CPSG 2020). Planning workshops are initiated and organised by government or non-government agencies in countries within the species’ range. Wherever possible, all stakeholders are assembled (typically 20-60) for 3-4 days of facilitated analysis and discussion. Alongside government agencies, local communities, and academia, both in situ and ex situ species conservation communities are represented and decision-making is supported by population viability analyses. Stakeholders participate actively in decision-making, proceeding by consensus to agree a definition of successful species recovery or conservation, to analyse challenges to this, recommend solutions and commit to action. Outcomes are documented within 6-12 months (see supplementary material for further details). Though the planning tools and elements described are in use across the wider species conservation community, as far as we are aware the IUCN SSC CPSG approach is the only one that routinely integrates all these features within a standard workshop format. The approach is a good candidate for evaluation as the long period over which it has been used (> 30 years), the relative stability of style and format, and the ready availability of information on planning projects, reduce some of the difficulties commonly encountered when assessing the impact of planning.

Past attempts to evaluate the impact of this approach have involved pre- and post-workshop surveys of participating stakeholders, to see how their work is affected by the planning deliberations and outputs (Vredenburg and Westley, 2003). Results indicate positive outcomes for participants, but to date no systematic studies have considered whether this is matched by an improvement in overall species conservation status. Given the effort and resources involved in this style of planning, evidence of impact would be useful to decision-makers charged with determining whether and how species planning is done. We therefore set out to fill this gap.

To assess the impact of this specific approach, we used a publicly available database of more than 250 well-documented species conservation planning projects, maintained by the IUCN SSC CPSG (http://cpsg.org/document-repository). Plans date from 1990 onwards and span more than 70 countries. To estimate progress on slowing or halting species extinctions following planning, we utilised the Red List Index (RLI) (Butchart et al. 2004; 2007; Mace et al. 2018). The RLI is calculated from the IUCN's published threat categories for individual species, which are generated by expert assessments of those species against independent criteria, with quantitative thresholds of extinction risk designed to be transparent and consistent across taxa (Mace et al. 2008). The RLI is widely used, readily interpreted by a range of audiences and has been adopted by the CBD for reporting on global species targets (IUCN 2021).

Impact evaluation assesses the degree to which changes in outcome can be attributed to an intervention rather than to other factors, which requires knowing what outcomes would have looked like in absence of the intervention (Ferraro, 2009). In other studies, the necessary counterfactual comparison has been provided by econometric matching of species with plans, to those without them (Ferraro et al. 2007; Bottrill et al. 2011), or by eliciting the judgement of experts to estimate the counterfactual trajectories of species in absence of specific programs of conservation management (Butchart et al. 2006; Hoffmann et al. 2015; Young et al. 2014). Neither of these methods were available to us due to the wide geographic distribution of projects in the database, the long timeframe over which planning projects took place and the disproportionate number of highly threatened, high-profile, and phylogenetically distinct taxa included. As a result of these factors, no set of species without plans met the equivalence requirements of a control, and no group of experts available to us could provide informed counterfactual judgements across all projects. Instead, we used observed patterns in extinction trend *before* planning (though in the presence of conservation actions), to simulate a counterfactual extinction trend for the group *without* planning. We then compared the simulated *without* planning trends, to the observed *with* planning trends, to estimate the overall impact of the planning intervention on the species' conservation status (see **Figure 3.1**).

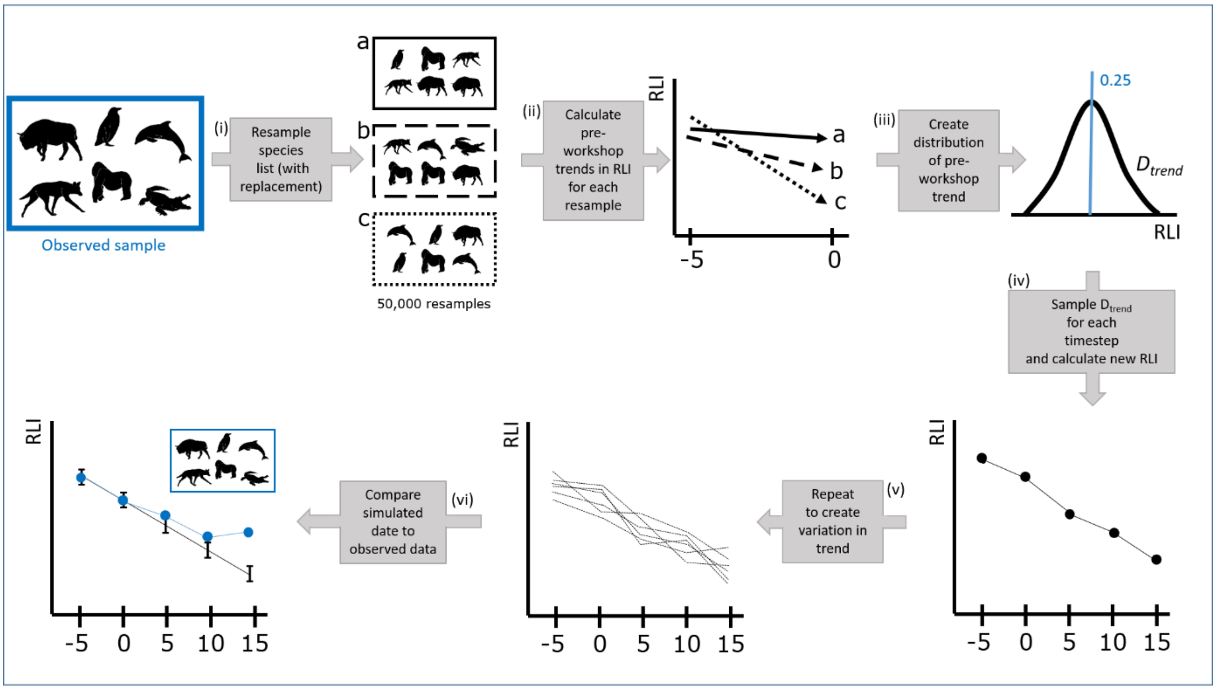


Figure 3.1. Conceptual illustration of the planning impact evaluation method used in this study. We used a counterfactual approach, comparing the predicted extinction trend for species without planning with the observed trend after planning. Red List Index (RLI) uses IUCN Red List categories to measure the projected overall extinction risk over time. D_{trend} is the distribution of simulated trends in threat status over the pre-workshop period.

This is the first use of the globally recognised Red List Index to evaluate the impact of a specific planning approach. Our work is relevant to those engaged in reversing the decline of threatened species and to planning practitioners seeking to evaluate longer-term impacts across multiple projects and taxa.

3.3 Methods

3.3.1 Building the dataset

We accessed species planning projects from the IUCN SSC CPSG database and, where possible, matched them with the IUCN Red List (RL) assessments for those species over time, to assess the impact of planning on conservation outcomes. IUCN Red List (RL) assessors assign species to one of seven RL extinction risk categories: Data Deficient (DD), Least Concern (LC), Near Threatened (NT), Vulnerable (VU), Endangered (EN), Critically Endangered (CR) and Extinct (EX). Assessments are repeated periodically (at least every 10 years for mammals and birds). To compare trends in population decline before and after planning, we selected projects for which the species involved had assessments extending either side of the planning workshop year. Projects were only included if the taxon had been assessed for the RL at least 5 years before, and at least 10 years after, the planning workshop. This asymmetry was considered reasonable because while a deterioration in species status can trigger an immediate elevation in risk category, 5 years of observed improvements are required to lower a risk category. Taxa also needed to have been assessed for the RL within 2 years of the planning workshop or to have identical categories before and after (to increase confidence in workshop-year category). Projects meeting these criteria were relatively rare in the database. Of 192 projects that were carried out before 2009 (thus allowing for at least 10 years of post-planning data): nine were excluded due to missing project information, two because they were area-based (and not species-based), and one because it was aimed at managing a feral species. Of the remaining 180 projects, 23 were either updates to a previous planning workshop or part of a workshop series and so were excluded on that basis. Thirty-six projects were for sub-populations of species and 24 for subspecies and all but one of these (*Gorilla b. beringei*) had not been assessed for the Red List, which is predominantly directed at species. A further six projects were for plant species that had also not been assessed for the Red List. Of the remaining 92 projects, one was excluded because it had a pre-existing conservation plan. Forty-six of the remaining projects were included in the study and the other 45 were excluded either because there were too few pre- or post-workshop RL assessments, because there were no assessments within 2 years of the workshop or, in a few cases, because the history of their assessments was interrupted by a 1990s change in RL categories that rendered some older categories with no direct equivalent. Categories were considered current until superseded by a reassessment. For each taxon we recorded the RL category: once 5 years before the planning workshop; once at the time of the workshop; once each at 5 and 10 years following the workshop, and (data permitting) at 15 years following the workshop. The 46 taxa meeting the criteria included projects with workshops held between 1990 and 2008, in 23 countries; of these 35 had data up to 15 years post-planning. The list of 46 species included 33 mammals, nine birds, two reptiles, one amphibian and one fish. Five of the species were categorised as Critically Endangered at the time of the workshop, 30 as Endangered, four were Near Threatened or Lower Risk/Near Threatened (a pre-1995 iteration of the RL category designations, directly equivalent to NT in the current system), and seven were Vulnerable (**Table 3.1, Table S3.1**).

3.3.2 THE RED LIST INDEX CALCULATION

The Red List Index (RLI) uses IUCN RL categories to measure the projected overall extinction risk of a set of species over time (Butchart et al. 2004). IUCN RL Categories are weighted according to their extinction risk, ranging from $W_{LC} = 0$ for Least Concern species to $W_{EX} = 5$ for Extinct ones. The RLI reflects the proportion of species in each category and is defined by Butchart et al. (2007) as:

$$RLI_t = 1 - \frac{\sum_s W_{c(t,s)}}{W_{EX} N}$$

where $\sum_s W_{c(t,s)}$ is the sum of weights (W_c) for all assessed species (s) at a given time (t), N is the total number of assessed species and W_{EX} is the weight assigned to extinct species (i.e. = 5). For the subset of 35 species for which 15 years of post-planning data were available, we calculated five observed RLI values, ranging from 5 years before the workshop to 15 years after ($RLI_{-5,obs}$ to $RLI_{15,obs}$). In addition, for the group of 46 species with at least 10 years of post-planning data, we calculated four observed RLI values ($RLI_{-5,obs}$ to $RLI_{10,obs}$). A RL index near 1 indicates that most species in the group are Least Concern (i.e. not threatened), while a RL index near 0 indicates that most species are Critically Endangered, Extinct in the Wild or Extinct.

3.3.3 STATISTICAL ANALYSES

To evaluate the impact of planning, we developed a counterfactual prediction of group extinction trend without planning. To do so, we extrapolated the observed pre-workshop trend between RLI_{-5} and RLI_0 over the post-workshop period and compared it to the observed post-workshop trend. We used a multi-step procedure to project outcomes without planning (See **Figure 3.1**). First, because our dataset only represents a *sample* of the global population of endangered species, we could only estimate the trend of RLI before the workshop, but no variability around this estimate, a crucial element to build the projection. To estimate the potential variation around the pre-workshop RLI trend, we used a classic bootstrap procedure (steps *i* to *iii*) (Efron 1979). More precisely, we (*i*) resampled our dataset with replacement 50,000 times (some species can appear several times). For each resample, (*ii*) we calculated the number of changes in threat status (*n*) between years -5 and 0 (e.g. *n* = 8 changes) and from there, the overall trend in threat status as *n* divided by the total number of species (e.g. trend = 8/35 = 0.23). By combining all trends in threat status from the 50,000 bootstraps, we (iii) generated a distribution D_{trend} of simulated trends in threat status over the pre-workshop period and therefore captured variability around the observed trend of RLI before the workshop (mean= 0.25, s.d.= 0.07; i.e. one quarter of species are expected to decline by 1 category every 5 years, if pre-workshop extinction rates continue). Using these estimates, we were able to propagate this trend and the associated variation through the 20-year period for the n=35 subset (steps *iv* to *v*). First (*iv*), we sampled a trend from the distribution D_{trend} (e.g. trend = 0.25) and randomly applied it to the observed dataset at year -5, thereby generating a simulated dataset of threat status at year 0. For example, for a trend of 0.23, the threat status of 23% of the species, selected randomly, was increased by one RL category between years -5 and 0. From this simulated year 0 dataset (i.e. workshop year), we generated a year 5 dataset of threat status, by applying a new trend sampled from D_{trend}. The same operation was repeated to generate simulated datasets for years 10 and 15. Species reaching the maximum category of 5 (Extinct) were removed from the sampling pool. (*v*) The entire process was repeated 50,000 times to create variation in the propagation of the extinction trend past t=0. For every simulated dataset RLI values were calculated for each time-step, to produce distributions of simulated $RLI_{t,sim}$ for the 35 species. Finally (*vi*), to assess the likelihood that the observed results arose by chance, we compared $RLI_{t,obs}$ (estimated from the observed data) to the simulated distribution of $RLI_{t,sim}$ at each time point. The observed data was considered significantly different from the simulated data when the observed RLI ($RLI_{t,obs}$) was larger than the 95th quantile of the simulated distribution ($RLI_{t,sim}$). P-values were calculated as the proportion of $RLI_{t,sim}$ values that were greater than $RLI_{t,obs}$. Using the simulated dataset, (*vii*) we also calculated the proportion of species that reached Category 5 (Extinct) after 15 years, to provide an estimate of average extinction risk without planning. Steps (*i*) to (*vii*) were repeated between years -5 and 10 for the complete dataset (46 species).

All analyses were performed using the R software (R Development Core Team, v. 3.2.0).

Table 3.1. Summary of the characteristics of projects included in the study: Wkshop Year = year in which the planning workshop was held; RL period = dates of first and last published Red List assessments (as of December 2019); Country = country in which the workshop was held; # People = number of participants listed as having attended part or all of the workshop; (orgs) = number of different organisations represented by participants (note that there may be some errors in this as some participants represented several institutions); PVA?/Ex situ recs? = presence (Y) or absence (N) of either PVA analyses or recommendations regarding ex situ management; -5 = Red List assessment category 5 years before the workshop; 0, 5, 10, 15 = Red List category at the year of the workshop and 5, 10, and 15 years after it; Least Concern (LC), Near Threatened (NT), Vulnerable (VU), Endangered (EN), Critically Endangered (CR) and Extinct (EX). LR/NT refers to Lowered Risk/Near Threatened and is equivalent to NT. All projects are housed on the CPSG website: (www.cpsg.org/document-repository).

Species Common Name	Scientific Name	Wkshop Year	RL period	Country	# People (orgs)	PVA?	Ex situ recs?	-5	0	5	10	15
Golden Lion Tamarin	*Leontopithecus rosalia*	1990	1982-2008	Brazil	46(32)	Y	Y	EN	EN	EN	CR	EN
Golden-headed Lion Tamarin	*Leontopithecus chrysomelas*	1990	1982-2003	Brazil	46(32)	Y	Y	EN	EN	EN	EN	EN
Black Lion Tamarin	*Leontopithecus chrysopypgus*	1990	1982-2003	Brazil	46(32)	Y	Y	EN	EN	EN	CR	CR
Black-Footed Ferret	*Mustela nigripes*	1992	1965-2015	USA	?	Y	Y	EN	EN	EX	EN	EN
Cotton-top Tamarin	*Saguinus oedipus*	1992	1982-2008	Colombia	?	Y	Y	EN	EN	EN	EN	EN
Bornean Orangutan	*Pongo pygmaeus*	1993	1965-2016	Indonesia	40(23)	Y	Y	EN	EN	VU	EN	EN
Baiji Dolphin	*Lipotes vexillifer*	1993	1986-2008	China	43(18)	Y	Y	EN	EN	CR	CR	CR
Lion Tailed Macaque	*Macaca silenus*	1993	1986-2008	India	93(51)	Y	Y	EN	EN	EN	EN	EN
Sumatran Rhino	*Dicerorhinus sumatrensis*	1993	1986-2008	Indonesia	59(47)	Y	Y	EN	EN	CR	CR	CR
Indian Rhino	*Rhinoceros unicornis*	1993	1965-2008	India	68(60)	Y	Y	EN	EN	EN	EN	VU
Javan Gibbon	*Hylobates moloch*	1994	1986-2008	Indonesia	55(35)	Y	Y	EN	EN	CR	CR	EN
Houston Toad	*Anaxyrus houstonensis*	1994	1986-2004	USA	50(29)	Y	Y	EN	EN	EN	EN	
Marsh Deer	*Blastocerus dichotomus*	1994	1982-2016	Brazil	35(24)	Y	Y	VU	VU	VU	VU	VU
Baird's Tapir	*Tapirus bairdii*	1994	1965-2016	Panama	23(17)	Y	Y	VU	VU	VU	EN	EN
Gharial	*Gavialis gangeticus*	1995	1982-2017	India	48(31)	Y	Y	EN	EN	EN	EN	CR
European Bison	*Bison bonasus*	1995	1965-2008	Poland	29(26)	Y	Y	VU	EN	EN	EN	
Barasingha	*Rucervus duvaucelii*	1995	1986-2013	India	61(27)	Y	Y	EN	EN	VU	VU	VU
Orinoco crocodile	*Crocodylus intermedius*	1996	1986-2017	Venezuela	27(23)	Y	Y	EN	CR	CR	CR	CR
Babirusa	*Babyrousa babyrussa*	1996	1986-2008	Indonesia	37-62(?)	Y	Y	VU	VU	VU	VU	
Tamaraw	*Bubalus mindorensis*	1996	1965-2014	Philippines	37(?)	Y	Y	EN	EN	CR	CR	CR

Species Common Name	Scientific Name	Wkshop Year	RL period	Country	# People (orgs)	PVA?	Ex situ recs?	-5	0	5	10	15
Lowland Anoa	*Bubalus depressicornis*	1996	1965-2014	Indonesia	37-62(?)	Y	Y	EN	EN	EN	EN	EN
Mountain Anoa	*Bubalus quarlesi*	1996	1965-2014	Indonesia	37-62(?)	Y	Y	EN	EN	EN	EN	EN
Mountain Gorilla	*Gorilla b. beringei*	1997	1965-2018	Uganda	68(44)	Y	N	EN	CR	CR	CR	CR
Iberian Lynx	*Lynx pardinus*	1998	1965-2015	Spain	52(32)	Y	Y	EN	EN	CR	CR	CR
Muriqui	*Brachyteles arachnoides*	1998	1982-2016	Brazil	27(21)	Y	Y	EN	EN	EN	EN	EN
Goodfellow's Tree-kangaroo	*Dendrolagus goodfellowi*	1998	1982-2016	PNG	47(35)	Y	Y	VU	EN	EN	EN	EN
Doria's Tree-kangaroo	*Dendrolagus dorianus*	1998	1982-2016	PNG	47(35)	Y	Y	VU	VU	VU	VU	VU
Humboldt Penguin	*Spheniscus humboldti*	1998	1988-2018	Chile	31(23)	Y	Y	LR/NT	VU	VU	VU	VU
Red Wolf	*Canis rufus*	1999	1982-2018	USA	43(25)	Y	Y	EN	CR	CR	CR	CR
African Penguin	*Spheniscus demersus*	1999	1988-2018	S. Africa	35(18)	Y	Y	LR/NT	LR/NT	VU	VU	EN
Ethiopian Wolf	*Canis simensis*	1999	1986-2011	Ethiopia	68(44)	N	N	EN	CR	EN	EN	EN
Arabian Tahr	*Arabitragus jayakari*	2000	1965-2018	UAE	50(29)	Y	Y	VU	EN	EN	EN	EN
Riverine Rabbit	*Bunolagus monticularis*	2000	1986-2016	S. Africa	21(17)	Y	Y	EN	EN	CR	CR	CR
Magellanic Penguin	*Spheniscus magellanicus*	2000	1988 2016	Chile	43(35)	N	N	LC	LR/NT	NT	NT	NT
Galapagos Penguin	*Spheniscus mendiculus*	2000	1988-2016	Chile	43(35)	N	N	VU	EN	EN	EN	EN
Giant Jumping Rat	*Hypogeomys antimena*	2001	1994-2016	Madagascar	14(10)	Y	Y	EN	EN	EN	EN	EN
Blue Swallow	*Hirundo atrocaerulea*	2002	1988-2016	S. Africa	25(20)	N	N	VU	VU	VU	VU	VU
Horned Guan	*Oreophasis derbianu*	2002	1988-2016	Mexico	38(26)	Y	Y	EN	EN	EN	EN	EN
Malayan Tapir	*Tapirus indicus*	2003	1986-2014	Malaysia	32(14)	Y	Y	VU	EN	EN	EN	
Harpy Eagle	*Harpia harpyja*	2003	1988-2016	Mexico	?	Y	Y	LR/NT	NT	NT	NT	
Mountain Tapir	*Tapirus pinchaque*	2004	1965-2014	Colombia	66(48)	Y	Y	EN	EN	EN	EN	
Maned Wolf	*Chrysocyon brachyurus*	2005	1965-2015	Brazil	51(47)	Y	Y	LR/NT	NT	NT	NT	
Okinawa Rail	*Gallirallus okinawae*	2006	1988-2016	Japan	62-90 (?)	Y	Y	EN	EN	EN	EN	
Lowland Tapir	*Tapirus terrestris*	2007	1986-2018	Brazil	74(64)	Y	Y	VU	VU	VU	VU	
Rio Grande Silvery Minnow	*Hybognathus amarus*	2007	1990-2018	USA	41(22)	Y	N	EN	EN	EN	EN	
Mangrove Finch	*Geospiza heliobates*	2008	1988-2018	Ecuador	18(10)	Y	Y	CR	CR	CR	CR	

3.4 Results

Compared to extinction risk at the time of the workshop, two of the 46 species with data for up to 10 years post-planning had improved in status after 10 years, 10 had declined and 34 were stable (with some of the latter declining initially before returning to their previous status). After 15 years, three of the 35 species with data for up to 15 years post-planning had improved in status, nine had declined and 23 were stable (**Table 3.1**). Before the planning workshop, mean status for these 35 species was between Vulnerable and Endangered. Afterwards, the mean extinction risk continued to increase until 10 years post-planning, after which it decreased, leaving mean status between Endangered and Critically Endangered by year 15 (**Figure 3.2**). No species went extinct in the timeframe. One species temporarily classified as Extinct in the Wild (by year 10) underwent revision to Critically Endangered following reintroduction (**Figure 3.2**).

For both the datasets (n=35 and n=46), there was no significant difference between simulated and observed RLIs for time-steps -5 and 0 (**Tables 3.2a, 3.2b, Figures S3.1a, S3.1b**), indicating the bootstrap procedure was unbiased and the overall pre-workshop trend was not driven by a few species with unusual trajectories, validating the pre- versus post-planning comparison.

Time step	RLI observed	RLI simulated	95^{th} quantile	p-value
-5	0.48	0.48 ± 0.03	0.52	0.52
0	0.43	0.43 ± 0.02	0.45	0.57
5	0.41	0.38 ± 0.03	0.41	0.11
10	0.37	0.32± 0.04	0.39	0.04
15	0.39	0.26± 0.04	0.34	0.00018

Table 3.2a. Observed and simulated Red List Index (RLI) values, 95^{th} quantile and associated p-value comparing observed and simulated RLI for 35 species (the number of projects with 15-years data post-planning workshop). Time steps began 5 years before the planning workshop was held and extended to 15 years after.

Time step	RLI observed	RLI simulated	95^{th} quantile	p-value
-5	0.49	0.49 ± 0.03	0.49	0.52
0	0.44	0.44 ± 0.02	0.42	0.57
5	0.43	0.39 ± 0.03	0.42	0.06
10	0.37	0.35 ± 0.04	0.39	0.01

Table 3.2b. Observed and simulated Red List Index (RLI) values, 95^{th} quantile and associated p-value comparing observed and simulated RLI for 46 species (the number of projects with 10-years data post-planning workshop). Time steps began 5 years before the planning workshop and extended to 10 years after.

For the species with data up to 15 years post-planning (n=35), observed RLI values post-planning were consistently higher than the simulated means (without planning) and increasingly so as time after planning increased (**Figure 3.3, Table 3.2**). By years 10 and 15 the difference was statistically significant (p-values <0.04 and <0.001 respectively), signifying a post-planning improvement in overall extinction trend unlikely to have arisen by chance. An increase in sample size (to n=46) strengthened the effect (p-values at 10 years are < 0.04 and < 0.01, for n=35 and n=46 respectively), as did increasing the number of years (p-values at 10 and 15 years for n=35 are < 0.04 and < 0.001 respectively). Without planning, over the 15-year timeframe following planning, the simulated trajectory predicted the extinction of 7.8 ± 2.5 species (15-29%) of the 35 considered (Year 15 RLI = 0.274).

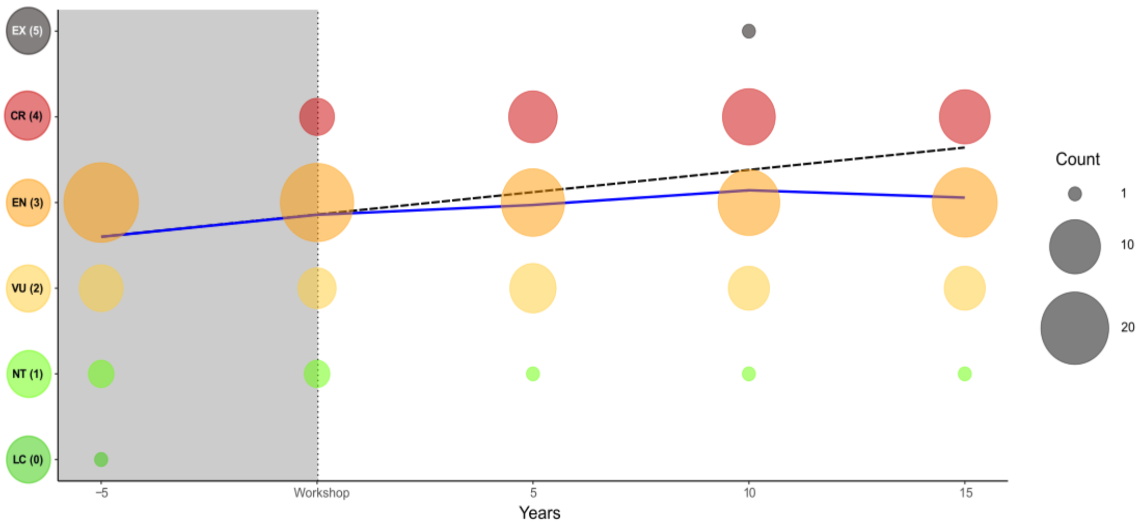


Figure 3.2. Relative allocation of 35 species to extinction risk categories at 5-year intervals, beginning 5 years before the planning workshop and continuing to 15 years afterwards. Extinction risk categories are: Least Concern (LC), Near Threatened (NT), Vulnerable (VU), Endangered (EN), Critically Endangered (CR) and Extinct (EX). By weighting these such that LC=0 and EX=5, we calculated the “mean extinction category” (blue line) and compared this to the extrapolated pre-planning trend (black dotted line).

3.5 Discussion

This is the first systematic study to demonstrate the benefits to species' conservation status resulting from an integrative, multi-stakeholder planning approach employed by the IUCN SSC. Systematic reviews of other approaches have drawn conflicting conclusions about the impact of planning, and the purpose of this study was to improve the information available to decision-makers charged with determining whether and how to invest in planning the conservation of threatened taxa. In this study, we measured the response of a group of species to this planning approach, to assess whether species' conservation prospects were better after planning than before it. Our results show that post-planning, the aggregate rate of decline to extinction was slowed significantly by year 10 and reversed by year 15. Meanwhile, our simulated counterfactual scenario (projecting the expected declines without planning), predicted the extinction of 7.8 (± 2.5) species over the same timeframe, in stark contrast to the zero extinctions observed with planning. However, because declines continued for a period after planning, only 3 (8.6%) of 35 species had improved in status by year 15, while 9 (25.7%) had declined and 23 (65.7%) had remained stable.

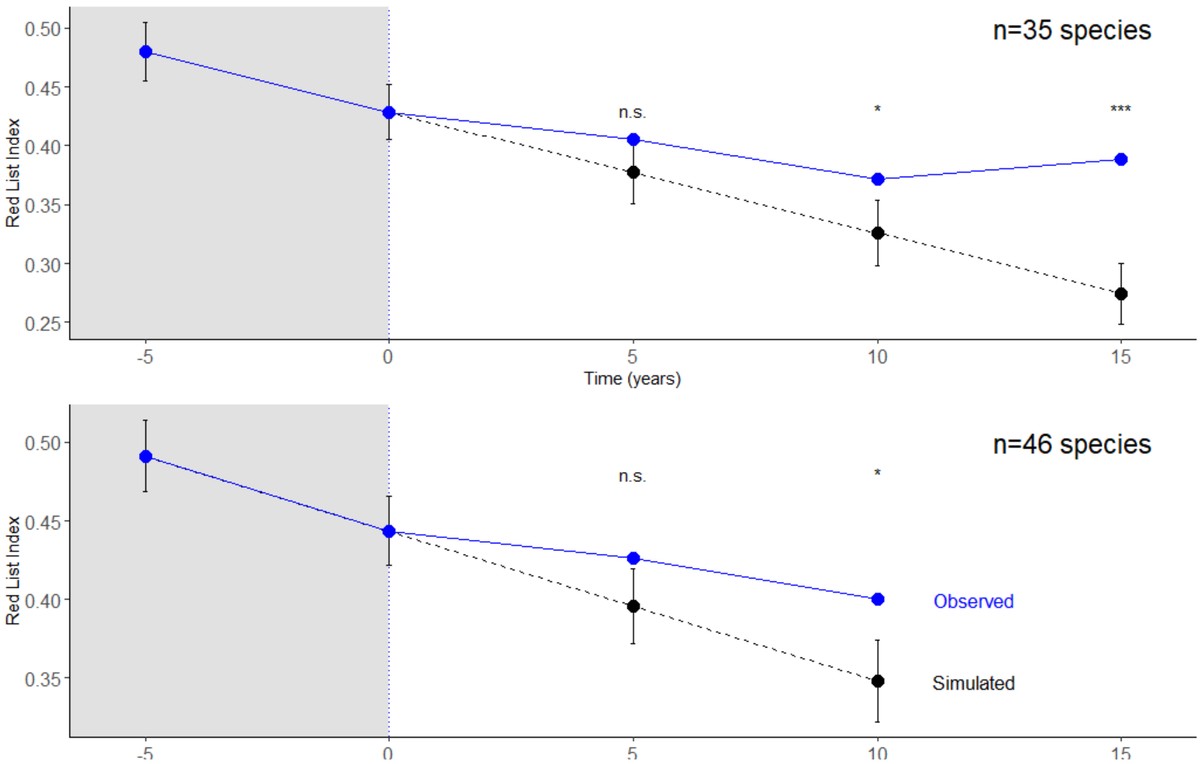


Figure 3.3. Aggregate extinction trends for species before planning and for up to 10 (n=46) and 15 (n=35) years after planning (blue line). Pre-planning trends projected to 10 and 15 years afterwards (black dotted line). Error bars depict standard deviations for simulated Red List Indices. * p-value <0.05; ** p-value <0.01; *** p-value <0.001; n.s. non-significant

Before assuming the observed turning point to be the result of planning, it was necessary to review and eliminate rival alternatives (Ferraro, 2009). We discounted the possibility that changes in threat status of these species simply reflected global trends for similar taxa over the relevant period as during this time (1990-2018), declines are reported in the overall RLIs for mammals (33 of the species examined here), birds (nine species), and amphibians (one species) (WWF 2020).

Next, we considered whether the turning point had been created or exaggerated by our project selection criteria. In relying on the availability of RL assessments pre-, post-, and at the time of planning, project selection may have been biased towards better-studied and potentially more recoverable species. However, of the 96 species-level projects available to us in the database, only six species (3%) were excluded because they had no Red List data. The 46 species that met the criteria for our study underwent a mean of 7.54 assessments (S.D. = 1.83) between 1986 and 2018 and the remaining 45 that did not, were assessed similarly often (mean = 6.09; S.D. = 3.23) with the timing of assessments relative to workshops the primary cause of exclusion. We conclude this eliminates project selection bias as the cause of the observed results.

We also considered whether the observed turning point could have been the result of measures set in place before the workshops, the results of which were only observed and recorded in the years after them; that is, that the workshops were a symptom of improved conservation efforts and not a cause. We ruled this out on the basis that workshop reports described the prevailing circumstances as those in which conservation efforts had stalled or were frustrated by, for example: conflicting views among stakeholders (e.g. projects 20, 28, 45, **Table S3.1**); uncertainty about how to proceed with conservation action (e.g. projects 15, 29, 46); or limited coordination or connectivity among implementers (e.g. projects 13, 24, 33).

Finally, we considered whether planning could have coincided with other events that were the real trigger of the turning point, such as the beginning of conservation action for the species or a sudden injection of resources. However, workshop reports and Red List accounts indicate that in all cases, conservation activities such as legal protections for the species or its habitat, had begun years and often decades before the workshops (see supplementary material **Table S3.1**). We found no evidence of sudden resource investment, with conservation in the countries considered reported to be chronically under-resourced at the time (James et al. 1999).

On balance, the information available supports the proposition that the post-planning outcomes observed in this study were triggered by the planning intervention itself, rather than by coincidental factors or project selection bias. It is important to stress that we do not suggest that planning caused the observed changes in species conservation status - these were the result of conservation action taken by multiple agencies and conservation donors, working over several decades, in many countries. Our proposition is that this planning approach created a turning point in conservation efforts for these species that led to an overall improvement in outcomes in the years that followed.

The information gathered also allows insights into why this approach was beneficial for these species. The approach routinely integrates four elements that are not widely practised in combination: i) population viability analyses (PVA); ii) inclusion of both *in situ* and *ex situ* conservation expertise; iii) facilitated participation of diverse stakeholders; and iv) an emphasis on rapid production of outputs.

At the time of planning, most of the study species had experienced population declines or fragmentation (see **Table S3.1**). Population viability analyses were included in 91% of projects (see **Table 3.1**), using simulation models built with the program *Vortex* (Lacy & Pollack 2021). These models supported not only investigation of the effects of deterministic threats to species, but also of the stochastic forces that can disproportionately influence the population dynamics of species with small or highly fragmented populations (Shaffer 1981), leading to improved understanding and prioritisation of risks.

For species with elevated stochastic risks, mitigation of deterministic threats (such as habitat destruction and over-harvest) may not be sufficient to avert extinction (Foose et al. 1995). For the species in this study, the habitat and legal protections that were in place at the time for most species were clearly achieving only limited success (see **Table S3.1**). In such cases, urgent and intensive management at the level of populations and individuals, which explicitly targets demographic and genetic stochastic risks, may also be needed (Goodman 1987; Foose et al. 1995; Frankham et al. 2017). These measures may involve *in situ* or *ex situ* activities, or a combination of both. Including the knowledge and know-how of both *in situ* and *ex situ* communities from the outset of planning can lead to better-integrated solutions, improving downstream results (Byers et al. 2013). *Ex situ* recommendations were included in plans for 87% of the study projects (see **Table 3.1**) and benefits accruing to several of these species as a direct result are reported elsewhere (e.g. CBSG 2017; Young et al. 2014).

Though still relatively rare in species conservation planning, the inclusion of diverse stakeholders in planning decisions is widely advocated in environmental decision-making, premised on the understanding that science-based prescriptions alone will not improve outcomes (Pullin & Knight 2004; Balmford & Cowling 2006; Knight et al. 2008). The inclusion of stakeholders (e.g. those representing governments, agriculture, fisheries, academia, NGOs, local communities or the private sector) can confer multiple benefits, including lowered cost of enforcing regulations, benefits of local knowledge, increased project capacity and the sharing of responsibility (Forgie et al. 2001). However, within and between these sectors, differences in background, education, influences and agendas can lead to divergent views on whether or how action is taken, and interaction and dialogue can be key to resolving these differences (e.g. Cummins 2004; Siebert et al. 2006; Brancalion et al. 2016; Maas et al. 2021). Success in this area is shown to increase where trust is secured (Young et al. 2016), conflicts are surfaced and managed (Madden & McQuinn 2014) and those involved are united behind a clear and common purpose (Black 2015). These outcomes can be advanced effectively in face-to-face workshops guided by third-party facilitation (Drolet & Morris 2000; Mackelworth et al. 2012). The challenges that precipitated planning for the species in the study set included stakeholder conflicts and uncertainty, and limited coordination among implementers (see **Table S3.1**). In these circumstances a facilitated multi-stakeholder approach conferring the benefits described above, should improve outcomes. All projects adopted this method, with an average participation of 46 individuals (range 14 - 93) and 31 institutions (range 10 - 64) per project (see **Table 3.1**). We do not have specific information on how effective this approach was in all the study projects. However, earlier evaluations of some of the same projects concluded the participatory approach of the workshops was effective in fostering collaboration (CBSG 2017; Vredenburg & Westley 2003). In particular, surveyed stakeholders reported improved clarity, and uniting of disparate groups over the short-term, promoting increased collaboration on action and research, improved understanding of other stakeholders' viewpoints and greater support for on-ground action, over the longer term. We assume that similar benefits were experienced by the study projects and that this contributed to the observed result.

Lastly, criticism has been levelled at plans that take years to produce (Tear et al. 1995) potentially creating a hiatus in decision-making, permitting or in undertaking key activities. For threatened species with urgent needs, such delays can facilitate further declines and exacerbate the difficulty of recovery (Martin et al. 2012; Hutchings 2015). In the approach studied here, planning participants committed in each case to documenting the agreed plan swiftly (within 6-12 months), with the accompanying aim of minimising the inevitable trade-offs between speed, and quality or completeness, by siting plans within an iterative cycle of regular review and adaptation (Salafsky et al. 2002). It is assumed that this contributed, at least in part, to the post-workshop momentum described in CBSG (2017). In short, this planning approach was effective because it brought analytical tools well-tailored to the conservation needs of the species targeted, and a participatory decision-making environment that supported those involved to transition swiftly to more effective ways of working together.

To date, this approach, along with species-based conservation planning in general, has been applied mainly to vertebrates, and among those, to larger-bodied, higher-profile and more charismatic species, reflecting a well-recognised human bias in the value (and therefore the resources) apportioned to different taxa (Tear et al. 1995; Metrick & Weitzman 1996; Laycock et al. 2009; Watson et al. 2011a, Brambilla et al. 2013; Drinan et al. 2020). The planning principles and tools involved could benefit a wider range of threatened species but resources and time are obstacles to broader application. There are currently 37 480 taxa classified as threatened (IUCN 2021) and though many of these may respond sufficiently well to general conservation measures targeted at area protection and threat mitigation, thousands may not. Planning for these species individually will be both too costly and too slow. Applying planning approaches such as this to well-chosen multi-species groups may be part of the solution.

Multi-species planning is not a new idea, though its application to date has received mixed reviews (e.g. Clarke & Harvey 2002; Moore & Wooller 2004; Cullen et al. 2005; Baptista 2019). Nevertheless, successful outcomes should be achievable with careful attention to the design of the planning approach and to the method of grouping species. Productive groupings for planning are expected to be among species that: share similar threats within a defined geographic or political area; rely on the same (threatened) ecosystem, habitat, or micro-habitat; are otherwise similarly affected by the same primary threats; share a need for intensive management either *in situ* or *ex situ;* or have needs that coincide closely with those of a higher-profile “umbrella species” (Burbidge 1996; Machado 2005; Foin et al*.* 1998; Jewell 2000; Clark & Harvey 2002; Branton & Richardson 2011; Ward et al. 2019)*.* The approach to planning described in this study, with some modification, has recently been trialled with multi-species groups of taxa including freshwater fish, reptiles, insects and trees (e.g. Gibson et al. 2020; Lees et al. 2020) and we recommend further application and evaluation, covering a broader array of taxa.

Studies have shown that time to recovery varies between species depending on biology and circumstances, with recovery particularly challenging for long-lived species, species with small and fragmented populations and species with particularly intractable threats (Abbitt & Scott 2001; Cardillo et al. 2005; Davidson et al. 2009; Hutchings 2015). Our study set was dominated by larger-bodied, longer-lived taxa with small or fragmented populations and some of the most difficult conservation challenges, including competition with people for habitat and food, and unsustainable harvesting (Ceballos & Ehrlich 2002; Sutherland 2001; Bennett 2015; **Table S3.1**). Further, obstacles linked to social, institutional, and organisational factors that can delay effective action (Ortega-Argueta 2020) were frequently reported (**Table S3.1**). As a result, we would have predicted the species in this study to show longer recovery times. Nevertheless, we chose this planning approach partly because of the long period over which it has been used (>30 years), which we hoped would provide enough time for plans to have been implemented, species to have responded to interventions, and for changes in status to have been measured and reported. However, available data provided only 10-15 years of post-planning information in most cases. Though by year 15 we were able to show an overall upward trajectory in species prospects, many taxa had not regained their pre-workshop conservation status in that time and a longer evaluation period is needed to confirm outcomes. This illustrates again one of the difficulties of evaluating planning impact.

In general, given that the species we examine do represent some of the most challenging for conservation, the timeframe to positive results shown in this study may sit at one extreme of the possible range. Though other studies also report times to recovery signals in excess of a decade (e.g. Beck et al. 1994; Schultz & Gerber 2002; Young et al. 2014) this may again reflect a general bias in the conservation attention assigned to particular taxa. Such long timeframes present a challenge for nations aspiring to measurably improve the status of threatened species within a decade, in line with CBD commitments. However, in the previous section we recommended expanding this style of planning beyond the usual targets, to currently neglected species of animals and plants. Many of these are smaller-bodied, with shorter generation lengths, larger population sizes and consequently shorter potential recovery times. An expected added benefit then, of this expanded effort, would be an overall increase in the rate of species recovery.

One of the reasons that the RLI is such a valuable metric is that it is based on RL categories which are designed to be comparable across taxa. However, because of the need for broad applicability, a considerable change in species' prospects is required to trigger a shift in category (and therefore in RLI), such that hard-won improvements (or declines) are masked within shorter timeframes. The IUCN's new Green Status assessment, which scores the recovery status of species at finer scales and accounts for recovery potential (Akçakaya et al. 2018), could be a valuable additional metric for use in future evaluation. We recommend that government and non-government agencies responsible for generating large numbers of threatened species plans use the Red List Index as a primary aggregate measure for evaluating planning impact and consider combining it with the Green Status assessment once this metric is more widely available.

In summary, this study demonstrates the benefits of a science-based, participatory planning approach to a group of species facing multiple threats, and where conflicting views, uncertainty, or lack of coordination among stakeholders constrained action. These circumstances are common to many threatened taxa for which planning is needed but not currently resourced. Given the results described here, we recommend extending the use of this approach to more of the taxa that could benefit. In addition, as an efficient way to extend its contribution beyond the usual targets (longer-lived, charismatic mammal and bird species) we recommend evaluating its application to carefully selected multi-species groups featuring some of the more neglected (though sometimes more easily recovered) animal and plant taxa.

Though we stress that it is conservation action on the ground that generates good outcomes for species, our study provides evidence that with the right approach, species conservation planning can provide a turning point in species conservation efforts, supporting those involved to transition quickly to more effective ways of working together.

CHAPTER 4. POPULATION VIABILITY ANALYSIS PROVIDES INSIGHTS INTO THE POTENTIAL FOR CONSERVATION INDEPENDENCE IN THE NEW ZEALAND KĀKĀPŌ.

"*All models are wrong, but some are useful*." Box (1976)

4.1 ABSTRACT

Efforts to restore endangered species should prioritise the creation of self-sustaining, ecologically functional populations capable of thriving despite future environmental challenges. Modelling can significantly enhance recovery plans by assessing how management strategies and ecological factors impact population viability. The kākāpō (*Strigops habroptilus*) is a threatened parrot endemic to Aotearoa New Zealand. Intensive management aimed at preventing extinction and driving recovery has successfully moved the population from 51 individuals in 1995 to 185 in March 2023. However, current management, which generates c. 5% annual growth cannot easily be sustained on a larger scale. Without this intervention, the population is predicted to decline by c. 1.6% annually. As release sites are approaching capacity, the need for new sites is evident. We developed models to explore two possible futures for the species: an unmanaged scenario in which either good fortune or well-targeted translocations lead the population to a self-sustaining state. The other scenario anticipates a slowly declining wild population continually supplemented by an intensively managed sub-population. In the unmanaged scenario, sustained growth depended on continued suppression of mammalian predators and high rimu (*Dacrydium cupressinum*) ripening rates (or an equivalent food source). Larger populations and higher density flocks played important support roles (though the latter is an untested intervention). Some proposed southern sites may not provide all requisite conditions and a shift north may be needed. Failing to meet these conditions might mean an ongoing decline, potentially necessitating an intensively managed sub-population of 500 to 5000 birds to counterbalance future population decline. However, at current rates the population may only reach c.700 by 2050, allowing time to explore alternative strategies. This study illustrates how population

viability analyses add value to species conservation planning by providing new insights and a means to test hypotheses for which empirical studies would be impractical.

4.2 Introduction

Population Viability Analysis (PVA) describes both the process and the set of quantitative tools used to estimate the probability that a population, or collection of populations, will persist, grow or remain genetically healthy, for a specified period, under a specific set of environmental or management conditions (Beissinger & McCullough 2002; Woodroffe 2001).

Three broad categories of PVA have been suggested: 1) “rules of thumb”; 2) analytical approaches; and 3) simulation approaches (Thompson 1991). Typically, PVAs attempt to evaluate and integrate the full range of forces affecting a population’s dynamics and persistence. Factors considered may include deterministic, stochastic, environmental, demographic and genetic influences, which can interact in complex ways that are affected by population size. For this reason, they are particularly useful for analyses of small or highly fragmented populations where the role of chance is most acutely felt (Goodman 1987; Shaffer 1987; Burgman et al; 1993; Lacy 2019). While the presence of these interacting forces is generally accepted, PVA models help to estimate the relative role of each, which can affect the evaluation of management options and decisions (Beissinger 2002).

In planning species conservation, PVAs have been used to: predict population trends and estimate the probability of populations going extinct over a given time (Shaffer 1981, 1987); assess which of a suite of management or conservation strategies is likely to maximize the probability of a population persisting (Akçakaya et al. 2000); help identify minimum viable population sizes to meet conservation objectives (Reed et al. 2003; Traill et al. 2007); identify the priority life-stage for conservation attention (Crouse et al. 1987); evaluate alternative strategies for reintroductions (Licht et al., 2017); determine potential impacts on populations from human-mediated threats (Luck et al. 2022); identify the biological or environmental factors most influencing population growth (Williams et al. 2017); and to investigate potential impacts of demographic and environmental stochasticity on survivorship and reproduction (Fox 2005).

The future cannot be predicted with certainty and many authors advocate caution in the use of PVA for this purpose (Coulson et al. 2001; Lacy 2019). However, making decisions for the future necessarily requires making some assumptions about possible future states and conditions. Therefore, as part of a broader toolkit for decision making, PVA is generally acknowledged to be of value in improving understanding about the processes affecting population dynamics, identifying key data gaps, and evaluating and comparing different scenarios involving changed management conditions or future states (Boyce 1992; Coulson et al. 2001; Brook et al. 2000; Gerber & González-Suárez 2010).

The kākāpō (*Strigops habroptilus*) is a large, nocturnal, flightless, ground-dwelling parrot endemic to Aotearoa New Zealand (henceforth Aotearoa). Once widespread across the North, South and Stewart Islands of Aotearoa, its distribution contracted dramatically following human colonisation and due to the introduction of rats (*Rattus norvegicus* and *R. exulans*) and stoats (*Mustela erminea*), and kākāpō are now no longer found in their former range. By 1977, the known population was reduced to 18 males in Fiordland and a newly discovered but rapidly declining population of c. 150 birds on Stewart Island (Rakiura). Between 1980 and 1992, 61 of these birds, along with the last remaining male from Fiordland, were transferred to mammalian predator-free offshore islands, where they remain today. In 1995, when the population was at an all-time low of N=51, a regime of intensive management was implemented that stabilised numbers and began to drive growth. By March 2023 the population numbered 185 birds distributed across four island locations (Higgins 1999, Clout & Merton 1998, Merton et al. 1999, Elliott et al. 2001, Clout 2006, Digby et al. 2023). The success in population growth has led to assessment of further offshore islands and mainland predator-free sanctuaries as possible future sites for translocations (**Table 4.1, Table S 4.1**).

Table 4.1: Summary of characteristics of current and proposed sites

Sites	Total adult birds present (2022)	Estimated adult bird carrying capacity	Estimated total kākāpō carrying capacity (all age-classes, based on stable age-structure)	Risk: % likelihood of occurrence; impact as a multiplier of annual survival. *note that for Maungatautari and Wainuiomate where risks have not been estimated, Whenua Hou values were applied.		
				Stoat	**Rat**	**Fire**
Existing breeding sites						
Whenua Hou (WH; Codfish Island)	74	82	122	0.5%; 0.96	3.3%; 0.95(chicks)	2%; 0.7
Pukenui (PUK; Anchor Island)	77	67	100	6.6%; 0.95	3.3%; 0.95(chicks)	2%; 0.7
Te Hauturu-o-Toi (HOT; Little Barrier Island)	9	94	138	0.5%; 0.96	3.3%; 0.95(chicks)	2%; 0.7
Te Kākahu-o-Tamatea (TKOT)	25	32	47	6.6%; 0.95	3.3%; 0.95(chicks)	-
Proposed breeding sites						
Five Fingers Peninsula (FFP; Taumoana)	0	194	291	-	-	2%; 0.7
Coal Island (COAL)	0	64	92	6.6%; 0.95	3.3%; 0.95(chicks)	2%; 0.7
Maungatautari* (MTT)	0	113	169	Not estimated	Not estimated	Not estimated
Resolution Island (RES)	0	1235	1817	14%; 0.85(adults) 0.6(chicks)	3.3%; 0.95(chicks)	2%; 0.7
Wainuiomata* (WAI)	0	197	298	Not estimated	Not estimated	Not estimated
Rakiura (RAK; Stewart Island)	0	8235	12117	0.5%/0.75	3.3%; 0.5(chicks)	2%; 0.7
Sites permanently considered “bachelor sites”						
Pearl Island	-	-	-	-	-	-

Despite this success, the kākāpō is classified as Critically Endangered by the IUCN (Birdlife International, 2018) and remains highly conservation dependent. Though the presence of introduced rats and stoats is considered the main reason this species cannot survive in areas of its former range, infrequent breeding, high infertility and low hatching success hamper conservation efforts even where introduced predator impacts are well-controlled (Clout 2006; Digby et al. 2023). To date, overcoming these challenges to create growth has required an intensive regime. This includes pest control alongside various measures such as nest management, supplementary feeding of chicks, artificial incubation of rescued eggs, hand-rearing, artificial insemination, inter-island translocations and implementing careful genetic management (Digby et al. 2023; A. Digby & D. Eason pers. comm.). These efforts aim to support appropriate pairings and ensure the lifelong reproductive output of individuals. Without these interventions, recently developed integrated Bayesian models fitted to data from 1995 – 2020 predict the population would revert to a decline of c. 1.6% p.a. (Kākāpō Recovery Group (KRG) pers. comm.).

4.2.1 SIGNIFICANCE FOR CONSERVATION MANAGEMENT

Species recovery efforts should aim to restore self-sustaining, ecologically functional populations of a species across available areas of its former range, and to ensure that they grow to sufficient abundance and are equipped with enough genetic diversity, to be able to adapt to future environmental changes (Redford et al. 2011; Grace et al. 2021). In Aotearoa, Indigenous stakeholders in kākāpō recovery advocate a similar concept when they talk of restoring the species' "wairua", a term that speaks to the holistic well-being of an individual and the spiritual synergy of the collective with which that individual identifies (Kākāpō Management Group 2006).

Moving towards this goal by removing the current reliance of kākāpō on intensive management is a complex issue due to the multiple challenges to survival and breeding as well as uncertainties about the long-term utility of current and potential sites, which vary in size, latitude, vegetation composition, carrying capacity, and risks from other species (especially stoats) (**See Table 4.1**). Any moves to withdraw or reduce support must proceed cautiously as the current intensity of care and ongoing presence of mammalian predators may mask underlying and irreversible reductions in species fitness resulting from recent bottlenecks and inbreeding. This could constrain unassisted population growth, especially at the outer limits of the former range.

Nevertheless, management of the kākāpō recovery is in an expansion phase. The population is outgrowing capacity at current sites and new sites must be found and populated. At the same time, sustaining current management intensity for such a large and growing population is a strain on resources (KRG pers. comm.). Options for greater conservation independence need to be identified and carefully weighed and this work is intended to inform thinking around this, in particular with regard to, the criteria for prioritising new sites, and decisions about how to distribute birds among proposed and existing ones.

The ideas considered here may also be relevant to other species that are reduced to small or highly fragmented populations and for which active management intervention *in situ* or *ex situ* is required to secure recovery (Foose et al. 1995; Bolam et al. 2023).

4.2.2 This Project

We considered the question, "Is there a plausible route to greater ecological independence for kākāpō?" We considered two potential end points: 1) Full ecological independence: an unmanaged, free-living population that is self-sustaining; and 2) Partial ecological independence: an unmanaged, free-living but slowly or periodically declining population maintained through supplementation from an intensively managed sub-population (note that our definition of ecological independence assumes some ongoing level of control for introduced mammalian pests – that is, we do not expect kākāpō to become resilient to these).

4.2.2.1 Full ecological independence

We defined pathways to full ecological independence as routes through which a change in projected population growth rate could occur that would be large enough to lead to stable or growing numbers in the absence of intensive management.

Currently projected negative growth rates in the absence of management are based on models developed with the Kākāpō Recovery Group (KRG) by Wellington-based Dragonfly Science (www. dragonfly.co.nz) using decades of KRG data. With good data, models can have high predictive value (Brook et al. 2000). Though kākāpō are relatively well-studied some parameters remain uncertain. Therefore, there are two types of pathway to better projected growth: one is parameter correction, where actual species vital rates stay the same but our pessimistic estimates of them improve resulting in more optimistic models; and the other is a genuine shift in vital rates, where rates respond to an intervention that does not require ongoing close management, and models improve accordingly.

Parameter correction: a) Lifespan; b) Inbreeding severity

Many kākāpō life history traits can be quantified with confidence. Others, such as lifespan and inbreeding severity are harder to quantify because this is a long-lived species re-discovered relatively recently, sample size is small and active management has a distorting effect. Therefore, current population models draw only on rough estimates for these parameters. Lifespan is currently estimated at between 60 and >100 years (Clout 2006; Elliot et al. 2006). In the absence of species-specific estimates (there are currently none for kākāpō), inbreeding depression severity is typically modelled at default rates drawn from multi-species studies of other taxa by O'Grady et al. (2006) and Ralls et al. (1988). <u>We hypothesise that (a) the species lives longer than currently estimated and/or (b) the species is more resilient to inbreeding than assumed, leading to stronger projected growth.</u>

Shift in vital rates: c) Reduced stochastic pressure; d) Increased egg fertilisation rate; e) Increased breeding rate.

Based on current knowledge and assumptions we hypothesised three potential pathways through which vital rates could improve in an unmanaged population. All pathways considered affect one or more specific model parameters and have a plausible mechanism for implementation that does not require ongoing intensive management:

c) Reduced stochastic pressure: population outgrows negative effects of small size. As of March 2023, there were approximately 185 kākāpō, with the largest concentration on the main breeding island (Whenua Hou/Codfish) which can house only a small population of birds (estimated carrying capacity = 82 adult birds; 2023 population size = 72 (**Table 4.1**)). Small populations can be destabilised by year-to-year environmental variation in food supply, temperature, or precipitation. Such populations may struggle to recover from catastrophic events such as disease outbreak, fire or flood; they lose gene diversity rapidly, leading to the loss of adaptive potential and generally lower vitality; inbreeding accumulates more rapidly, potentially resulting in the expression of deleterious alleles; and skews in sex-ratio, breeding or mortality rates – even those within the normal range – can influence population declines. These influences operate independently of other threats and can continue in their absence. Recovery programs for species reduced to small numbers therefore often need intensive management support in the early stages, until populations are large enough to overcome these pressures (Foose et al. 1995; Goodman 2002; Bolam et al. 2023). <u>We hypothesise that as the kākāpō population grows via population expansion in existing sites, and establishment of new sites, the downward pressure on population growth from stochastic forces could diminish, raising mean growth rate.</u>

d) Egg fertility rate increases with increasing population density. Egg failure is exceptionally high for kākāpō. Sixty-one per cent of eggs have failed in the last four decades (401 out of 662 between 1981 and 2019; KRG unpublished data cited in Savage et al. 2022), whereas the average across birds is estimated to be 10%–15% (Koenig 1982; Spottiswoode & Møller 2004) and four studies of different parrot species show a range of 10%-23% (Beissinger & Waltman 1991; Montinerrubio et al. 2002; Cantor et al. 2019; Masello et al. 2002). Savage et al. (2022) showed that from a sample of N=252, most egg losses for kākāpō (74%) were the result of embryo deaths with 20% the result of infertility. A likely cause is inbreeding. Kākāpō have been through a severe population bottleneck and inbreeding accumulation is exacerbated by their lek breeding system through which a small number of males dominate the gene pool (Merton et al. 1984). Inbreeding in this species can contribute to both embryo death and to reduced fertility (White et al. 2015; Savage et al. 2022; Digby et al. 2023). In a small, closed population such as this, inbreeding accumulation cannot be avoided, only slowed, and management of kākāpō is already achieving this using pedigree analysis tools to prioritise pairings (KRG pers. comm.; Guhlin et al. 2023). Another possible cause is founder effect. The relict population from Rakiura from which most current birds are descended, showed high levels of inter-relatedness (Bergner et al. 2014; Guhlin et al. 2023). The smaller number of individuals from which they are all descended may by chance have carried genetic characteristics for lower breeding success that are now widespread in the population. Whatever the route to the current situation, recent studies indicate that egg fertility increases when females mate with the same male several times and even more so when females mate with several different males and that this is more likely to occur in populations living at higher density and with a female bias (Digby et al 2023). <u>We hypothesise that increases in population</u>

density as the population grows could reduce egg infertility, raising mean population growth rate.

e) Breeding rate increases with increased availability of preferred food (e.g. due to translocation of birds to sites where rimu mast-ripening intervals (RRI) are shorter or an equivalent abundance of an alternative food source is present). Diet is an important factor in avian reproductive output but has not been studied in most wild bird species (Selman and Houston 1996; Klasing, 1998; Assersohn et al. 2021). Kākāpō breeding occurs irregularly, synchronised with the mass-fruiting (masting) of certain tree species, particularly the rimu tree (*Dacrydium cupressinum*). Rimu masts every 2–5 years depending on location (Merton et al. 1984; Powlesland et al. 1992; Eason et al. 2006; Harper et al. 2006) and is the predominant food fed to chicks when available (Cottam, 2006). However, a successful breeding year, in which kākāpō can raise their own young without support, requires that the rimu mast ripens. On the main breeding island (Whenua Hou/Codfish Island) this occurs on average only once every 10-11 years but on Pukenui/Anchor Island where the species also successfully breeds the frequency is once every 3-4 years (Eason pers. comm.). The ripening rate on other current or potential kākāpō islands is not well-studied and on one of the islands (Te Hauturu-o-Toi/Little Barrier Island) there is no rimu. We hypothesise that concentrating the population of birds in places where rimu mast ripens more frequently, or where there is an alternative food source delivering the same outcome, could increase population growth rate.

4.2.2.2 Partial Ecological Independence

For reasons described above it is possible that without intensive management kākāpō would not become self-sustaining after translocation to all currently proposed sites. However, the species is long-lived and projected declines in the absence of management (in areas where introduced predators are effectively controlled) could be relatively slow. Once populated, new sites could be left unmanaged, except for predator control, and gradual declines corrected by periodic supplementation from a sub-population of birds that remains under intensive management. The feasibility of this option depends upon the rates of supplementation required and, therefore, the size of the intensively managed population needed. These in turn depend on the specific rates of decline expected at potential release sites, which can vary due to environmental conditions, carrying capacity, and risk and likely impact of predator incursion. We used site-specific models to look at the likely scale of supplementation needed once sites are at capacity and, separately, considered the timeframe over which this capacity might be reached under different management scenarios.

4.2.2.3 VORTEX

We use the VORTEX modelling program throughout these analyses, to test the hypotheses described and to estimate supplementation rates required for sites where sustained growth is not achieved. VORTEX is a simulation software package written for PVA (Lacy and Pollak 2021; Lacy et al. 2021). It was selected because it is particularly well-suited to emulating the pressures on small wildlife populations both from deterministic forces and from genetic and demographic sources of stochasticity.

4.3 METHODS

4.3.1 BASELINE SINGLE SITE MODEL FOR KĀKĀPŌ

A baseline simulation model was built using VORTEX. The baseline was designed to emulate a population with the characteristics of that currently living on Whenua Hou, but with all management removed (other than current predator control measures). Data inputs (including life history, demographic, genetic and environmental parameters) were drawn from the published literature with remaining gaps filled by best-estimates from Kākāpō Recovery Group members. Most available data are for intensively managed kākāpō. The components of vital rates attributable to management were calculated or estimated for the previous KRG project with Dragonfly Science. Those values were supplied by the KRG and are applied in various places in the baseline model (referred to as KRG-DF unpubl.). **Table 4.2.** provides a summary of baseline model values and their sources. Further details are provided in **Appendix B.**

Four types of uncertainty are included in the baseline (and in all subsequent scenarios):

<u>Demographic uncertainty</u>: random fluctuations in individual mortality and breeding rates that arise because these rates are discrete and probabilistic events. These events are modelled as such in Vortex, which is an individual-based model. As these effects tend to average out in large populations, demographic stochasticity is most important in small populations where it

can cause or exacerbate population fluctuations (Shaffer 1981, 1987; Lande 1993; Lacy et al. 2021).

Year-to-year environmental uncertainty: variation in mortality and breeding rates attributable to environmental perturbations affecting all individuals (within each age-class) in a population. This affects both large and small populations. In Vortex the variation in breeding and mortality rates caused by this are described by a normal distribution, the mean and standard deviation of which are set by the user (Lande 1993; Lacy et al. 2021). In absence of empirical data, standard deviations are set in the baseline models at 10% of the mean for age-specific mortality and 20% for the annual percentage of females breeding.

Catastrophes: occasional, extreme fluctuations in breeding or mortality rates that have a large impact on population size. Catastrophes affects both large and small populations. In Vortex these fluctuations can be positive or negative. They are included in the model by setting an annual probability of occurrence and a multiplier to “normal” breeding or mortality rates (Lande 1993; Lacy et al. 2021). Catastrophes are used in the baseline model to emulate the effects of rimu ripening years and in other models to emulate low frequency, high impact threats (stoats, rats, fire and flood).

Genetic uncertainty: depressions in fitness arising from declining genetic variability. Vortex tracks the accumulation of inbreeding in the population and allows the user to set its severity and mode of influence (either through the impact of recessive, lethal alleles, through the general loss of fitness resulting from the accumulation of sub-lethal deleterious alleles or through a mixture of both). Unless otherwise configured, models apply the impact of inbreeding only to first year mortality. These effects have a bigger impact on small populations and, within a given time period, on species with shorter generation times. In absence of empirical values for this species, default values are used in all models for both Lethal Equivalents present and for the percentage allocation of these to recessive lethal alleles (Lacy et al. 2021).

Incorporation of this uncertainty into the baseline (and into all other models) helps provides greater insight into the potential range and distribution of possible outcomes for the population and is especially pertinent for smaller population size scenarios.

Table 4.2. Summary of values used in the baseline VORTEX model for kākāpō with justification and sources. KRG-DF unpubl. denotes values provided by the Kākāpō Recovery Group from a previous modelling project. All baseline VORTEX input parameters are provided in **Appendix B**.

Vortex parameters	Baseline	Justification and sources
Inbreeding Depression	Yes	Reported for this species (Bergner 2014; White 2015)
Lethal equivalents (LEs)	6.29	No species-specific value. VORTEX default applied (O'Grady et al. 2006).
Breeding system	Polygynous	Lek breeder (Merton et al., 1984)
Median age of first offspring, both sexes (years)	10	Calculated from known age birds (Elliott et al. 2006; Eason et al. 2006).
Maximum age of reproduction, both sexes (years)	65	Oldest current breeders of unknown age. Assumed no reproductive senescence (Holmes et al. 2003).
Maximum lifespan (years)	65	Precautionary end of available estimates (Clout 2006; Butler 1989 cited in Elliott et al. 2006; Digby et al 2023).
Maximum number of broods per year	1	Observed value (e.g. Clout et al. 2002).
Maximum number of progeny per brood	4	Observed value (e.g. Clout et al. 2002)
Sex-ratio at birth in % males	50	Male bias described (Clout and Merton, 1998; Robertson et al., 2000); adjusted analysis indicated parity at birth (Digby pers. comm.)
% adult females breeding	13 - 95	Reported range 5-95% varying with mast-ripening events (Elliott et al. 2006). Lower limit adjusted based on KRG-DF unpubl. values.
Mean number of offspring per female per brood	2.4 - 3.27	Observed mean 2.53±0.1 (Eason et al 2006). Adjusted to vary with mast-ripening events and no management (KRG-DF unpubl.).
Mean % mortality Females/Males		Note: Age 0-1 mortality includes egg loss from all causes (including infertility) as well as post-hatch mortality.
Age 0 to 1	84-88	Age ≥ 2yrs from observed rates (Clout 2006; Elliott et al. 2006) Savage et al 2022). Estimated 0-2yrs without management and rate elevation in mast-ripening years from KRG-DF (unpubl).
Age 1 to 2	47/50	
After age 2	1.5	
Male Monopolisation: % males breeding pool	25%	Estimated from Eason et al (2006); Savage et al (2022); Miller et al. (2003).
<u>Rimu mast-ripening events:</u> Modelled as benign “catastrophes” with a likelihood of occurrence and consequences for reproduction and mortality limited to the year of occurrence.	Freq.=9% Rep: 7.3 fold increase Surv: 25% reduction age 0-1 yr	Rate of occurrence of mast-ripening varies with location. Baseline emulates c. 11-year interval observed for Whenua Hou. In year of occurrence % females breeding increases to 95% (from 13%); mean clutch size increases to 3.27 (from 2.4); and 0-1 year mortality increases to 88% (interpreted from KRG-DF unpubl values).

4.3.2 Full ecological independence models

4.3.2.1 Single site scenarios:

Five potential pathways were identified, with associated model parameters. For each parameter a plausible range of values was assigned (see **Table 4.3**). **Simple scenarios** were modelled for each pathway by varying the relevant parameters across the plausible range keeping all other baseline parameters the same. **Composite scenarios** considered two pathways simultaneously, varying all plausible values for one against all those for another. In these scenarios, initial population size (Ni) and carrying capacity (K) were set to 500 birds to reduce noise from stochastic factors. Standard VORTEX outputs were captured for comparison: P(Ex) = population extinction risk; Stoc-r = population mean instantaneous growth rate; GD = gene diversity retained.

Table 4.3 Hypothesised pathways to ecological independence: rationale and plausible ranges of values for testing.

	Proposed pathway	Model adjustment and assumptions
a.	**Parameter correction:** The species lives longer than currently estimated.	Lifespan & age at last breeding for both sexes was varied from 65 to 100yrs at increments of 5, covering the range of published estimates. No reproductive senescence was included (Holmes et al. 2003).
b.	**Parameter correction:** The species is more resilient to inbreeding than assumed.	Impact of inbreeding depression (modelled in VORTEX as elevated first year mortality in inbred individuals) was reduced from the wild default for wild populations (6.29 LEs (O'Grady et al 2006)) to below the default for captive populations (3.14 LEs (Ralls et al. 1988)). Range modelled was 6.3 – 3.0 LEs at increments of 0.3
c.	**Population outgrows negative effects of small size.**	Population size was modelled for N=100 to 500 at increments of 50, for N=750 and for N=1000. All populations began at site carrying capacity and at stable age-structure.
d.	**Egg fertility increases with increasing population density.**	First year mortality comprises 13% egg infertility (data from Savage et al. 2022). Scenarios reduce this to 1% at increments of 2%.
e.	**Breeding rate increases with increased availability of preferred food.**	Interval between rimu mast-ripening years is varied from 11 to 2 years at increments of 1 year, spanning the range between the rate observed on Whenua Hou and that reported in the literature (Harper 2006). Ripening-associated impacts on breeding rate, clutch size and year 1 mortality remain the same (see **Table 4.1**).

4.3.2.2 Site-specific scenarios

Ten release sites were of interest: Whenua Hou on which the baseline model is based; three other current sites; and six additional proposed sites. Sites vary from Whenua Hou in characteristics likely to affect the performance of translocated kākāpō populations, potentially escalating rates of population decline. To account for this, models were created to emulate the other sites,using estimates of site-specific characteristics and risks developed by Kākāpō Recovery Group members. The following parameters were modified in the models as a result: starting population size and carrying capacity; likelihood and impact of wildfire; likelihood and impact of stoat and rat incursion (see **Table 4.1** and **Appendix B Table S4.1**,). Having identified rimu mast-ripening interval as the most influential pathway to independence, long, short and very short ripening intervals (11, 4 and 2 years respectively) were tested for each site model to illustrate a range of pessimistic and optimistic outcomes.

4.3.3 Partial ecological independence models

The partial ecological independence models look at the implications of supporting a largely unmanaged and on average declining metapopulation of birds, using a smaller, intensively managed sub-population as a supplementation source.

4.3.3.1 Supplementation rates

The total amount of supplementation required over a 100-year period, for each site, was modelled as follows: steps were added to each site-specific model to: i) calculate the difference between carrying capacity and population size at the end of each year, in each iteration; ii) top up the population by that number, with birds aged < 2 yrs using 50% females and 50% males. Where the site had a surplus of birds at the end of the year, the number of surplus was counted prior to the Vortex default step of applying an additional round of probabilistic mortality to truncate population size to capacity (set in this case to apply to birds < 2yrs old). All sites were assigned baseline (unmanaged) parameters and initiated at full carrying capacity. Trials were repeated for all sites for long (11yrs), short (4 yrs) and very short (2 yrs) rimu mast-ripening intervals. Birds of < 2yrs were specified to emulate likely management choices (for supplementation) and losses or failure to recruit (truncation step). The number of years requiring supplementation, the total number of < 2yr-old birds supplemented over 100 yrs, and the total number of surplus birds produced over the same period, were calculated from model outputs, for each site and for all three ripening intervals.

4.3.3.2 Potential harvest rates for intensively managed populations of varied size

A second series of models was built to estimate how big an intensively managed population would need to be to maintain the metapopulation at full capacity in an unmanaged state. Initial population size and carrying capacity were set to the same value, which was varied from Ni=K=100 to 1000, in a baseline model set to “intensive management” (i.e. 0-1yr mortality reduced from 84% to 62%; no catastrophes related to fire, stoats, or rats). In addition to the standard outputs the models were set to report the number of birds surplus to carrying capacity at the end of each year, for each population size modelled. This was used as a proxy for the number of birds available for translocation.

4.3.3.3 Implications for population size to 2050

The supplementation requirements modelled assume that all sites begin at the full carrying capacity targeted by 2050, whereas the population currently stands at a small fraction of that and will not reach that size for some time. No attempt was made to simulate population growth into the projected capacity over time because plausible models would require information not currently available about the likely sequence of site mobilisation, whether sites would be populated in series or in parallel, whether some sites would remain a priority over others for maintaining stocking levels, and the size and age-structure of likely release cohorts. Instead, the current adult population size (N=185) was multiplied forwards to 2050 using mean stochastic growth rates (stoc-r) from a range of scenarios spanning no management, through increasingly optimistic scenarios without management, to full intensive management, to approximate possible population sizes by 2050. To calculate population size at 2050, stoc-r values were first converted to lambda (λ) using the formula: $e^r=\lambda$. Lambda was then applied to the current population size to predict the population size in 27 years ($N_{2050}=185(\lambda^{27})$). See **Table 4.4** for the scenarios used.

Table 4.4. Example scenarios of improved prospects for unmanaged kākāpō populations and associated changes in the models (for estimating potential population growth to 2050). Current management is included as the baseline.

Example scenarios	Egg Infertility*	Ripening Interval**	Year 1 mortality
Baseline: unmanaged except for mitigation of stoat incursions, all birds at sites where RRI=11yrs.	13%	11 yrs	84%
No management, enhanced environment 1: all birds moved to sites where RRI=4yrs.	13%	4 yrs	84%
No management, enhanced environment 2: all birds moved to sites where RRI=4yrs; density improves fertility, reducing year 0-1 mortality by 2%.	11%	4 yrs	82%
No management, enhanced environment 3: all birds moved to sites where RRI=3yrs.	13%	3 yrs	84%
No management, enhanced environment 4: all birds moved to sites where RRI=3yrs; density improves fertility, reducing year 0-1 mortality by 4%.	9%	3 yrs	80%
Current management: current level of management intensity continues.	13%	11 yrs	62%

*Modelled as a component of Age 0-1 mortality (which includes egg stage)

**Modelled as a change in frequency of a benign catastrophe that increases % females breeding, mean number of offspring, and reduces Age 0-1 survival in the year of occurrence.

4.4 RESULTS

4.4.1 BASELINE SINGLE SITE MODEL FOR KĀKĀPŌ

The population described by the kākāpō baseline model (unmanaged) declines at approximately 1.6% per year (λ=0.9839), reducing population size from N=122 at Year=0 to N=23.99 at Year=100 (**Figure 4.1**). Note that N=122 is used to incorporate all ages classes at stable-age structure, within which there are N=74 adults reflecting the population at Whenua Hou. The mean intrinsic growth rate (stoc-r) is -0.0162±0.0703 and generational growth (Ro) is 0.6287. One percent of all simulated populations go extinct during the period and gene diversity declines from GD=0.9959 at Year=0 to GD=0.9300 at Year=100. The modelled generation time is 34.50 years for both sexes.

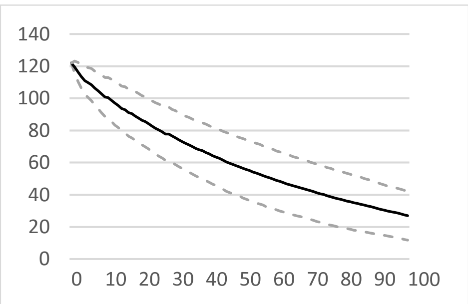


Figure 4.1 Results of 1000 iterations of the kākāpō baseline model (without intensive management) showing mean population size across iterations (solid black line) and standard deviation (dashed grey lines).

4.4.2 Full ecological independence models

4.4.2.1 Simple single site scenarios

The results of the simple scenarios (in which a single parameter is varied, keeping all others fixed) are illustrated together, for direct comparison (**Figure 4.2a-e**). The outcomes are described below:

a) Parameter correction: The species lives longer than currently estimated.

The population remains in decline when maximum lifespan is increased from 65 to 100 years. However, the rate of decline decreases as lifespan extends, reaching stoc-r=-0.0071 at a maximum lifespan of 100 years.

b) Parameter correction: The species is more resilient to inbreeding than assumed.

For the range of lethal equivalents considered (LE=3.0 – 6.30), and over the period considered, there is no directional change in the rate of population decline.

c) The population outgrows negative effects of small size.

The population remains in decline regardless of its size, over the range of values modelled (N=100–500 at increments of 50, with additional models set to N=750 and 1000). However, the rate of decline decreases most sharply as size increases from N=100-200, less so between N=200 and N=500 with no directional change at sizes beyond N=500.

d) Egg fertility increases with increasing population density.

Reducing the percentage of unfertilised eggs from 13% of total age 0-1 mortality to 1%, in 2% increments, steadily increases the stochastic growth rate of the baseline model from Stoc-r=-0.0163 to a slightly positive value of r=0.0006.

e) Breeding rate increases with increased availability of preferred food.

As the interval between rimu mast-ripening years decreases, growth rate increases. Over the range of intervals modelled (2-11 years) growth rate ranges from Stoc-r=-0.0162 to 0.0139. Population growth transitions from negative to positive at a ripening interval of approximately 3-4 years.

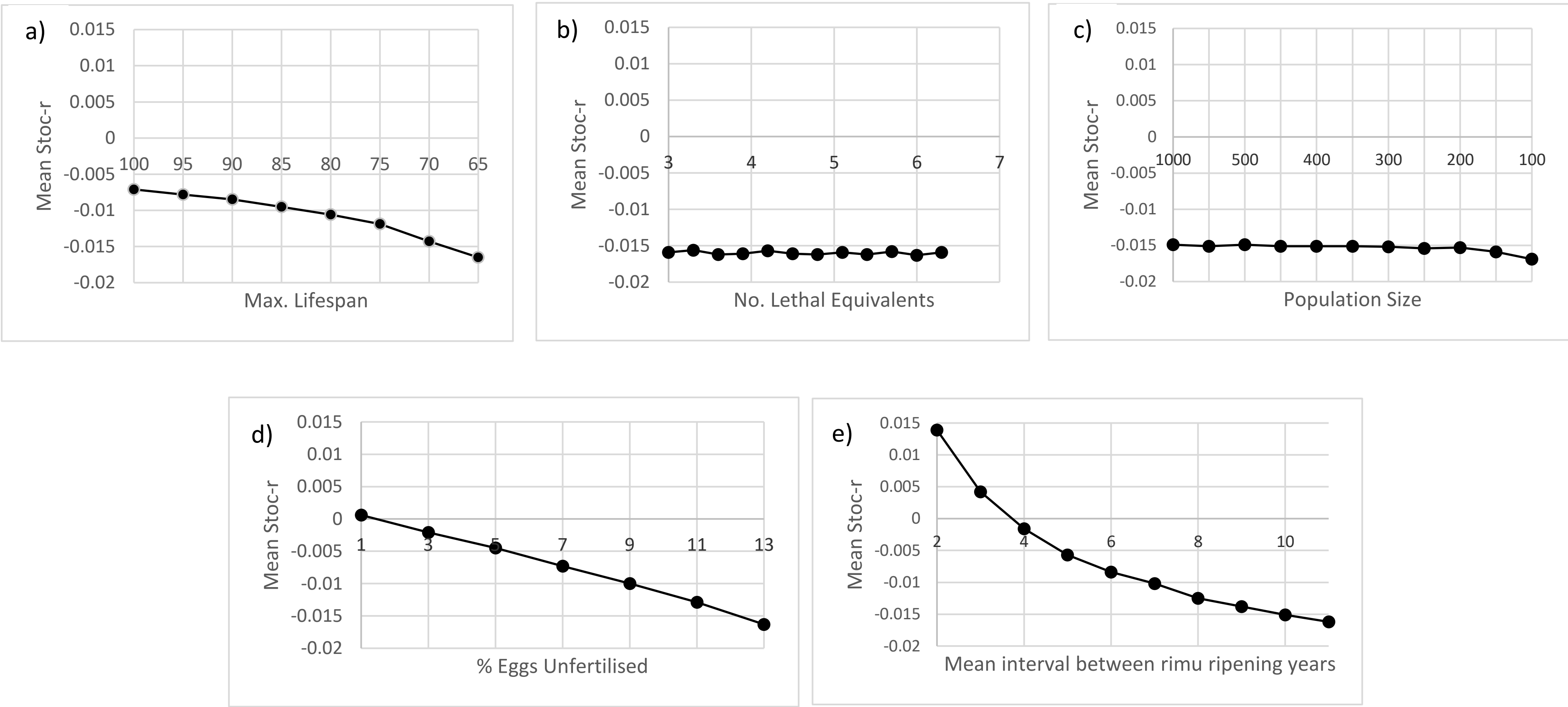


Figure 4.2a-e Impact on mean stochastic growth rate (stoc-r) of the kākāpō single site baseline model (unmanaged), of varying individual parameters across a plausible range: a) maximum lifespan varied from 65 – 100 years; b) number of Lethal Equivalents varied from LE=3.0 – 6.3; c) population size varied from N=1000-100; d) % 0-1 yr mortality due to egg infertility varied from 13-1%; e) mean interval between rimu ripening varied from 11-2 years. Points display the mean across 1000 iterations.

4.4.2.2 Composite single site scenarios

The results of the composite scenarios (in which values of two parameters were varied against each other, keeping all other parameters fixed as per **Table 4.2**) are provided in **Tables 4.5a-c**. In contrast to the baseline and simple models above, in these composite models the initial population size was set to 500 to reduce noise from stochastic effects (i.e. beyond N=500 there is no discernible difference in growth rate attributable to increased population size, see **Figure 4.2-c**). The broad outcomes are described below:

a) Percentage unfertilised eggs varied against rimu mast-ripening interval.

With starting population size fixed at Ni=500, varying the percentage of unfertilised eggs against the rimu mast-ripening interval shows a low-point in stochastic growth of r=-0.0151 and a high-point of r=0.034. Populations with the highest percentage of egg infertility (13%) show positive growth where the ripening intervals are very small (2-3 years) and populations experiencing the longest ripening interval (11 years) show positive growth where percentage of 0-1 yr mortality due to unfertilised eggs is reduced to 1% or less (see **Table 4.5a**) For -rimu mast-ripening intervals of 4 years, reduction to 11% 0-1 yr mortality due to unfertilised eggs generates positive growth.

b) Lifespan varied against rimu mast-ripening interval.

With starting population size fixed at Ni=500, varying the maximum lifespan against the rimu mast-ripening interval shows a low-point in stochastic growth of r=-0.0151 and a high-point of r=0.174. Populations with the shortest lifespan (65 years) show positive growth only where the ripening interval falls below 4 years. For the range of maximum lifespans modelled (65-100 years) positive growth rates are recorded only for ripening intervals of 5 years or less (see **Table 4.5b**)

c) Lifespan varied against % unfertilised eggs.

With starting population size fixed at Ni=500, varying the maximum lifespan against the percentage of unfertilised eggs shows a low-point in stochastic growth of r=-0.0151 and a high-point of r=0.007. Populations with the shortest lifespan (65 years) show positive growth only where egg infertility is limited to 1%. For the longest lifespans modelled (95-100 years), growth is still positive when the percentage of unfertilised eggs rises to 7% (see **Table 4.5c**).

Stoc-r	Rimu Ripening Interval (Years)									
Unfertilised Eggs (%)	2	3	4	5	6	7	8	9	10	11
1	0.034	0.0241	0.0175	0.013	0.0098	0.0074	0.0054	0.0035	0.0024	0.0016
3	0.0308	0.0217	0.0151	0.0103	0.0073	0.0048	0.0032	0.0016	0.0003	-0.001
5	0.0281	0.0189	0.0125	0.0081	0.0022	0.0026	0.0005	-0.0011	-0.0023	-0.0033
7	0.0252	0.016	0.0092	0.005	-0.001	-0.0002	-0.0021	-0.0039	-0.0047	-0.0059
9	0.0215	0.0122	0.0063	0.0024	-0.0008	-0.0032	-0.0049	-0.0062	-0.0077	-0.0089
11	0.0177	0.0093	0.003	-0.0014	-0.004	-0.006	-0.0081	-0.0096	-0.0107	-0.0117
13	0.0138	0.0052	-0.0004	-0.0048	-0.0077	-0.0095	-0.0115	-0.0127	-0.0138	-0.0151

Table 4.5a. Impact on stochastic growth rate of varying rimu mast ripening interval against the % of 0-1 yr mortality due to egg infertility. Starting population size and carrying capacity are set to Ni=K=500. All other values are those described for the baseline (unmanaged). Green shading intensifies as values become more positive; red shading intensifies as values become more negative. 1000 iterations.

Stoc-r	Rimu Ripening Interval (Years)									
Max. Lifespan	2	3	4	5	6	7	8	9	10	11
100	0.0174	0.0099	0.0051	0.0022	-0.0002	-0.0022	-0.0035	-0.0044	-0.0055	-0.0062
95	0.0174	0.0097	0.0048	0.0014	-0.0008	-0.0026	-0.004	-0.0052	-0.0062	-0.007
90	0.0172	0.0095	0.0046	0.0008	-0.0013	-0.0033	-0.0045	-0.0063	-0.007	-0.008
85	0.0167	0.0091	0.0038	0.0001	-0.0024	-0.0042	-0.0057	-0.0068	-0.0081	-0.0089
80	0.0161	0.0083	0.0031	-0.0007	-0.0033	-0.005	-0.0065	-0.0081	-0.0091	-0.01
75	0.0157	0.0075	0.002	-0.0018	-0.0044	-0.0062	-0.0077	-0.009	-0.0104	-0.0115
70	0.0149	0.0064	0.001	-0.0028	-0.0058	-0.0076	-0.0094	-0.0109	-0.0121	-0.013
65	0.0138	0.0056	-0.0009	-0.0043	-0.0076	-0.0094	-0.0114	-0.0128	-0.014	-0.015

Table 4.5b. Impact on stochastic growth rate of varying rimu mast-ripening interval against the maximum lifespan. Starting population size and carrying capacity are set to Ni=K=500. All other values are those described for the baseline (unmanaged). 1000 iterations

Stoc-r	Unfertilised Eggs (%)						
Max. Lifespan	1	3	5	7	9	11	13
100	0.007	0.0048	0.003	0.001	-0.0014	-0.0037	-0.0062
95	0.0066	0.0043	0.0026	0.0005	-0.002	-0.0046	-0.0072
90	0.0059	0.0039	0.0017	-0.0003	-0.0026	-0.0051	-0.0076
85	0.0052	0.0033	0.0014	-0.001	-0.0037	-0.006	-0.0087
80	0.0049	0.0025	0.0004	-0.0017	-0.0043	-0.0072	-0.0098
75	0.0039	0.0016	-0.0006	-0.0031	-0.0054	-0.0083	-0.0114
70	0.0028	0.0007	-0.002	-0.0044	-0.0072	-0.0097	-0.013
65	0.0012	-0.001	-0.0033	-0.0063	-0.0086	-0.0115	-0.015

Table 4.5c. Impact on stochastic growth rate of varying the % of unfertilised eggs against maximum lifespan. Starting population size and carrying capacity are set to Ni=K=500. All other values are those described for the baseline (unmanaged). 1000 iterations.

4.4.2.3 Site-specific scenarios with varied rimu ripening interval

Figure 4.3 illustrates the mean growth rate at each site, over 100 years, once site-specific risks from stoats, rats and fire are included, and with either long (11yrs), short (4yrs) or very short (2yrs) average rimu mast-ripening intervals. At long intervals, growth rates are all negative, ranging from r=-0.0176 to -0.0442. At short intervals growth is also universally negative, but less so (ranging from r=-0.0018 to -0.0301) and at very short intervals growth is positive everywhere but Resolution Island (r=0.009 to 0.0227; with -0.0137 for Resolution). The highest risk of extinction over the period is 26% (PE=0.26) at Maungatautari. However, risk of extinction over the period is generally low, with 14 of 30 scenarios showing no extinction risk. Extinctions are most likely to occur later in the 100-year period (mean time to extinction for scenarios in which PE>0.00 ranges from 84.3 – 91.0 years).

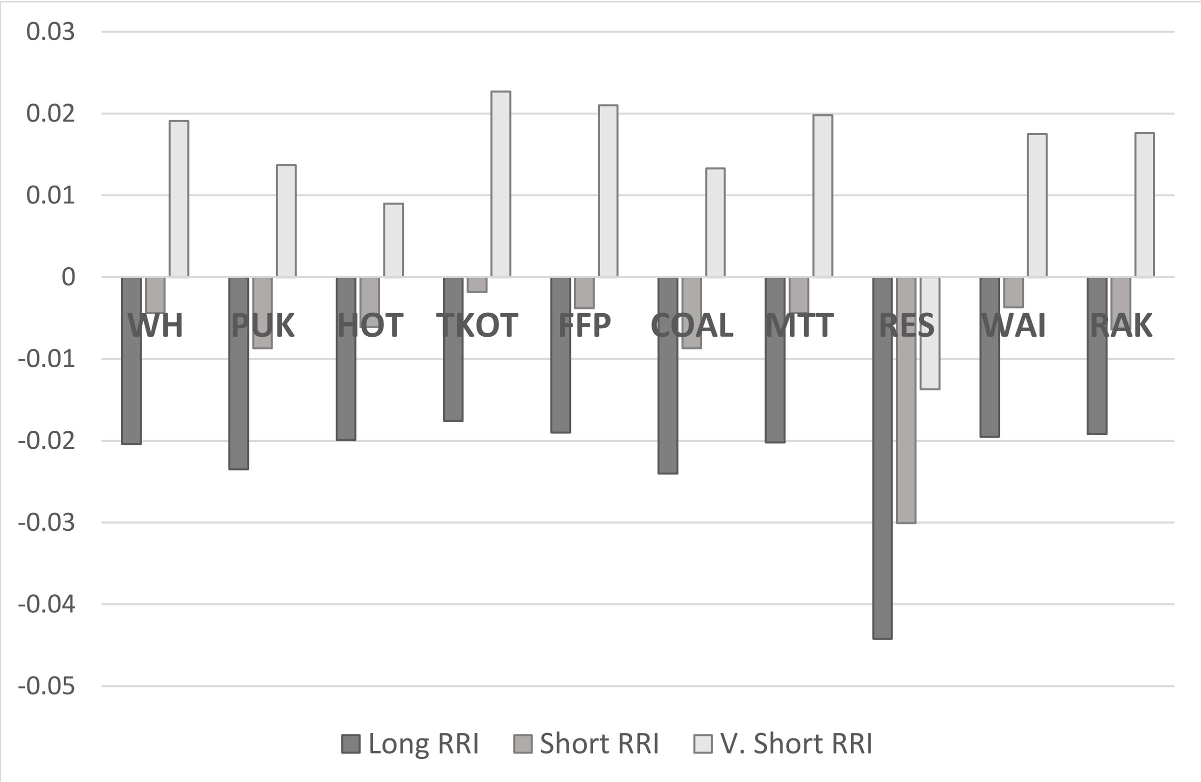


Figure 4.3. Site-specific growth rates (stoc-r) in the presence of local characteristics and risks, at long, short and very short rimu mast-ripening intervals (11, 4 and 2 yrs respectively). WH=Whenua Hou (Codfish Island); PUK=Pukenui (Anchor Island); HOT=Hauturu-o-Toi; TKOT=Te Kākahu-o-Tamatea; FFP=Five Fingers Peninsular; COAL=Coal Island; MTT= Maungatautari; RES= Resolution Island; WAI=Wainuiomata; RAK=Rakiura (Stewart Island)

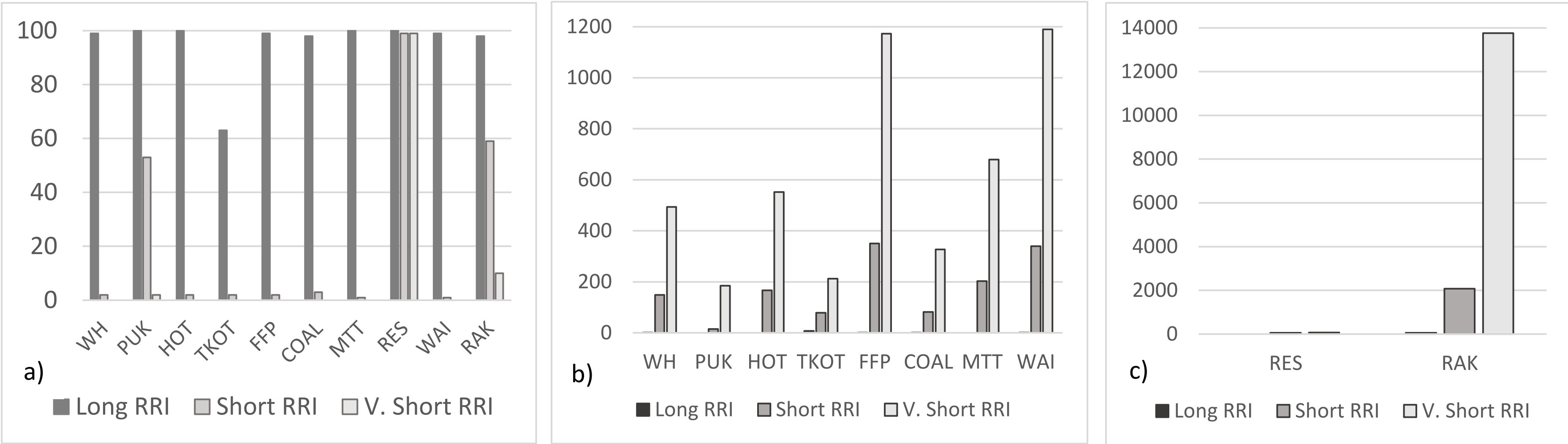
4.4.3 Partial ecological independence

The following section reports results from trials that consider an overall declining population periodically supplemented from an intensively managed sub-population.

4.4.3.1 Supplementation scenarios

Figures 4.4a-e show the results of the partial ecological independence scenarios. **Figure 4.4a** shows the mean number of years (out of 100) in which each site model required supplementation (i.e. site population size at the end of the year fell short of the carrying capacity, noting that all populations were initiated at capacity). Resolution Island required supplementation in either 99 or 100 years (out of 100) in all scenarios and generated no surplus over 100 years. For the other nine sites, the number of years in which supplementation was required varied from: 63-100 for long rimu ripening intervals, 1-59 for short intervals, and 0-10 for very short intervals.

Figures 4.4b-c show the mean total number of birds surplus to capacity generated at each site over the 100 years. Values ranged from c. 0-7 birds for long rimu ripening intervals, to c. 15-350 birds for short intervals, to 212-2076 birds for very short intervals. **Figures 4.4d-e** show the mean total number of birds supplemented at each site over the 100 years. Total mean annual supplementation requirements for the metapopulation range from 232 to 94 to 25 birds each year at long, short and very short intervals respectively.



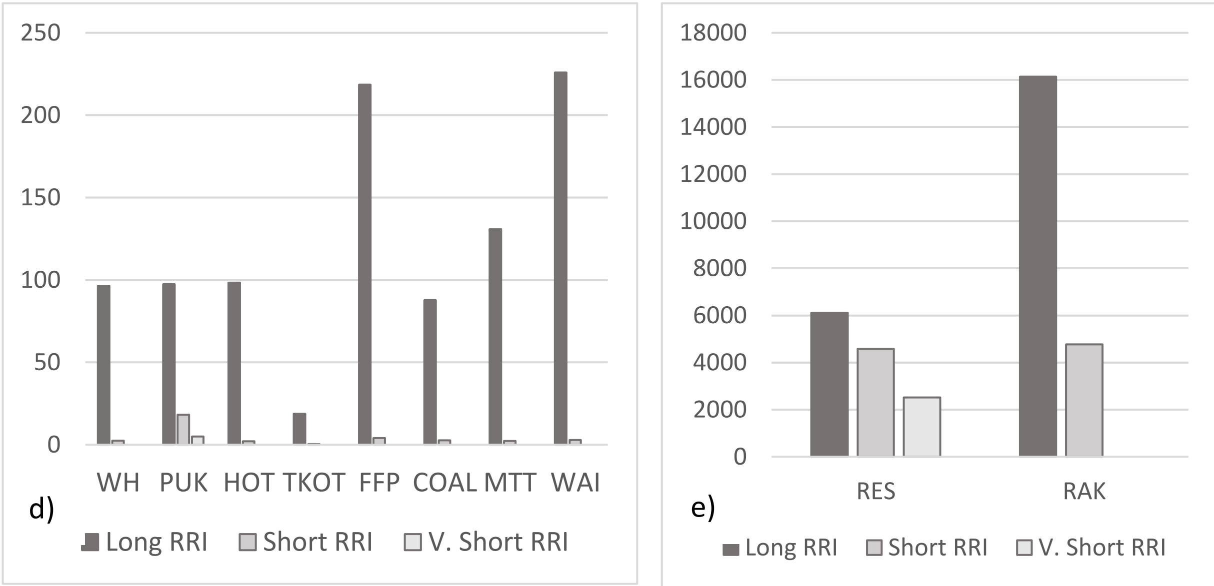


Figure 4.4 a-e. Model outputs for scenarios in which sites begin at full capacity and at the end of each year are either topped up (if below carrying capacity) or the surplus harvested (if above capacity). Long (11yrs), Short (4 yrs) and Very Short (2 yrs) rimu mast-ripening intervals are shown. Graphs show: a) No. of years in 100 requiring supplementation; b) & c) total surplus birds generated over 100 yrs; d) & e) total birds supplemented over 100 yrs. Supplementation/harvest is of birds < 2yrs-old. Site abbreviations as in **Figure 4.3.**

4.4.3.2 Potential harvest rates for intensively managed populations of varied size

The results of harvest rate analyses are shown in **Figure 4.5**. Potential annual harvest rates from an intensively managed population varied from 5.18 birds per year for a population size of 100, to 52.38 birds per year for a population size of 1000.

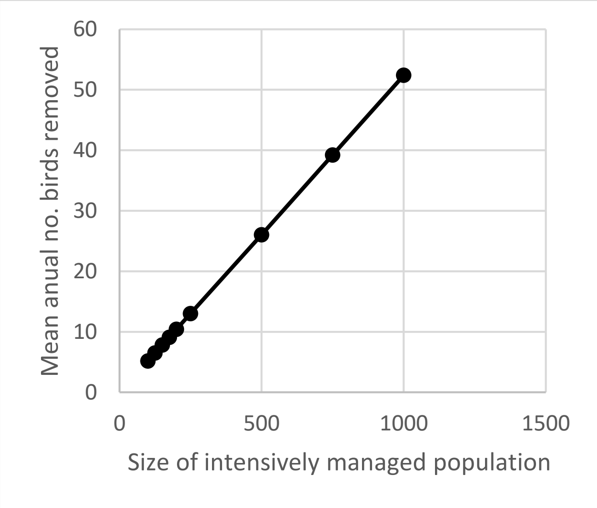


Figure 4.5. Potential number of birds able to be removed from an intensively managed sub-population each year for 100 yrs, without detriment to the population.

4.4.3.3 Implications for population size to 2050

Under the scenarios modelled, projected population size at 2050 ranged from 119 birds (with no management) to 712 birds (with the current level of intensive management) (**Table 4.6**). Starting population size in 2023 was set at 185 birds.

Table 4.6. Results of projecting forwards to 2050 the current population (N=185) using mean growth rates for an unmanaged population, a fully manged population, and two hypothetical "enhanced unmanaged" populations.

Scenario (Ni=185)	Mean growth rate (Stoc-r)	Lambda (λ)	N_{2050}
Baseline: unmanaged except for mitigation of stoat incursions, all birds at sites where RRI=11yrs.	-0.0162	0.9839	119.4
No management, enhanced environment 1: all birds moved to sites where RRI=4yrs.	-0.0004	0.9996	183.0
No management, enhanced environment 2: all birds moved to sites where RRI=4yrs; density improves fertility by 2%.	0.0030	1.0030	200.6
No management, enhanced environment 3: all birds moved to sites where RRI=3yrs.	0.0052	1.0052	212.8
No management, enhanced environment 4: all birds moved to sites where RRI=3yrs; density improves fertility by 4%.	0.0122	1.0123	257.3
Current management: current level of management intensity continues.	0.0500	1.0512	712.3

4.5 Discussion

In this analysis we explored two possible futures for reduced conservation dependence of kākāpō, both of which relied on continued suppression of mammalian predators. One scenario in which the species reaches full ecological independence with sustained growth depended on an abundant food resource suitable to support breeding. An alternative scenario required a smaller, intensively managed sub-population to continually supplement a larger but slowly declining one that is otherwise ecologically independent.

4.5.1 Full ecological independence

In pursuit of plausible pathways to ecological independence, we simulated changes in five factors that could influence population growth in an unmanaged population. Of these, shorter rimu ripening interval had the biggest positive impact on population growth, followed by reduced egg infertility, then increased lifespan, and, finally, increased population size. A reduced severity of inbreeding depression did not have a noticeable impact on population growth rates across the modelled range for the severity of inbreeding depression.

4.5.2 Inbreeding and lifespan

The severity of inbreeding depression and lifespan are uncertain parameters for this species. It may be decades before values can estimated with confidence. Over the period and ranges of values considered, inbreeding severity had no discernible impact on population growth but lifespan extension (from 65 – 100 years) increased growth until it was only slightly negative (stoc-r=-0.0071 up from r=-0.0165). Neither lifespan nor inbreeding severity are open to manipulation by managers in an unmanaged setting and so are not active solutions to conservation dependence. Their inclusion here illustrates the potential value of resolving these areas of uncertainty, which in the case of lifespan is significant. More accuracy around this parameter will provide a better understanding of the species' inherent strengths and weaknesses, improve the predictive value of models and potentially alter management priorities. However, note that in the models we assume no reproductive senescence. If this does not hold true the results of longer lifespan will be milder. It is also worth noting that though changes in inbreeding severity made no observable difference, the models were run for only 100 years which at current parameter estimates is less than three kākāpō generation lengths (T=34.97 years). This would allow for relatively little inbreeding to accumulate on top of that already present the impacts of which, though possibly considerable, are already captured in the models which are built with observed reproduction and mortality rates. Calculation of species-specific lethal equivalents would help improve long-term projections of these effects and ongoing improvements in genomic resources will help with this (e.g. Guhlin et al. 2023).

4.5.3 Small population effects

Small population effects are not discernible once N≥ 500 and are most acute below N=200. The living population of N=185 birds, which exists as small sub-populations of N=9-77 individuals on four islands, is currently managed as a single population with regular inter-island exchanges to optimise population-level genetic and demographic outcomes (D. Eason, pers. comm.). Without this, as shown in the site-specific scenarios where populations are not connected by translocations, inbreeding accumulates more quickly in the smaller site models, causing a steady increase in first year mortality and contributing to local extinction risk. The accumulation is slower on the larger islands because models are initiated at capacity. In practice this would not be possible. As a result, inbreeding accumulation could also be more rapid there than models show and its population-level effects more severe. Note also that reducing the interval between successful breeding events (by selecting sites where rimu ripens more frequently) has the side-effect of shortening species' generation time and, as a result, accelerating inbreeding accumulation. For all these reasons, small sites will always be problematic in any unmanaged scenario and larger sites will be preferable, all other risk factors being equal. Though concentrating birds at one or more of the larger sites currently targeted for release is an opportunity to reduce small population pressures over the longer term (there are four sites with carrying capacities > 200), this should be weighed carefully against the possible disadvantages of increasing disease risk through this increased connectivity (Shoemaker et al. 2014; Parker et al. 2006; White et al. 2015; Alley et al. 2019) and, at some sites, an increased risk of predation (see **Tables 4.1 & S4.1**).

4.5.4 Egg infertility

Unfertilised eggs are an important component of the unusually high hatching failure in kākāpō (Savage et al. 2022) and in the models this makes up 13% of first year mortality. To date, egg infertility is the only component of hatching failure for which a potential pathway to improvement has been identified that does not involve intensive management, hence its inclusion here. Egg fertility rates are higher if females mate multiple times with the same male, and higher still if with different males (Digby et al. 2023). Due to the known mate guarding behaviour of females this is considered more likely where birds live at higher densities, with a female bias (Digby et al. 2023). It seems possible that higher densities of birds may naturally carry this bias due to increased male mortality from aggressive encounters. Therefore, increasing the density of birds may create good conditions for increased egg fertility without intensive management. This hypothesis has not been tested and there is no indication of the size of effect that might be achievable. From the composite scenarios illustrated (see **Table 4.5a-c**) at sites where rimu ripening interval is 4 years, decreasing egg infertility from 13 to 11% of first year mortality could move the population into positive growth. However, in practice, increasing the density of birds can generate negative effects such as increased aggression resulting in egg trampling and damage, affecting mortality in both sexes (J. Beggs, pers.comm.), or increased rate of disease spread (White et al. 2015; Alley et al. 2019), either of which could outweigh any effects on egg fertility.

4.5.5 Rimu mast-ripening interval

Of the five potential pathways to full ecological independence, reducing the rimu ripening interval had most impact, that is, reducing the interval between food abundance events of sufficient size to enable birds to rear chicks successfully. Intervals of less than 4 years led to positive growth in the models. Intervals of 4-5 years, with modest increases in lifespan and/or egg fertility also led to growth.

The long ripening interval on Whenua Hou (11 yrs) is considered an anomaly (Digby, pers. comm.) and intervals of 2-5 years are more often cited in the literature (Norton & Kelly, 1988; Clout 2002; Harper, 2006). However, average intervals clearly vary and those at current and proposed sites other than Whenua Hou are not yet known (KRG pers. comm.).

Whenua Hou was not part of the kākāpō's historic range, which extended much further north. It is considered to have included Rakiura/Stewart Island which occupies a similar latitude, though Powlesland et al. (2006) suggest that this population may also have been moved there by humans. Currently targeted southern sites are therefore at the edge of natural range and may always have been marginal. Reductions in species fitness caused by recent bottlenecks may render these areas untenable for unassisted populations and sites further north, with milder winters and more abundant food may offer greater hope for independence (Lentini et al. 2018).

Food sources other than rimu that carry the same benefits for breeding success have not been confirmed but may exist. In a promising study of coprolites and frozen scats from a period when kākāpō were more widespread, researchers found that southern beeches (Nothofagaceae) dominated kākāpō diets c. 400–1900, in upland habitats (c. >900 m elevation) (Boast et al. 2023). If kākāpō can breed successfully on beech this could open up new areas of habitat where more frequent reproduction could be assured.

The influence of temperature anomalies as cues for synchronized masting suggests that the timing and intensity may be sensitive to global climate change (Schauber et al. 2002). In coming decades this may benefit the species through increased rates of masting for rimu and, potentially, other food species.

Understanding rimu mast-ripening rates at prospective release sites and identifying alternative food sources with similar benefits are already key areas of research for the recovery effort (KRG pers. comm.). Early data for Pukenui indicate intervals of 3-4 years, though there may be other risks there (for example, predation from petrels) that could offset that advantage.

4.5.6 Impact of local risk factors

Positive changes in the five factors explored have less impact on growth at sites where environmental risks are greater than those at Whenua Hou, the risk profile of which was used as the baseline. Site-specific models based on estimated local risk factors (i.e. small carrying capacity, incursion by stoats & rats, or fire) show that where site capacity is larger and risks at the lower end of the range, a rimu ripening interval of four years may be sufficient to prevent population declines. However, where capacity is smaller and risks greater, further compensation is required, for example through shorter ripening intervals and/or a density-induced reduction in egg infertility (if this proves feasible). Despite its large estimated carrying capacity (K=8235), current models suggest that neither of these effects would be sufficient to offset declines on Resolution Island, where estimated risks from stoats are particularly high (see **Tables 4.1 & S4.1**).

4.5.7 Partial ecological independence

The most optimistic changes modelled in the "Full Ecological Independence" scenarios still generate only slow growth which could easily be offset by additional risk factors. Most of the values in the models were generated from Whenua Hou data, where introduced mammalian predator risks are low.

The "Partial Ecological Independence" models explored the feasibility of using a sub-population of intensively managed birds to supplement a larger, free-living one in which individual birds have complete autonomy but the population otherwise declines slowly. The models began by assuming a metapopulation occupying the full capacity potentially available by 2050 (K=15191) and calculated the amount of supplementation needed (i.e. number of birds added) each year, to maintain it at this size. The number of birds required varies considerably depending on whether the rimu mast-ripening interval sits at the long, short, or very short end of the range, but even at the short end the mean number of birds needed each year is N=25, requiring an intensively managed population of around 500 birds. This is more than twice the number currently managed.

The size of the population needed might be able to be reduced by further increasing productivity through increased use of techniques such as double-clutching and artificial insemination (to reduce egg infertility and improve genetic outcomes), though the difference these might make is not yet quantified (Digby et al. 2023). However, as under current intensively managed growth (c. 5% per year) the population may only reach c. 700 birds by 2050, there is no immediate need to resolve the problem of sustaining a population at full targeted capacity.

The supplementation rates calculated here should be considered only a rough guide to the likely scale of requirements for two reasons. Firstly, carrying capacity for the purpose of the models refers to a maximum value beyond which a corrective, additional round of mortality is applied, to maintain numbers at or around a specified limit. The thresholds applied are rough estimates only. In reality, carrying capacity does not operate as a hard line, more gradual, density-dependent impacts on vital rates are expected and any thresholds may vary over time and with fluctuating environmental conditions (Brook & Bradshaw 2006). Secondly, the models apply a precise calculation of the difference between carrying capacity and population size and react with an annual correction where needed. For a large and widely dispersed metapopulation, precise annual counts coupled to annual inter-site translocations might not be efficient. Managers might instead opt for periodic corrective action, or to remove altogether any small or otherwise volatile sites from the program once larger, more stable sites are operational. And finally, all site-specific models are based on limited knowledge of the magnitude of risks at those sites. As minimum viable population size is predicted more reliably by environmental factors than by species biology (Brook et al. 2006), this is an important information gap.

4.5.8 Limitations of PVA

Population viability analyses will not give absolute and accurate "answers" for what the future will bring for a given wildlife population or program of management. This limitation arises partly from our inevitably incomplete knowledge of the complex systems we are aiming to model, and partly from our inability to identify and quantify accurately the future influences on those systems, either natural or human mediated. Consequently, many researchers have cautioned against the exclusive use of PVA results in promoting specific management actions for wildlife populations (e.g. Beissinger and McCullough 2002; Reed et al. 2003; Lacy 2019) and this advice is reiterated here. However, for threatened species PVA remains a valuable tool alongside others, for testing ideas and supporting planning and management decisions within an adaptive framework. As shown here it is particularly valuable for small or highly fragmented populations where the combination of stochastic and deterministic risks can mask or obscure key drivers of growth.

4.6 Conclusions

As currently constructed, models indicate the kākāpō population will not easily outgrow conservation dependence. The most promising pathway to long-term independence would be through the preferential allocation of birds to sites where:

- the interval of rimu mast-ripening events (or the availability of an equivalent natural food source that supports successful breeding) is less than 4 years;
- risk of loss to stoats is low or can be adequately managed;

- the site topography or location of nesting habitat encourages relatively high density living and, as a result, lower rates of egg infertility (assuming the costs and benefits of this have been firmly established); and
- carrying capacity is sufficient to support a mean population size of ≥200 and ideally ≥500 individuals.

Where this cannot be achieved, a slowly declining population can be sustained through supplementation but this would be highly resource intensive.

Resolving uncertainty around rimu masting phenology and how this varies geographically, alongside whether kākāpō can successfully breed successfully on an alternative diet, are likely to provide the biggest gains towards understanding the potential for conservation independence in this species.

CHAPTER 5. POPULATION AND HABITAT VIABILITY ASSESSMENT: A ONE PLAN APPROACH TO SAVING THE DEVIL

5.1 ABSTRACT

In 2003, as a precautionary response to the recent emergence of Devil Facial Tumour Disease (DFTD), the Zoo and Aquarium Association Australasia (ZAA) (formerly the Australasian Regional Association of Zoos and Aquaria - ARAZPA) established a captive insurance population for the Tasmanian devil, *Sarcophilus harrisii*, on mainland Australia, in partnership with the Tasmanian Government's Save the Tasmanian Devil Program (STDP). By 2007, based on population models, the prognosis for the species had deteriorated to one of possible wild extinction (McCallum et al. 2007) with an estimated timeframe of 30 years. At this point two things became clear: firstly, that the insurance population was the only conservation strategy in place potentially able to prevent extinction and enable recovery; and secondly, that if called on to do so, the existing population would be inadequate and a larger, more ambitious one, would be needed. A concept for this "insurance metapopulation" was drafted and agreed upon by the Australian and Tasmanian governments and the Australasian Zoo community. However, its deployment posed many challenges, including its unprecedented scale and complexity, the uncertainty then surrounding DFTD management, and the experimental nature of some aspects of its design. Further, expansion of the *ex situ* response was not universally popular, and accurate and consistent communication on the issue was hindered by the large and devolved structure of the wider devil conservation response, and its significant media profile. To overcome these problems a workshop was held in Hobart in July 2008, jointly hosted by the STDP and Taronga Conservation Society Australia. This workshop was designed and facilitated by the IUCN Species Survival Commission's Conservation Planning Specialist Group (CPSG) (formerly the Conservation Breeding Specialist Group – CBSG). This chapter describes the 2008 workshop, its main outcomes and the role it played in shaping the Tasmanian devil insurance metapopulation.

5.2 INSURANCE METAPOPULATION CONCEPT

The concept for the insurance metapopulation was based on the agreed goals of the STDP's Tasmanian devil insurance strategy, which were, "to establish and maintain a healthy, viable metapopulation of Tasmanian devils for 50 years, that would:

- remain DFTD-free;
- be genetically representative of the species;
- be able to sustain a harvest of animals for release to the wild;
- maintain a suite of associated flora and fauna (commensal, symbiotic and parasitic); and
- maintain wild behaviours."

The 50-year timeframe allowed for extinction after 30 years, 10 years of active re-establishment (based on Beck et al. 1994) and a 10-year precautionary buffer.

The broad concept was to achieve the goals using a combination of three management systems: *Captive*: intensively-managed, zoo-style facilities; *Free Range Enclosures* (FREs): less intensively-managed groups of devils in larger enclosures, managed to enable higher densities than observed in the wild; and *Wild Islands:* sites within Tasmania able to be protected from DFTD (such as actual islands or "virtual islands" created on mainland Tasmania by suitable barriers, **Figure 5.1**).

The rationale for this was that a sufficiently large metapopulation held entirely in intensive captive facilities would be achievable using well-established techniques and would be genetically efficient, productive and well protected from DFTD. However, it would be costly, would not retain devil-associated flora and fauna, and would be at risk to both genetic and behavioural adaptation to captivity. Holding a proportion of the metapopulation under wild conditions was considered essential to compensate for these deficiencies but was also expected to be costly and possible only on a small scale. An intermediate form of management allowing large numbers of devils to be maintained under relatively natural conditions and at a reduced cost, was proposed as an alternative but at the time was untested, existing only in theory. Pursuing an integrated combination of all three systems seemed at the time to offer the best chance of meeting all insurance metapopulation goals, while also supporting the broader STDP goal of maintaining the devil as an ecologically functioning component of the Tasmanian landscape.

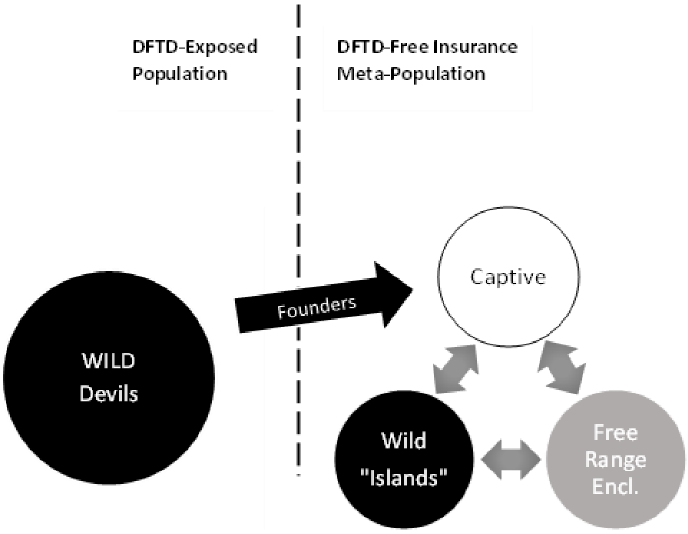


Fig. 5.1: Illustration of metapopulation concept (arrows indicate animal movement) with a tabulated summary of relative strengths and weaknesses of each component. Comparisons are based on assumptions made at the time of the PHVA and gene diversity retention comparisons are for populations of equal size.

Insurance population goals	Wild "Islands": DFTD-free islands and fenced peninsulas, wild conditions, natural densities, genetic exchange only **UNTESTED, AVAILABLE ≥ 5 YRS**	Free Range: large enclosures, high densities and less intensive demographic & genetic mgmnt. **UNTESTED, AVAILABLE > 3 YRS**	Captive: intensive management in traditional zoo-style enclosures. **WELL-TESTED, IMMEDIATELY AVAILABLE**
Secure from DFTD	Partly achieved	Achieved	Achieved
High Gene Diversity Retention	Not achieved (N too small)	Partly achieved	Achieved
Harvest for wild release	Achieved	Achieved	Achieved
Associated Flora and Fauna	Achieved	Partly achieved	Not achieved
Maintain all wild behaviours	Achieved	Partly achieved	Not achieved

The metapopulation concept included provisions for ensuring that the genetic goal was met. Following Marshall & Brown (1975), the capture of at least 150 DFTD-free wild devils was proposed, from across the species' range, to secure a representative founder population (based on the assumption that a random sample of 149 individuals should confer a 95% chance of securing alleles occurring at a frequency of at least 1%). Though an unusually large harvest for a managed program, it was considered appropriate given the wild prognosis.

To assist with retention of this genetic profile through time, rapid amplification of this founder base, with as little genetic distortion as possible, was proposed. Target sizes were developed to meet the aim of achieving mutation-drift balance (Franklin, 1980). Recognising the likely difference in genetic efficiency between captive and wild metapopulation components effective to actual size ratios of 0.1 (wild) and 0.3 (captive) were assumed based on Frankham, (1995) and Frankham et al. (2002). This resulted in a target size of 1,500 captive, or 5,000 wild-living devils, or some interim combination of these, and other systems, able to deliver a genetically effective population size (Ne) of 500. This assumed no further supplementation from the wild which was the advice at the time. Post-PHVA, safe supplementation with quarantined first-year devils from affected populations became possible allowing genetic goals to be met at a smaller population size.

Finally, the concept assumed that using existing tools and internationally recognised standards for intensive population management, breeding rates and transfers would be coordinated within, and among, metapopulation components to control numbers, retain genetic diversity, slow the accumulation of inbreeding, manage disease risk and when needed, generate appropriate numbers of suitable release animals.

Turning this concept into an achievable and affordable program was the challenge posed to PHVA workshop participants.

5.3 THE CONSERVATION PLANNING SPECIALIST GROUP

The Conservation Planning Specialist Group (CPSG) is a discipline-based network housed within the Species Survival Commission (SSC) of the International Union for Conservation of Nature (IUCN). CPSG's primary role is to support the conservation of species by helping groups to plan effective action collaboratively. Its workshops provide an objective environment, expert knowledge and independent group facilitation, and are designed to address problems systematically using sound science.

For the past 30 years, its signature planning process has been the Population and Habitat Viability Assessment (PHVA). PHVA uses population simulation models to support situation analysis and consensus decision-making towards action. CPSG has also developed a workshop-based process that enables groups to collaborate on Disease Risk Analyses (DRAs) (Jakob-Hoff et al., 2014); that is, to identify the disease risks relevant to particular wildlife management situations and build consensus on appropriate mitigation.

Further to this, CPSG's longstanding relationship with the global zoo community has helped shape its One Plan Approach philosophy (Byers et al., 2013) which, "…*considers all populations of the species, inside and outside its natural range, under all conditions of management, engaging all responsible parties and all available resources from the start of any species conservation planning initiative*". The One Plan Approach aims to maximise the resources available to a project and the efficiency with which they are applied, and at the same time to generate support among key stakeholders for the conservation directions agreed. Though formally defined only in 2012 these principles have long underpinned the design and delivery of CPSG workshops.

Six months ahead of the PHVA workshop a team was assembled, comprising two CPSG program officers, staff from the ZAA office, Taronga and Auckland Zoos, and the Tasmanian Department of Primary Industries, Parks, Water and Environment (DPIPWE) (formerly the Tasmanian Department of Primary Industry and Water – DPIW). In the lead-up to the PHVA, this team worked to identify and enlist key stakeholders and subject experts, to compile relevant data and design a suitable workshop process. Three baseline models were developed using the *VORTEX* population simulation application (Lacy and Pollak, 2014). One was based on DFTD-free wild population data, one on studbook data from the zoo-based population, and the third was built to reflect the intermediate characteristics expected of devil populations housed in free-range enclosures. These were used to support the deliberations of working groups, which are described in more detail below.

5.4 THE PHVA WORKSHOP

In July 2008, 40 stakeholders and subject matter experts from 19 institutions, including zoos and wildlife parks, government agencies, and the academic and non-profit sectors, gathered in Hobart, Tasmania for four days of intensive discussion. The aim was to build a shared understanding of the new insurance metapopulation concept and, using the expertise and experience present, to explore, evaluate and prioritise options for deployment and management.

Following a series of scene-setting presentations participants formed working groups around the three major management themes (captive, free range enclosures and wild islands) and disease risk management. A fifth "metapopulation integration" group was convened towards the end of the workshop, tasked with ensuring adequate integration of metapopulation components. Recommendations from this group are included here within the sections they were directly relevant to.

5.4.1 The Captive Management Working Group

The **Captive Management Working Group** was tasked with designing a plan firstly to support the total insurance metapopulation for the first three years, until FREs or "islands" could be mobilised, and secondly to continue supporting a component of the metapopulation beyond that three-year period.

The 2005 Captive Management Plan (CMP) and Husbandry Guidelines. The CMP had established protocols for effective management of the mainland captive insurance population, including provision for management of demographics, genetics, program administration and governance. A Memorandum of Understanding was in place between DPIPWE, ZAA and participating ZAA zoos, which identified the responsibilities of each party towards the program. Transfer and breeding recommendations had been prepared for the 2007 – 2008 season. A husbandry training course at Trowunna Wildlife Park was running annually, husbandry guidelines and a husbandry-centric email discussion group were in place. A brief review of these generated topics for discussion.

Capacity. Models indicated that supporting the insurance metapopulation for the first three years would require 400 devil spaces. By June 2008, there were 108 adult devils on the mainland and 150 committed spaces. In Tasmania, there were 63 devils in government quarantine facilities, a similar number in Tasmanian wildlife parks and an estimated capacity commitment of 100 spaces. This left a capacity shortfall of 150 – 250 devils. Discussions had begun with sister zoo associations in Europe and the USA to meet this shortfall but for political reasons this was put on hold prior to the PHVA, and discussion was restricted to Australian solutions. A funding workshop was prioritised to explore how this shortfall might be resourced.

Senescent devils. In a zoo environment, devils live longer than in the wild. Up to 20% of program spaces can be taken by senescent animals. Removing these offers an opportunity to increase breeding program capacity. Euthanasia had been ruled out. Options discussed included: holding outside the program (in non-member mainland fauna parks or overseas); and use in behavioural research and release trials.

Founders: Maximising founder base was a priority. The speed at which the disease was spreading across the island, the incidence of local extinction and the period of DFTD latency were not well understood. It was assumed that founders could be collected only from areas ahead of the disease front, making representative sampling across the range of the species an urgent priority, along with the movement into breeding situations of founders already captured.

Evolutionarily Significant Units: The location of founder devils was known and plotted, with genetic profiles of some individuals mapped. The CMP prescribed a single management unit for devils, comprising two Evolutionarily Significant Units (ESUs). To ensure representation of a 1:1 ratio of eastern and western founders both were targeted for future sampling.

Selection for resistance. A small group of experts convened briefly during the PHVA to discuss the option of actively selecting for resistance in the captive program should resistant devils be identified. The group had agreed that given the knowledge of disease resistance mechanisms at that time, it would be unwise to select narrowly for it at the risk of losing genome-wide diversity. Instead, and recognising that any emerging resistance in the wild would be aggressively selected for, resulting in a further genetic bottleneck for the species, it was agreed that captive management should respond to such an event by increasing investment in the maintenance of vulnerable genotypes, as insurance against future non-DFTD disease-related threats.

Triggers for gearing up and winding down: Recommended triggers for <u>expanding</u> the population beyond 400 included: delayed or failed deployment of FREs and “Islands”; lower than expected genetically effective to actual population size (Ne/N) ratio; new research prescribing higher census or genetically effective sizes; or initiation of a global plan. Recommended triggers for <u>reducing</u> the intensive commitment included: successful replacement by “islands” or FREs; decline in disease prevalence; or insufficient funding to maintain the program.

Agreed priorities were to:

- ensure DFTD-free founders recently captured from the wild for the insurance population were placed in breeding situations in time for the 2009 season;
- convene a costing workshop to discuss ZAA and DPIPWE captive capacity expansion and the establishment of FREs;
- development of management options for senescent animals to optimise space in intensively managed facilities.

5.4.2 The Free Range Enclosures (FREs) Working Group

The **FREs Working Group** was tasked with designing low-cost options for the maintenance of 1,500-5,000 DFTD-free, genetically and phenotypically diverse devils, in controlled and ecologically relevant environments.

Two schemes were discussed: 'fenced islands' scattered across Tasmania, each containing up to 40 animals; and a 'complex' on mainland Australia comprising smaller, modular pens, each holding four adult males and four adult females. In both models, breeding adults selected their mates at random each year from those present in their enclosure, with inbreeding accumulation slowed through an inter-enclosure Maximal Avoidance of Inbreeding strategy (Princée 1995). Issues covered by the group are summarised below.

Capacity. The managed environment in fenced enclosures (for example, supplementary feeding and veterinary intervention) was expected to increase longevity above that expected in the wild, prompting discussions similar to those of the captive management group around housing senescent animals. Dedicated enclosures were proposed, similar to those for crèching young prior to breeding placement.

Research. The working group recognised the experimental nature of FREs, acknowledging that a cycle of establishing, testing and adapting strategies would be the most effective and efficient way to pursue program goals. Research priorities included:

- animal welfare: e.g. understanding how to ensure resource saturation to minimise conflict;

- life history strategies: e.g. the impact of sperm storage and potential male domination of multiple females;
- genetic management: strategies for tracking paternity and encouraging equalised founder representation;
- nutrition: a better understanding of baseline nutrient requirements, to avoid overweight animals and nutritional disease.

This led to the development of a significant research program with important insights for program management.

Since the PHVA, technology has solved three of the issues discussed at the PHVA, in the areas of reproductive management, monitoring, and funding. Firstly, it has proved quick and relatively inexpensive to establish paternity through molecular markers such that accurate pedigrees can be maintained and used to apply management by mean kinship (Ballou and Lacy, 1995), at least in the smaller enclosures. Secondly, the development of remote camera monitoring has meant that animals can be sighted and evaluated for injury once a day, a key criterion for facility licencing and resource management. And thirdly, funding, initially considered a problem because by their very nature FREs are remote and not interactive, has to some extent been resolved. The evolution of web-based crowd funding means that the public can be reached remotely.

Participants recognized that FREs were a largely untested option, particularly with respect to the impact on devil dynamics of increased density. However, with cost potentially placing an upper limit on intensively managed captive facilities, and a range of social, political and logistical difficulties preventing "islands" from short-term implementation, it was also acknowledged that FREs offered the most promising means by which insurance population capacity could be achieved within the time-frame required.

Agreed priorities were:

- securing STDP Steering Committee endorsement of FREs as an integral component of the insurance population and their prioritisation for funding;
- an implementation committee to ensure a strong relationship between facility managers and researchers for optimisation of this novel strategy;

- to expedite the establishment and expansion of free range enclosures in Tasmania and mainland Australia.

5.4.3 The Islands Working Group

The overriding purpose of the STDP was to maintain the devil as an ecologically functional species in the wild. The **Islands Working Group** was tasked with assessing the potential for maintaining DFTD-free wild populations in Tasmania to assist maintenance of ecological functionality, including conservation of parasitic communities, wild behaviour and adaptations, and the Tasmanian devil's intrinsic, social, economic and political value.

The working group limited consideration to Tasmania because of the political importance of maintaining Tasmanian devils within the State, and the legislative and impact assessment challenges associated with introducing devils into other jurisdictions.

Candidates for consideration included devil-free offshore islands to which the species could be introduced, and "virtual islands" able to be created on the Tasmanian mainland by fencing peninsulas or other wild areas already supporting devils.

The working group selected several offshore islands as potential translocation sites, focusing on larger islands with greater estimated devil capacity such as King (≈1,500), Maria (≈80-120) and Bruny (≈300) islands. Maximising population size would minimise the need for management though would increase monitoring difficulty and reversibility. Several peninsulas and habitat 'islands' were also identified as potential sites provided they could be sufficiently fenced and secured. Examples included DFTD-free sites such as Woolnorth (≈250) and Cape Sorell (≈100 or 400-600 depending on fence location), and DFTD-affected sites such as the Freycinet (≈100-130 devils) and Tasman-Forestier (≈350) Peninsulas. Ten potential sites were assessed and prioritized with respect to: disease status, carrying capacity for devils, feasibility, timelines, costs, required resource management (food and water), population management intensity, biosecurity risk, ecological risks, and stakeholder issues.

Population modelling was used to explore the projected viability of populations of varying size (25 to 1,500 devils) at these sites. Modelled populations with fewer than 100 devils showed poor viability. Those with at least 250 devils showed positive growth and little extinction risk over 100 years but lost substantial genetic variation over time. Large populations of 500 or more devils are required for long-term (i.e. 100 years) demographic and genetic health if isolated. Higher (but realistic) impacts of catastrophic events and inbreeding severity result in the need for 1,000 devils or more for long-term viability. Models were also used to explore the viability of managing multiple interacting small sites (e.g., mining sites) with periodic translocations or supplementation. Models predicted poor viability under the management scenarios explored.

In addition to small population size vulnerabilities, introductions to offshore islands posed challenges associated with potential negative impacts to other threatened species, either those affected directly by introduced devils into new areas, or due to the loss of these habitats as potential release sites for other, non-compatible threatened species. The group recommended further assessments and weighing of environmental and social risks against potential benefits.

Virtual islands have their own problems: in DFTD areas they have to be biosecure and may require active eradication of devils inside the fence line. Despite this drawback, a fencing feasibility study was recommended for priority “virtual islands” such as Woolnorth, Cape Sorell, Freycinet and Robbins Island.

The group agreed that protecting populations of disease-free devils in the wild, within fenced peninsulas and other suitable areas in Tasmania, was of considerable value to the overall recovery effort.

Agreed priorities:

- undertake a feasibility study to assess the ecological, financial and community impact of (i) fencing Woolnorth in time to prevent disease exposure of resident devils and (ii) releasing devils on an offshore island;
- establish and fill a project officer position to realise fencing proposals for priority wild areas in Tasmania.

5.4.4 The Disease Management Working Group

The **Disease Management Working Group** was tasked with comprehensively assessing DFTD and other disease risks associated with moving devils, to establish and then maintain the insurance population, and second with developing a disease risk management plan.

This group conducted a qualitative disease risk analysis (DRA) following the systematic, evidence method applied by the World Organisation for Animal Health (OIE), developed from the methodology of Covello and Merkhofer (1993) and subsequently adapted for wildlife (Jakob-Hoff et al. 2014). The group comprised nine individuals with credentials in the specialist fields of wildlife disease, cancer genetics, veterinary pathology and immunology. Significantly, the group included Tasmania's Chief Veterinary Officer whose input and support was critical to the implementation of disease risk mitigation recommendations.

As data on wildlife disease are scarce, compared to disease of people and domestic animals, risk analyses typically involve greater uncertainty. One method of reducing this is to elicit expert opinion from a group such as this. A DRA aims to, "*identify diseases that may enter a specified animal population, identify the likelihood of such introductions, assess their consequences and identify measures that may be applied to mitigate either the likelihood of introduction or the magnitude of consequences*." Information gaps are identified and provide a basis for prioritizing research effort that will reduce the level of uncertainty associated with the analysis (Jakob-Hoff et al. 2014).

Previously, members of the group had been provided with a comprehensive review of relevant published, and unpublished, information to which they added further comments during the facilitated discussion. The analysis responded to the STDP's metapopulation concept, which involved establishing captive and semi-captive populations of devils in Tasmania and on the Australian mainland. Key DRA questions were:

1. Which potentially pathogenic organisms need to be actively managed or maintained and what are the most effective disease mitigation options?
2. What are the biosecurity risks to Tasmania associated with movement of devils from mainland Australia back to Tasmania as part of the meta-population management of this species?
3. What is the likelihood and consequence of DFTD affecting other species?

Following the problem description step, a graphic representation of all potential movements of Tasmanian devils within Tasmania and between Tasmania, its offshore islands and the Australian Mainland was developed (**Figure 5.2**). All documented infectious and non-infectious disease hazards of devils were reviewed and prioritized applying a semiquantitative estimate of the likelihood of exposure to a range of disease (high for unmanaged, unmonitored free-living populations, medium for more closely monitored populations in large areas of fenced habitat (exclosures), and lowest in the controlled environment of full captivity. However, extra weighting was applied to the captive setting in recognition of increased disease susceptibilities associated with high density and stress. Using this rough assessment of relative risk, prioritized critical control points (CCP) for risk mitigation actions were identified on the movements diagram (**Figure 5.2**).

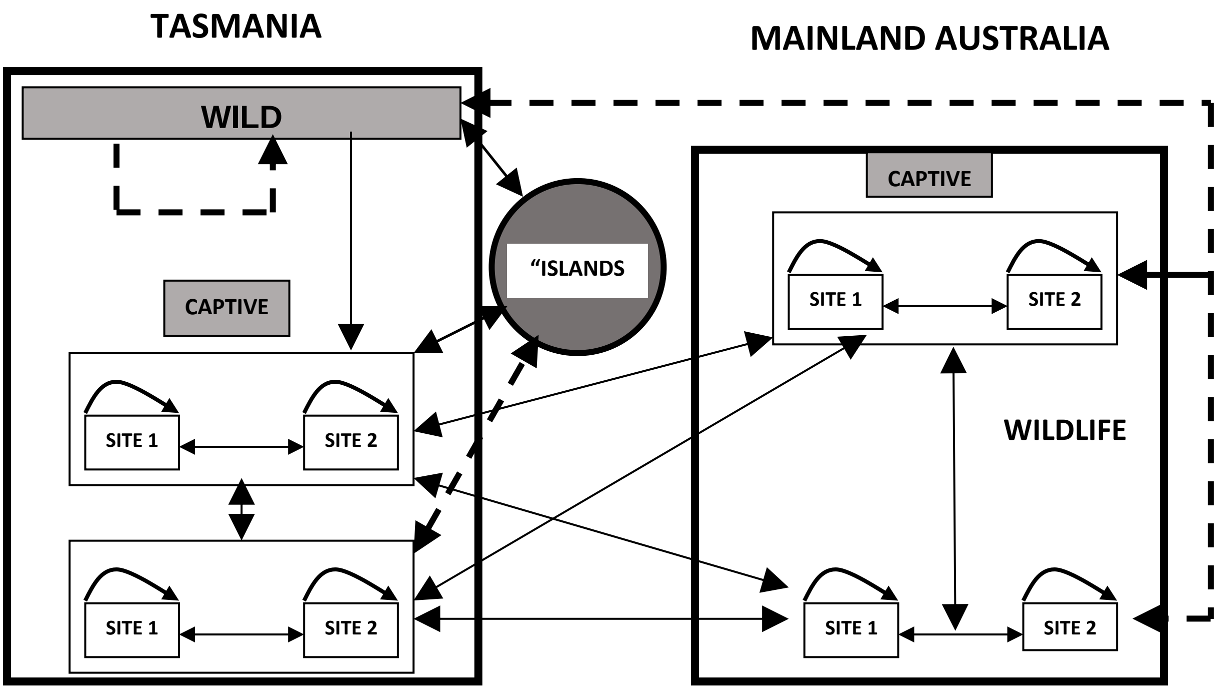


Figure 5.2 Potential movements of Tasmanian devils associated with active metapopulation management. Dashed arrows indicate Critical Control Points for the movements considered to involve the highest risk of disease transfer (from CBSG 2008).

Of the 24 disease hazards assessed, seven were identified for active risk mitigation as part of metapopulation management: DFTD, ectoparasites, salmonellosis, lymphoproliferative diseases, pseudotrichinosis, young age onset neoplasia and other neoplasia. Detailed risk mitigation strategies were developed for each translocation option (wild to captive, captive to captive, wild to wild, wild to island and exclosure to wild). A further option, not available at the time but considered possible in the future, was the movement of devils outside Australia. A preliminary disease risk identification and mitigation plan was developed for this also. Communications and disease risk mitigation implementation plans were developed, incorporating feedback from the wider workshop.

Agreed priorities:

- review current disease management protocols and risk categorisation in light of the DRA;
- develop biosecurity guidelines for responding to: 1) the detection of DFTD in an insurance population; 2) known incursion of a wild devil into a captive population; and 3) diagnosis of another significant (non-DFTD) disease in a devil population.

5.5 CONCLUSIONS

The role of the PHVA workshop was to review the metapopulation concept with key stakeholders, assess the feasibility of its component parts, reach agreement on contentious issues and, from the result, build a plan of action and broad support for deployment. In the years since, many activities have been pursued that were conceived during the workshop. There is room to mention only a few here but for a full account see the PHVA Report (CBSG, 2008).

Within months of the workshop, disease risk management protocols were revised, optimal transfers of new founders had been implemented and a costing workshop for captive facility expansion held and reported on. CPSG was further engaged by DPIPWE to turn PHVA recommendations into a succinct strategy for management and governance of the developing metapopulation (CBSG/DPIPWE/ARAZPA, 2009) and based on this a Metapopulation Advisory Committee (MAC) was set in place to guide and enable metapopulation deployment. Its first role was to disperse government seed grants to institutions willing to assist in metapopulation expansion, and particularly to those piloting FRE systems. A research group reporting to the MAC directs interested researchers to priority metapopulation topics. More recently, a small population was established on Maria Island and the fencing of, and eradication of diseased devils from, the Tasman-Forestier Peninsula allowed the establishment of a DFTD-free population there. Proposals to establish free-living populations on mainland Australia have also been proposed but are not currently being pursued. Trials of FRE systems have proved highly successful. For Devil Ark on mainland Australia, the largest FRE project (comprising smaller, modular pens, each holding four adult males and four adult females; now termed Managed Environmental Enclosures), annual costs per devil are roughly one-fifth of those calculated for other systems (AUD$2,200 vs AUD$10,000 per devil per year (Izzard et al.

2019). A DPIPWE-commissioned evaluation in 2012 showed the insurance metapopulation program was on-track to meet, or exceed, all 50-year program goals (Lees and Andrew 2012). A decade later, the insurance metapopulation remains relevant to devil recovery. The current large, geographically dispersed, genetically diverse and DFTD-free metapopulation is a resilient and flexible resource that keeps options open for managers in the face of this ongoing extinction crisis. Its success cannot be attributed to any single factor, other than perhaps to the extraordinary passion and commitment of those engaged in devil recovery, both within and outside Tasmania. However, it is reasonable to conclude that the PHVA played an important role in catalysing and shaping metapopulation implementation at a critical time in its development, demonstrating the power of the PHVA planning method and the One Plan Approach that sits behind it, in galvanising diverse, multi-disciplinary communities and their resources behind a single, coherent plan of action.

Chapter 5. *Population and Habitat Viability Assessment: a One Plan Approach to Saving the Devil*, was initially published as a book chapter and adheres to the brief and word limit for that publication. **APPENDIX C.** provides unpublished, supplementary material that provides a more detailed account of workshop preparation and delivery and increases the relevance of this Chapter to the thesis question, "*What can applying recommended principles and tools look like in reality?*".

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CHAPTER 6. GENERAL DISCUSSION

This thesis consolidates existing knowledge on species conservation planning, outlining essential elements for a national or global framework capable of generating plans on a large-scale. It provides evidence that species conservation planning can provide positive outcomes for species. Additionally, it underscores the key role of Population Viability Analysis (PVA) in supporting planning efforts for highly threatened species, highlights the value and importance of stakeholder participation and documents an example of the integration of these elements within a single planning project.

6.4 THESIS SYNOPSIS

Biodiversity is in precipitous decline with current rates of extinction set to increase (WWF et al. 2020; IPBES, 2019; Monroe et al. 2019). The need for urgent action is highlighted by the Kunming-Montreal Post-2020 Global Biodiversity Framework (CBD 2022) which prioritises protecting areas for nature, addressing major threats and the sustainable use of biodiversity. Recognising that these measures are often too slow or inadequate for species already threatened with extinction, Target 4 of the framework urges more intensive, immediate *in situ* and sometimes *ex situ* action. Species conservation planning has a special role to play in preventing many pending extinctions by ensuring that the action taken for threatened species is well-targeted, efficient and effective (Mace et al. 2018; Byers et al. 2013).

6.4.1 CHAPTER 1. THE CASE FOR SPECIES CONSERVATION PLANNING.

I explored how species conservation planning fits into broader biodiversity conservation planning approaches, evaluating its current provision and how much of it is needed. I conducted a literature review and analysis of publicly available databases storing plans or information about them.

I found that while measures focused on area-based protection and management, and on systemic threat reduction are critical, their implementation often falls short, leaving many species at risk of extinction (Foose et al. 1995; Groves and Game 2016; Skerratt et al. 2007; Sutherland 2001). Species conservation planning is used to help determine how best to address these gaps. The global estimate for the number of threatened species potentially requiring this kind of attention (≈42,100 (IUCN 2023)) may be an underestimate as the world's conservation assessments reveal taxonomic and geographic biases (Troudet et al. 2017; Drinan et al. 2020; IUCN 2023a). Fewer than 5,000 operational plans exist globally (**Table 2.2**) and substantial shortfalls in species conservation plans are reported (Rossi et al. 1995; Bolam et al. 2023; Heywood et al. 2019). My analysis suggests many thousands more plans are required to adequately cover species at risk.

6.4.2 Chapter 2. Species conservation planning approaches and lessons to date

Globally, the gap between the species that need targeted planning and those receiving it is large. I aimed to understand why that gap exists. Explicitly, whether because of inadequate information for species planning or because national or global frameworks are not designed for the challenge of planning at the scale required.

To investigate, I collated and reviewed planning guidance literature from government agencies, NGOs and academic studies, identifying commonalities and differences. I summarised collective advice, which identifies the following core elements of good species conservation plans: i) a definition of long-term successful conservation of the species; ii) a review of relevant information; iii) an analysis of threats to the species and obstacles to addressing them; iv) clear objectives; and v) details of the actions required to achieve these objectives; vi) a description of how the plan will be implemented, monitored and adapted. There's consensus on these elements, but divergence on methods such as the use of population viability analysis and approaches to stakeholder engagement.

I found current evidence of 10 national frameworks systematically generating and implementing species conservation plans (Gaywood et al. 2016; Rossi et al, 2016; Baptista 2019; NMFS 2020; Yang et al. 2020; Kraus et al. 2021). I found further evidence of past frameworks now discontinued (Machado 1997; Rossi et al. 2016). Elements found in more than one of the frameworks identified were: i) A method to identify which species should be planned for; ii) guidance on how to develop plans for the local context; iii) a mechanism for streamlining species conservation planning to reduce duplication and capitalise on existing plans or initiatives; iv) a funding mechanism for plan implementation; v) coordinating bodies to implement, monitor and adapt plans; vi) a program of public awareness raising; vii) a central, curated record of plans; viii) standardised reporting across plans for review and improvement. However, many countries lack such frameworks, instead relying only on national red lists (113 countries, ZSL and IUCN National Red List Working Group 2022) and occasional, *ad hoc* development of plans (Heywood et al. 2019).

In conclusion, there is broad agreement across governments and non-government sectors on the required written content of species conservation plans. However, ambiguity remains regarding development methods such as population viability analysis and how best to involve stakeholders. While there are promising models for planning at scale, they are currently active in too few countries to fill the planning gap.

6.4.3 Chapter 3. Science-based, stakeholder-inclusive and participatory conservation planning helps reverse the decline of threatened species

There is good convergence on how to plan well and on what is needed to do this at scale. More investment is needed to close the gap between species that need plans and those that are planned for. To appeal to investors, evidence is needed that planning has a positive influence on outcomes for species.

I reviewed available studies of the impact of planning on species conservation status and found conflicting conclusions. Although studies report that planning led to improved status of endangered species in the USA (Schultz & Gerber 2002; Taylor et al. 2005), a further study showed it to be detrimental if not adequately resourced (Ferraro et al. 2007) and an Australian study showed no effect once biases associated with prioritising species for planning were accounted for (Bottrill et al. 2011).

Prior to this research, impact studies had focused mainly on plans developed by national governments. We used a different approach, sourcing 35 plans developed across 23 different countries from a database held by the International Union for the Conservation of Nature's Species Survival Commission (IUCN SSC). These plans used an integrative, multi-stakeholder approach including population viability analyses and considered both *in situ* and *ex situ* management.

To assess the impact of planning, we matched the species with their histories of IUCN Red List assessments of extinction risk. We used the Red List Index and a counterfactual approach, comparing the overall predicted extinction trend without planning with the observed trend after planning. Post-planning, threatened species declines continued, but gradually slowed, and then reversed, with an upward trend of recovery within 15 years. No species became extinct. Simulated counterfactual projections indicated outcomes would have been worse without the planning intervention; around eight species would have become extinct over that timeframe. To date, this planning approach has been applied to relatively high-profile species facing multiple threats, and where conflicting views, uncertainty, or lack of coordination among stakeholders constrain action.

We examined the likely reasons for the success of this approach using information and evidence from other studies and concluded that while only on-ground action can change the outcomes for species, science-based, participatory approaches to planning can create a turning point for threatened species by supporting stakeholders to transition quickly to more effective ways of working together.

6.4.4 Chapter 4 Population viability analysis provides new insights into the potential for conservation independence in the New Zealand kākāpō.

Population viability analysis (PVA) is a widely used tool in wildlife biology (Burgman et al; 1993; Akçakaya & Sjögren-Gulve 2000; Beissinger 2002; Gerber & González-Suárez 2010; Lacy 2019). Its broad use in guiding species conservation planning has been both promoted (e.g. Machado 1997, Himes Boor, 2014) and cautioned against (e.g. Redford 2011; Wolf et al. 2015). PVA is integral to a method called Population and Habitat Viability Assessment that is used by the IUCN SSC and proven successful in Chapter 3 at influencing good conservation outcomes. However, its inclusion in planning adds time and cost, necessitating a clear understanding of how and when it adds essential value to this process.

My research explored the utility of PVA models to conservation planning through an in-depth analysis of New Zealand's critically endangered, and intensively managed, kākāpō (*Strigops habroptilus*). With mammalian predators controlled, the kākāpō population is currently predicted to decline at c. 1.6% p.a. I built models to simulate five hypothetical pathways to achieving full or partial ecological independence (ongoing unassisted growth). Population growth was achieved by increasing mean mast-ripening intervals to 3 years or various combinations of increased mast-ripening, larger population size and reduced egg infertility. Longer than anticipated average lifespan and higher than anticipated inbreeding resilience could also contribute to growth. Without growth, offsetting ongoing expected declines at proposed sites could eventually require an intensively managed source population of c. 2.5-25 times the current size. In summary, securing conservation independence for kākāpō needs both stringent control of mammalian predators and securing larger sites with more abundant natural food.

The kākāpō population is subject to complex viability influences including genetic and demographic stochastic risks, threats from introduced mammalian predators, intensive management, and natural environmental fluctuations. Using PVA models to uncouple these influences and examine each in turn helped clarify their impacts, which could otherwise be masked by the effects of intensive management. This research underscores how PVA can support planning for highly threatened, small, and fragmented populations by providing insights into viability drivers and a means to test hypotheses where empirical studies would be impractical and risky. It adds weight to the argument for integrating PVA into planning projects for species sharing similar characteristics, within which its role can prove pivotal.

6.4.5 Chapter 5. Population and Habitat Viability Assessment: a One Plan Approach to saving the devil

The importance of including stakeholders in planning is regularly advocated in the literature and is widely viewed as essential to the development, acceptance, and implementation of species conservation plans (e.g. NMFS 2020; Sande 2005; IUCN SSC 2008; CPSG 2020). There are different models for involving stakeholders that range from informing them, through consultation, to full participation in the creation of plans (Arnstein 1969). The IUCN SSC planning process evaluated as part of this study, which showed good conservation outcomes for species, emphasises a participatory approach integrating population viability analyses and consideration of both *in situ* and *ex situ* management.

Using a 4-day workshop for the Tamanian devil, *Sarcophilus harrisii*, as a case-study, we described the mechanics of this form of planning method, explaining the workshop context, preparation, discussions and recommendations. In addition, we described subsequent progress on the conservation priorities agreed by consensus during the workshop. Further to the published text submitted in the chapter, supplementary materials are provided as Appendix C, describing in more detail the preparatory work, process steps and consensus-building approaches used in this project.

This work further demonstrates the value of PVA, in integrating planning for *in situ* and *ex situ* populations under a single plan. It describes the practical dimensions of using PVA successfully, as well as in this case, formal wildlife Disease Risk Analysis (Jakob-Hoff et al. 2014) within participatory workshops with diverse stakeholders. It emphasises the importance of local community engagement and further supports claims that participatory planning benefits conservation progress.

6.5 DISCUSSION

This thesis considered how species conservation planning could contribute to global efforts to reverse biodiversity declines, aligning specifically with Target 4 of the Kunming-Montreal Global Diversity Framework. This target calls on countries to prevent extinctions and take urgent action to recover known threatened species. On the basis that the action called for should be planned, we first sought to gauge the scale of planning required by identifying the number of known threatened species.

The IUCN provides a readily accessible figure for the number of globally threatened species. However, this provides only a partial estimate of the total number of plans needed. Most species-level conservation (and therefore species-level planning) is delivered by, or with the support of, national or sub-national governments which often have their own red lists for taxa within their jurisdiction. The combined number of threatened species on these lists, despite the inevitable duplication, is a better indication of the number of plans needed. A register of national red lists is held at the Zoological Society of London (ZSL and IUCN National Red List Working Group 2022); however, the 113 contributing countries use a range of formats preventing easy compilation and rendering the records difficult to search. National red lists are a priority for the IUCN (Miller et al 2007) and the aim is to develop a searchable database but no funding is currently available (Ledger pers. comm.).

In the absence of a complete estimate for the number of plans needed, I took the number of globally threatened species as a minimum baseline. However, assessing how many of these are already planned for proved difficult because of limited publicly available databases and scattered unpublished information. For example, the Philippines currently has only one "official" species conservation plan, for the Tamaraw, *Bubalus mindorensis* (J. Burton, pers. comm.). Despite this, the IUCN SSC planning database lists several plans for Philippines species developed in partnership with the national Department of Environment and Natural Resources (DENR) and there is an unpublished DENR multi-species plan covering dozens of species that is inaccessible via web search (IUCN/CPSG/CI/DENR-BMB 2022). This situation is likely to exist in other countries. Meanwhile, the IUCN's Red List database houses information on whether species have plans. However, the relevant fields are optional for assessors and rarely completed (Hilton-Taylor pers. comm.). A 2021 requested download of entries to this field, for known threatened species, showed only c. 1700 entries, many of which were unrelated to planning. This is a missed opportunity to track planning, and planning gaps, for the world's most threatened species.

Similar challenges were faced when attempting to evaluate the conservation impact of plans. Change in red list category was selected as a measure of a planning project's success because it is comparable across taxa and is a key performance indicator for Target 4. The IUCN's SSC planning database includes hundreds of species plans developed at the invitation of governments. As a result, many cover national or local populations. With no searchable national red list categories, the IUCN Global Red List categories were the only metrics available and these are mostly species-level listings which were not compatible with the units planned for. As a result, the sample for this project was significantly reduced from that envisaged at the outset.

The PVA work completed as part of this research would have benefited from a small workshop involving members of the Kākāpō Recovery Group (KRG) as well as the Dragonfly Data Science team involved in developing the original models that predicted population declines in absence of intensive management. The baseline model presented here generated similar rates of decline to the Dragonfly Science models. However, this result could have been achieved in various ways and a direct and detailed comparison of the Dragonfly Science and VORTEX model structures and inputs would be helpful in ensuring that data and insights from the KRG have been fully captured and accurately interpreted.

This workshop was about the conservation of Tasmanian devils and there is no doubt that the complexity of the situation for this species required targeted, species-level attention. Most workshop participants were either devil or wildlife health specialists and, as a result, discussions about which offshore (DFTD-free) islands might be suitable for housing introduced “insurance” devils initially focused solely only potential suitability for devils. However, the Traditional Owners present, who carried unique insights into the broader ecology of the islands discussed, drew attention to the risks to local non-threatened but highly valued “mutton bird” colonies. Damage to these would not only have a negative impact on the local ecology but could prejudice essential local support for the project. This served as an important reminder of one of the disadvantages of taking a single-species planning route, which is the potential for becoming siloed from broader biodiversity conservation values and aims. It underlines the importance of pursuing diversity among stakeholders as well as the benefits of engaging local communities in planning at all stages (Conallin et al. 2018).

At the Tasmanian Devil workshop, and in the projects described in Chapter 3 the approach to building consensus was through facilitated discussions in which all participants were encouraged to participate. The risk of building consensus in this way (instead of, for instance, building on individual views canvassed privately) is that it will lead to biased outcomes because some participants will be unwilling to speak-up. However, in a planning forum like this one, dominated by those who will be working together to implement the plan, it is important that those involved learn how to: raise their issues respectfully; listen actively when others raise their issues; and disagree agreeably and constructively. This can be critical to the future success of the project. Well-facilitated discussions at the planning stage can help develop this culture of constructive dialogue and collaboration (Drolet & Morris 2000; Mackelworth et al. 2012; Madden & McQuinn 2014; Black 2015).

Compared to most conservation planning workshops the large emphasis on disease in the Tasmanian Devil planning project example described in Chapters 5 was unusual and reflected the primary threat driving Tasmanian devil population declines. Wildlife Disease Risk Analysis (DRA) is a well-documented step-wise method for evaluating disease risk and recommending remedial responses (see Jakob-Hoff et al. 2013). It is most often done as a desk-top exercise focused on mitigating negative health impacts resulting from the deliberate movement of wild or domestic animals from one location to another, though it can be applied to other wildlife disease risk situations. In this case, to improve integration between the wildlife health management community and the broader devil conservation community, the DRA steps were carried out by a working group operating within, and interacting with, the broader planning process for the insurance metapopulation. Despite its obvious value there are few published examples of this kind of “braided” approach. Yet as all planning essentially converges on defining a successful outcome, identifying the challenges to achieving it and working out how to overcome those challenges to achieve success, it should be possible to align different methods in exactly this way, that is, by: 1) matching the steps of one method to comparable steps in another; 2) identifying key and comparable points for working group reporting; and 3) designing to ensure all working groups will be on track to develop their final recommendations at roughly the same time. An important lesson from the devil workshop was that while the species conservation planning and DRA steps align well on paper, the difference in time required for matched steps was considerable. In practical terms this meant that groups were not always reporting on comparable steps during report-back sessions, and the DRA work was only partially complete by the close of the workshop. Though not ideal for workshop dynamics, the necessary work was finalised in the following weeks creating no obvious set-backs. Importantly, subsequent CPSG DRA-species conservation planning workshops have factored into their design additional preparatory work (in particular the pre-selection and prioritisation of disease hazards) to improve alignment (e.g. Jakob-Hoff et al. 2017).

Building on this idea, like DRA there are other conservation action disciplines that have their own planning methods which, with careful customisation, could be formally braided into a PHVA-style workshop structure to provide a richer output and advance the likelihood of successful implementation. Examples include: *ex situ* conservation needs assessment and planning (e.g. McGowan et al. 2017); and planning behaviour change campaigns (e.g. Wallen & Daut 2018). Structured Decision Making (SDM) and Open Standards for the Practice of Conservation (O/S) approaches (described briefly in Chapter 2) could also be usefully integrated into the PHVA-style planning projects described in Chapters 3 and 5. For example, when working to agree which actions to recommend, participants are often faced with choices between alternative courses of action where there are multiple potentially conflicting objectives (such as minimising extinction risk, minimising cost, maximising animal welfare) requiring difficult trade-offs and/or where there are risks or uncertainties associated with the species’ response to those alternative courses of action. Chapter 4 demonstrates the way in which PVA can help with this, by demonstrating the comparative viability of modelled populations under different management scenarios. SDM tools can improve this analysis in certain cases: by supporting the integration of objectives unrelated to viability into the decision-making process improving overall transparency; by helping groups to work out which information gaps or areas of uncertainty are worth pursuing based on the potential impact of perfect information on management outcomes; and, in situations where decisions are iterative, supporting adaptive management by helping to select actions that will both advance conservation aims and reduce uncertainty leading to better decisions in future (Canessa et al. 2015, 2016, 2019a, 2019b, 2023). Though simple SDM tools can easily be accommodated within a 3-4 day stakeholder workshop, more complex tools and analyses may need to be done later but their downstream acceptance by stakeholders can still benefit from the trust and understanding built during workshop discussions (Conallin et al. 2018; Siebert et al. 2006; Brancalion et al. 2016; Young et al. 2016; Maas et al. 2021).

Open Standards approaches can result in tightly defined operational plans, the implementers of which can more easily talk to and learn from each other because of the shared concepts and vocabulary. Though Open Standards practitioners have their own strategic planning method, the plan implementation components could, with some adjustments, be used with other strategic planning products such as the outputs of A PHVA workshop. For some planning projects this could improve the chances of successful implementation.

Introducing multiple methods into a single process can add time and cost, both of which are usually in short supply and in any case should remain proportionate to the project itself. Nevertheless, establishing the feasibility and limits of braiding potentially complementary methods would be a valuable target for further research and testing.

6.6 Recommendations for the future

Establish a searchable database of national red lists to aid in estimating global planning needs and measuring plan impact across taxa, aligning with the Red List Index.

1. Explore additional metrics to work alongside the Red List Index to evaluate species outcomes on smaller scales and over shorter time-frames, beginning with the Living Planet Index and Green Status Assessments (Grace et al. 2021; Ledger et al. 2023).
2. Promote inclusion of planning method as part of the information captured in plans to enable future comparison between methods. This study was one of the first to

evaluate multiple species plans sharing a common planning method. Though we were able to hypothesise about why this method might create the conditions for success, we were unable to compare it to other methods, partly because this information rarely travels with plans.

3. Using appropriate success indicators, evaluate the short-term outcomes of recently developed “light” planning methods to identify which are most promising.
4. Leverage relevant IUCN Global Red List database fields to capture information on which and how many globally threatened species are covered by a plan. Though not exhaustive, the information on publicly available plans generated in this thesis, would be a useful starting point for populating these fields. Equipping assessors with information on where to locate plans would help increase the likelihood that these fields are used.
5. Further work on the kākāpō models in collaboration with the KRG and Dragonfly Science experts is recommended to verify model alignment, incorporate recent data and discuss with stakeholders the ideas presented.
6. Design and test species planning projects that include braided processes integrating multiple discipline-specific planning methods, with the aim of enhancing multi-disciplinary collaboration and rigorous, transparent decision-making, towards better species outcomes.

While it should be stressed that it is conservation action on the ground that generates good outcomes for species, this thesis emphasises how species conservation planning can drive measurable change in species conservation status by helping those involved to work more effectively. This evidence underscores the need for increased investment in planning in an era of rapid and accelerating global change and biodiversity loss.

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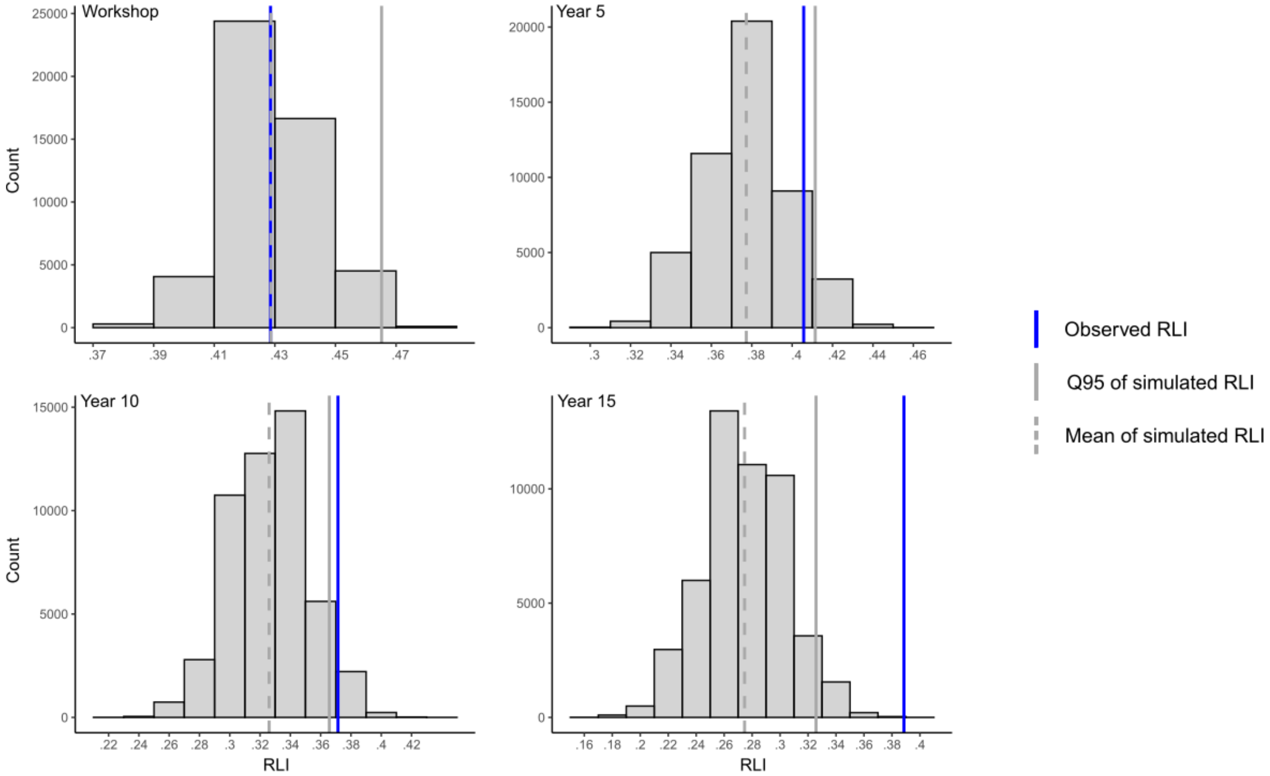

Figure S3.1a. Simulated distributions of RLI values (generated through bootstrapping) for the species set 0, 5, 10 and 15 years after planning workshops (n=35 projects) showing the 95th quantile (solid grey line) for simulated results, the mean simulated RLI (dashed grey line) and the observed RLI (blue line).

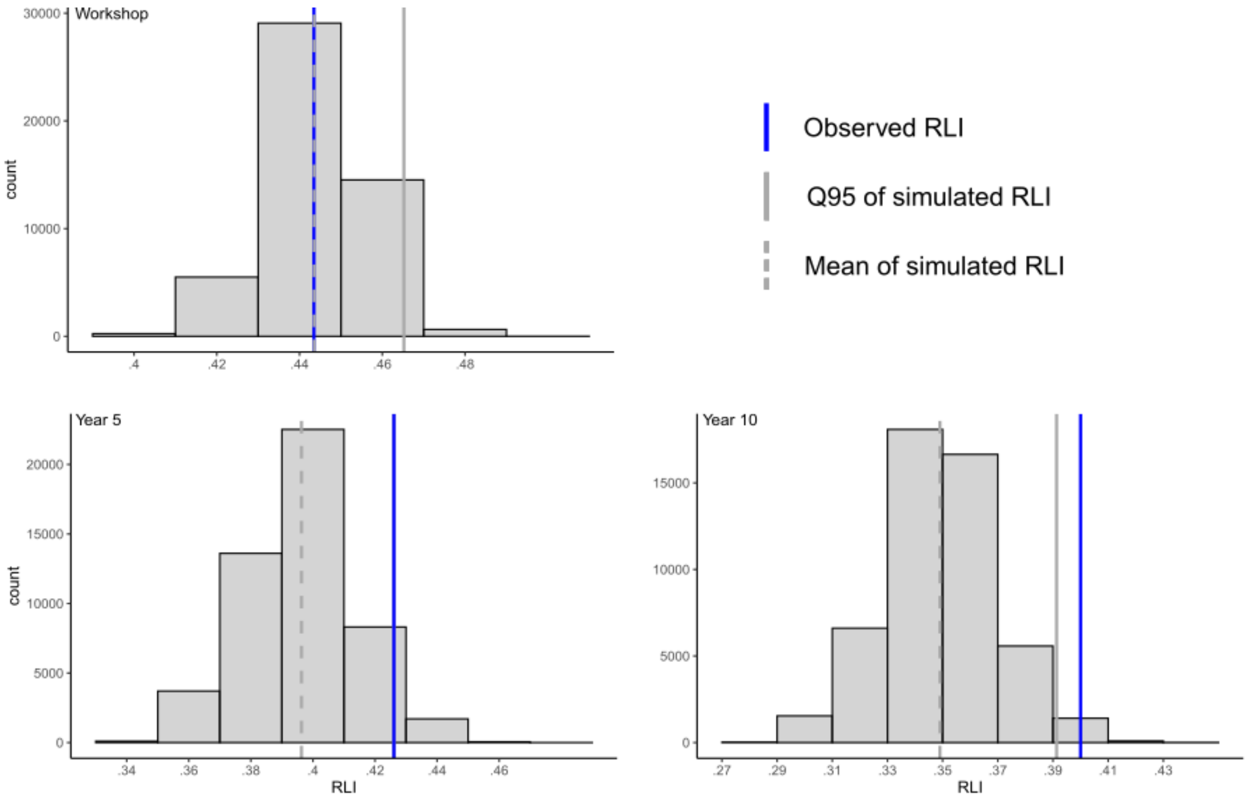

Figure S3.1b. Simulated distributions of RLI values (generated through bootstrapping) for 0, 5, and 10 years after workshops (n=46 projects) showing the 95th quantile (solid grey line) for simulated results, the average simulated RLI (dashed grey line) and the observed RLI (blue line).

Table S3.1: Details of workshops included in the study

Information on the background to the workshop, the main organisers or instigators, and the circumstances of the species at the time, including information on conservation measures already taken or in place. Information was drawn from workshop reports (www.cpsg.org/document-repository) and supplemented with information from IUCN Red List assessments where available (www.iucnredlist.org).

Project #	Workshop focus	Workshop background, purpose and conservation action already taken or in place
1	**Golden Lion Tamarin** (*Leontopithecus rosalia*)	The objectives of the workshop were to identify and evaluate the severity of threats that increase the probability of extinction for the species and to recommend actions and schedules needed to assure the long-term survival. Pre-existing conservation measures included surveys, area protection and a management committee for the species was in place. A captive population was in place and had provided animals for reintroduction.
	Main organisers:	*Instituto do Meio Ambiente e dos Recursos Naturais Renovàveis (IBAMA); Fundação Biodiversitas;, IUCN SSC CPSG; Jersey Wildlife Preservation Trust (now Durrell).*
2	**Golden-headed Lion Tamarin** (*Leontopithecus chrysomelas*)	The objectives of the workshop were to identify and evaluate the severity of threats that increase the probability of extinction for the species and to recommend actions and schedules needed to assure the long-term survival. Pre-existing conservation measures included area protection and a management committee for the species was in place. A captive population was in place. No animals had been reintroduced from it. This workshop was run concurrently to the one above, but with different people involved in each.
	Main organisers:	*Instituto do Meio Ambiente e dos Recursos Naturais Renovàveis (IBAMA); Fundação Biodiversitas; IUCN SSC CPSG; Jersey Wildlife Preservation Trust (now Durrell).*
3	**Black Lion Tamarin** (*Leontopithecus chrysopypgus*)	The objectives of the workshop were to identify and evaluate the severity of threats that increase the probability of extinction for the species and to recommend actions and schedules needed to assure the long-term survival. Pre-existing conservation measures included area protection and a management committee for the species was in place. A captive population had been established.
	Main organisers:	*Instituto do Meio Ambiente e dos Recursos Naturais Renovàveis (IBAMA); Fundação Biodiversitas; IUCN SSC CPSG; Jersey Wildlife Preservation Trust (now Durrell).*
4	**Black-Footed Ferret** (*Mustela nigripes*)	This species has been subject to several CPSG-facilitated planning workshops. Formerly widespread in central North America, the species declined through the 20th century to near extinction by the late 1970s, primarily because of prairie-dog control actions and sylvatic plague; by 1987, it was considered Extinct in the Wild. A range of *in situ* and *ex situ* measures were underway at the time of the workshop, including a captive programme.
	Main organisers:	*United States Fish and Wildlife Service; IUCN SSC Conservation Planning Specialist Group (CPSG).*
5	**Cotton-top Tamarin** (*Saguinus Oedipus*)	The workshop report is not available for this species. There is some basic information on when and where it was held, and on the process followed. Red List data from that time show that the species had a restricted, fragmented range, that its range included at least one protected area, that researchers had been monitoring its status and that an *ex situ* programme was in place.
	Main organisers:	*No information*

Project #	Workshop focus	Workshop background, purpose and conservation action already taken or in place
6	**Bornean Orangutan** (*Pongo pygmaeus*)	Pre-existing conservation measures were centred around increasing the effectiveness of rainforest protection and habitat restoration in Indonesia and Malaysia. The PHVA workshop was triggered by a 1991 International Great Ape Conference. The aim of the workshop was to build consensus around more explicit objectives for conservation action among field managers and researchers, using PVA models built collaboratively by those stakeholders using all available information. Explicit workshop goals were to: 1) Assess current wild status; 2) Identify and evaluate deterministic and stochastic threats to wild orangutan populations; 3) Review life history information as needed for simulation models; 4) Employ computer simulation models to evaluate risks of extinction to wild orangutan populations under current conditions and explore effects of various management scenarios; 5) Define requirements for 'viability' and delineate metapopulation structures that could be used to achieve it; and 7) Determine critical habitat requirements needed to achieve viability of wild orangutan populations and evaluate the status of current protected areas to satisfy these needs.
	Main organisers:	*Indonesian Forest Protection and Nature Conservation (PHPA); and IUCN SSC CPSG*
7	**Baiji Dolphin** (*Lipotes vexillifer*)	Precipitous decline of wild stocks had been observed in the decades preceding the PHVA. Action was being taken. A facility for study and breeding dolphins had been established, a reserve had been established upstream from Wuhan at Shishou, and a semi-natural reserve was established at Tongling. Plans were developed for five baiji conservation stations at various sites along the Yangtze. However, the species was continuing to decline. Workshop goals were: to use available data and expert knowledge to assess risks of extinction, identify critical factors affecting this, examine the effectiveness of suggested management scenario, identify specific projects likely to benefit from mobilising international cooperation and assistance, and recommend future priorities.
	Main organisers:	*Bureau of Fisheries Management and Fishing; Port Superintendence of China; Nanjing University; Mammalogical Society of China; IUCN SSC Cetacean SG; and IUCN SSC CPSG.*
8	**Lion Tailed Macaque** (*Macaca silenus*)	At the time of the workshop there were thought to be 3000-4000 Lion-tailed macaques spread across three southern states of India. Habitat fragmentation had created many isolated patches. Pre-existing conservation or protective measures were at the local level only. The species was present in about 20 protected areas though the level of protection afforded was variable and not considered adequate. There had been several studies of biology, ecology, behaviour, and distribution, including some repeated census surveys. At that time this had not been translated into action. About 570 animals were managed in zoos worldwide under regional (though not global) coordination of genetic and demographic attributes. The purpose of the workshop was to: examine the viability of remaining fragmented populations and recommend supportive measures; identify key threats and quantify the type and amount of mitigation required; explore the role of community education and awareness and recommend priority directions for this; identify key information gaps for further research; recommend management improvements to the captive population to increase its conservation relevance.
	Main organisers	*Arignar Anna Zoological Park; Tamil Nadu Forest Dept.; Indian Zoo Directors' Association; Central Zoo Authority of India; AZA Species Survival Plan for LTM; IUCN SSC CPSG; Zoo Outreach Organisation, India.*
9	**Sumatran Rhino** (*Dicerorhinus sumatrensis*)	Despite considerable efforts by governments and NGOs, in the two years preceding the workshop the global population was estimated to have dropped from around 600 individuals to less than 250 and possibly to as few as 185. Though a conservation strategy had been prepared previously, it had not resulted in sufficient protective measures to stop poaching and *ex situ* conservation breeding efforts were failing. Workshop goals were to bring together key players to assemble and evaluate available information and build consensus around recommendations for urgent action.

Project #	Workshop focus	Workshop background, purpose and conservation action already taken or in place
	Main organisers	*Indonesian Forest Protection and Nature Conservation (PHPA; IUCN SSC Asian Rhino SG; International Rhino Foundation; IUCN SSC CPSG.*
10	**Indian Rhino** (*Rhinoceros unicornis*)	At the time of the workshop the species was spread across six protected areas with some anti-poaching measures in place. None of the areas were at carrying capacity. Goals of the workshop were: to quantify and assess population parameters and habitat requirements; to evaluate the viability of each sub-population using PVA models; to analyse challenges to viability; to evaluate and make recommendations for mitigating these challenges; to explore potential use of translocation and reintroduction to strengthen non-viable populations and establish new ones and to make recommendations on how and when these should be implemented; and to cost the resulting recommendations for submission to funding agencies.
	Main organisers	*Forest Dept. of West Benga; Ministry of Environment; Govt. of India; IUCN SSC Asian Rhino SG; IUCN SSC CPSG; Zoo Outreach Organization.*
11	**Javan Gibbon** (*Hylobates moloch*)	Pre-existing conservation actions included area protection, habitat management and a captive population.
	Main organisers	*Indonesian Directorate General of Forest Protection and Nature Conservation (PHPA); Indonesian Primatological Society (IPS; IUCN SSC CPSG; The Gibbon Species Survival Plan (SSP) of the American Zoo and Aquarium Association (AZA).*
12	**Houston Toad** (*Anaxyrus houstonensis*)	The species was the first amphibian granted protection under the U.S. Endangered Species Act. A critical habitat was designated in 1978, in areas supporting the largest populations known at that time. In the 1970s, the state of Texas acquired land to aid in conservation. An effort was started in 1978 by the Houston Zoo to identify and supplement known populations or establish new ones. New populations were not established despite introducing over 500,000 individuals (adults, juveniles, larvae) into sites at the Attwater Prairie Chicken National Wildlife Refuge. The Houston Toad Recovery Plan was published by the USFWS in 1984 but was not considered to have addressed Endangered Species Act requirements (i.e. did not address known threats).
	Main organisers	*United States Fish and Wildlife Service; and the IUCNSSC Conservation Breeding Specialist Group.*
13	**Marsh Deer** (*Blastocerus dichotomus*)	Widespread marshland species at risk from habitat loss and degradation, and poaching. Before the workshop the species had declined to extinction in Peru and Uruguay. It persisted in Brazil, Paraguay, and Argentina but in varied degrees of fragmentation. Existing conservation actions included area protection – the species inhabited protected areas in all three countries – and anti-poaching laws. However, the level of protection offered by these measures was not sufficient to halt the rapid declines. There was an *ex situ* population, but this was small, spread across several facilities and was experiencing high mortality rates. Workshop aims were: to gather updated information from multiple experts in field biology, veterinary care and captive breeding; to use this information in PVA models to evaluate conservation implications and alternative management actions; and to use these analyses to consolidate previously isolated endeavours to conserve the species behind a single international conservation strategy, integrating *in situ* and *ex situ* work.
	Main organisers	*IUCN SSC Deer SG; and IUCN SSC CPSG*
14	**Baird's Tapir** (*Tapirus bairdii*)	Pre-workshop this species was listed on Appendix I of CITES and present in several protected areas. A captive population existed. Goals of the workshop were to review data from wild populations and use this to build PVA models to estimate extinction risk; to review the

Project #	Workshop focus	Workshop background, purpose and conservation action already taken or in place
		state of knowledge regarding species habitat requirements; to review the role of different threats as factors in the decline of the species; and to review the role to be played by captive management in the long-term management of the species.
	Main organisers	*Associación Nacional para la Conservación de la Naturaleza (ANCON); IUCN SSC Tapir Specialist Group; IUCN SSC Conservation Planning Specialist Group.*
15	**Gharial** (*Gavialis gangeticus)*	Prior to the workshop the species had been one of three targets for a conservation-directed crocodile programme that had been running for 20 years (since 1975). Work had included a supplementation programme, and area and habitat protection. The purpose of the workshop was to address concerns by some researchers and wildlife managers about the actual success of conservation measures to date and about the direction of the programme. The workshop reviewed activities and progress to date and set directions for the future.
	Main organisers	*Jiwaji University Gwalior; Forest Department of Madhya Pradesh; Ministry of Environment, Government of India; Zoo Outreach Organisation; IUCN SSC CPSG India.*
16	**European Bison** (*Bison bonasus*)	Historically, the species was distributed through western, central, and south-eastern Europe. By the early 20th century, free-ranging populations were extinct throughout their range except for a population in the Bialowieza Primeval Forest, which declined rapidly from 785 individuals in 1915 and became extinct after World War I (April 1919), and a population in the northwestern Caucasus region which met the same fate in 1927. At the time of the workshop the species was protected by law in each range country and had been the subject of a captive breed for release programme. The workshop was attended by representatives from 10 countries.
	Main organisers	*IUCN SSC European Bison SG; Poznan Zoo; IUCN SSC CPSG; EEP (European Zoo Association captive population management programme).*
17	**Barasingha** (*Rucervus duvaucelii*)	Pre-workshop, the species occurred as three subspecies numbering around 3500-4000 animals. Sub-populations were fragmented. Conservation action included area protection, with the species occurring in several protected areas with poaching controls. There was an *ex situ* population in zoos in the region. The workshop aimed to: assemble available information; bring together relevant individuals to identify problems and possible solutions; develop objective models of individual sub-populations to assess risk of extinction; formulate and test alternative management actions; recommend priorities for integrated *in situ* and *ex situ* management of the species.
	Main organisers	*Wildlife Institute of India; and Zoo Outreach Organisation. Facilitated by IUCN SSC CPSG*
18	**Orinoco Crocodile** (*Crocodylus intermedius*)	At the time of the workshop the species was virtually extinct in Colombia, with isolated sub-populations comprised of few individuals, in Venezuela. Widespread hunting had decimated the species in the 1930s, but hunting had mostly ceased since 1960, continuing only in some remnant populations. Despite this, and the legal protection of some sub-populations, natural recovery of wild populations was not observed, and other threats were thought to be operating. Workshop goals were to: bring together biologists that had worked with the species and international collaborators; assemble and evaluate available information using PVA models; and develop integrated *in situ* and *ex situ* conservation strategies for the species.
	Main organisers	*Grupo de especialistas en crocodrilos de Venezuela (GECV'); Servicio autonomo profauna – Marnr (PROFAUNA); Fundacion nacional parques zoologicos y acuarios (FUNZPA); Fundacion para la defensa de la naturaleza (FUDENA); Instituto de zoologia tropical=faciltad ciencias (IZT-UCV); Universidad central de Venezuela (UCV); IUCN SSC Crocodile SG; IUCN SSC CPSG.*
19	**Babirusa** (*Babyrousa babyrussa*)	The species declined during the 20th Century due to habitat loss, hunting and in some places natural disasters. Declines were precipitous in some places. Prior to the workshop the species had been fully protected under Indonesian law for many decades. It was listed on CITES Appendix I and was present in several protected areas, some very large. There was also a captive population.

Project #	Workshop focus	Workshop background, purpose and conservation action already taken or in place
	Main organisers	*Indonesian Directorate of Forest Protection and Nature Conservation (PHPA); Taman Safari Indonesia; IUCN SSC Pigs and Peccaries SG; IUCN SSC CPSG.*
20	**Tamaraw** (*Bubalus mindorensis*)	Prior to the workshop, the population had declined from an estimated 10,000 in 1900 to 100-400 animals, due to habitat loss, hunting and disease. Remaining sub-populations were fragmented. Efforts to establish a practical conservation management and research program for this species had been hampered by conflicting recommendations from outside organisations and multiple changes in supervising authority which had led to altered priorities, unreliable funding and suspension of activities. The purpose of the workshop was to assist local managers to: formulate priorities for a practical management program for survival and recovery of the species in wild habitat; develop a risk analysis and population simulation model for the species which could be used to guide and evaluate management and research activities; identify and initiate useful technology transfer and training; and identify and recruit potential collaborators for the conservation program.
	Main organisers	*Philippines Dept. Environment and Natural Resources, University of the Philippines Los Banos Foundation, Inc., IUCN SSC Asian Wild Cattle SG, IUCN SSC CPSG.*
21	**Lowland Anoa** (*Bubalus depressicornis*)	At the time of the workshop the species was considered to have been in decline since the 1900s due to hunting and habitat loss. It was legally protected from international trade (CITES I), local laws were in place to prevent hunting and it occurred in several protected areas, some small but others large. The aims of the workshop were to: review data from the wild and captive populations as a basis for assessing extinction risks, assessing different management scenarios, evaluating the effects of hunting, determining habitat and capacity requirements, the role of captive propagation, and priority research needs.
	Main organisers	*Indonesian Directorate of Forest Protection and Nature Conservation (PHPA); Taman Safari Indonesia; IUCN SSC Asian Wild Cattle SG; IUCN SSC CPSG.*
22	**Mountain Anoa** (*Bubalus quarlesi*)	At the time of the workshop the species was considered to have been in decline since the 1900s due to hunting and habitat loss. It was legally protected from international trade (CITES I), local laws were in place to prevent hunting and it occurred in several protected areas, some small but others large. The aims of the workshop were to: review data from the wild and captive populations as a basis for assessing extinction risks, assessing different management scenarios, evaluating the effects of hunting, determining habitat and capacity requirements, the role of captive propagation, and priority research needs.
	Main organisers	*Indonesian Directorate of Forest Protection and Nature Conservation (PHPA); Taman Safari Indonesia; IUCN SSC Asian Wild Cattle SG; the IUCN SSC Conservation Planning Specialist Group.*
23	**Mountain Gorilla** (*Gorilla b. beringei*)	The species was restricted to two small populations: one of about 300 individuals in the Bwindi Impenetrable National Park in Uganda, and the other of about 320 animals in the Virunga Volcanoes region. Both fell completely within existing protected areas. The species had previously been the focus of much monitoring and research, but civil unrest had been hampering these activities for some years leading to increases in habitat destruction and poaching. The objectives of the PHVA workshop were to assist local managers and policy makers to: 1) formulate priorities for a practical management program for survival, recovery, and long term viability of the two mountain gorilla populations in their wild habitat; 2) develop a risk analysis and population simulation model for the mountain gorilla which can be used to guide and evaluate management and research activities; 3) identify specific habitat areas that should be afforded strict levels

Project #	Workshop focus	Workshop background, purpose and conservation action already taken or in place
		of protection and management; 4) identify and initiate useful technology transfer and training; and 5) identify and recruit potential collaborators from central Africa as well as the greater international community.
	Main organisers	*Uganda Wildlife Authority Office; Rwandais de Tourisme et Parcs Nationaux; Institut Congolais pour la Conservation de la Nature; the IUCN SSC Primate Specialist Group; the IUCN SSC Conservation Planning Specialist Group.*
24	**Iberian Lynx** (*Lynx pardinus*)	At the time of the workshop the species was threatened by small numbers (<1,000 individuals), the high degree of fragmentation and the small size of the fragments (largest n=60). Management of the species was not centralised but rather spread across several different local authorities in Spain and Portugal. Conservation efforts were taking place to a greater or lesser extent in these different localities and a portion of the species was in protected areas. The aims of the workshop were to: assemble the available information about species biology ecology, habitat requirements, current and potential distribution and to identify key gaps; to discuss the biological, socio-political and legal issues around the species' conservation; to explore, with models, the conditions for a viable, inter-connected meta-population; and to work towards a cohesive, integrated management plan covering Spain and Portugal.
	Main organisers	*Ministerio de Medio Ambiente ; Dirección General de Conservation de la Naturaleza ; IUCN SSC Felid SG ; IUCN SSC CPSG.*
25	**Muriqui** (*Brachyteles arachnoides*)	Prior to the workshop and from the late 1970s and early 1980s, work to conserve Brazil's Atlantic Rainforest and its resident species had been underway, with this species one of its flagships. Basic research, conservation efforts and public awareness campaigns had been undertaken. At the time of the workshop, the species' distribution was fragmented, and numbers were small. Fragments varied in the amount of protection and monitoring in place and in the degree of 'naturalness' in the environment and its management. The aims of the workshop were to bring together stakeholders to consolidate species-wide information and to produce a collaborative and systematic conservation assessment for the muriqui, including recommendations for future action and priority research.
	Main organisers	*Fundação Biodiversitas, IBAMA; Conservation International – Brazil; IUCN SSC Primate SG; and IUCN SSC CPSG.*
26	**Goodfellow's Tree-kangaroo** (*Dendrolagus goodfellowi*)	Prior to the workshop efforts to conserve tree-kangaroos were confined to localised projects operated by several different local and international organisations. Land in PNG is mainly individual rather than government-owned which complicates the delivery of centralised initiatives. The aims of the workshop were to: elicit and consider relevant and species-wide information from all people with a stake in the future of the species; to use this in simulation models to determine risk of extinction under current conditions, factors making the species vulnerable to extinction, and which factors, if changed or manipulated, may have the greatest effect on preventing extinction; and to use these deliberations to develop an action plan for the long-term conservation of genetically viable populations of tree-kangaroos in PNG.
	Main organisers	*Dept. Environment and Conservation; National Museum and Art Gallery; Rainforest Habitat – University of Technology; IUCN SSC CPSG.*
27	**Doria's Tree-kangaroo** (*Dendrolagus dorianus*)	Prior to the workshop efforts to conserve tree-kangaroos were confined to localised projects operated by different local and international organisations. Land in PNG is mainly individual rather than government-owned which complicates the delivery of centralised initiatives. The aims of the workshop were to: elicit and consider relevant and species-wide information from all people with a stake in the future of the species; to use this in simulation models to determine risk of extinction under current conditions, factors making the species vulnerable to extinction, and which factors, if changed or manipulated, may have the greatest effect on preventing extinction; and to use these deliberations to develop an action plan for the long-term conservation of genetically viable populations of tree kangaroos in PNG.

Project #	Workshop focus	Workshop background, purpose and conservation action already taken or in place
	Main organisers	*Dept. Environment and Conservation; National Museum and Art Gallery; Rainforest Habitat – University of Technology; IUCN SSC CPSG.*
28	**Humboldt Penguin** (*Spheniscus humboldti*)	Species is restricted to the coasts of Chile and Peru. Prior to the workshop the species had been declining for more than three decades due to a range of factors. At the time of the workshop total numbers were estimated at 7500 in Chile and 5500 in Peru. Studies and surveys had been undertaken and conservation measures discussed and promoted but the issues were complex and available data not always adequate to answer key questions and ensure general agreement. Further, there were conflicting points of view on the severity of the problem and on appropriate mitigating action, primarily between researchers and fisheries managers. The aim of the workshop was to bring together stakeholders to elicit and evaluate the available information using collaboratively developed PVA models, to unpack key areas of conflict and in general to contribute to the development of a conservation strategy for the species acceptable to the main stakeholders.
	Main organisers	*Servicio Nacional de Pesca del Miniterio de Economia, Fomento y Reconstrucción; and IUCN SSC CPSG*
29	**Red Wolf** (*Canis rufus*)	The species was once relatively widespread but was brought to the brink of extinction through forest clearance and aggressive predator control measures, exacerbated by hybridisation issues. Prior to the workshop a captive programme had been established and wolves had been successfully bred and released to the wild. At the time of the workshop several critical issues were challenging the expansion of the programme in meeting its recovery goals: selection of additional restoration sites; assessing and managing the threat hybridisation represents to recovery; the need for an effective and feasible monitoring programme; and an assessment of the role of the captive breeding programme in facilitating further recovery in the wild.
	Main organisers	*US Fish and Wildlife Service; and IUCN SSC CPSG*
30	**African Penguin** (*Spheniscus demersus*)	The species had been subject to a range of challenges in the decades before the workshop, including fisheries interactions, disturbance and over-extraction, predation, and oiling. Previous work had included monitoring and implementation of local colony/site management plans, and oiled sea-bird rehabilitation. Prior to and during the workshop there were differences of opinion about what conservation directions to pursue. The aims of the workshop were to gather and evaluate the available information with stakeholders and use it in PVA models to determine: risk of extinction under current conditions; the factors that make the species vulnerable to extinction; which factors, if changed or manipulated, may have the greatest effect on preventing local extinction; and to use this information to develop a conservation action plan to improve the status of African penguins.
	Main organisers	*Birdlife International Seabird Conservation Programme; Avian Demography Unit, Uni. Of Capetown; IUCN SSC CPSG.*
31	**Ethiopian Wolf** (*Canis simensis*)	Pre-workshop the species had been officially protected since 1974, with hunting only under permit. None had been issued in the previous 15 years. Species was present in several protected areas. An Ethiopian Wolf Conservation Programme was in place. The purpose of the workshop was to develop a national conservation strategy to improve the status of remaining populations.
	Main organisers	*Ethiopian Wolf Conservation Programme; IUCN SSC Canid Specialist Group; WildCRU; Born Free; IUCN SSC CPSG.*
32	**Arabian Tahr** (*Arabitragus jayakari*)	The species had experienced severe past declines and remained in small, fragmented populations. Total numbers had grown from an estimated 2000 in 1978 to about 6400 in 1998, but the species was thought to be still at risk from a range of pressures. In the years preceding the workshop conservation action included protection from hunting and area protection. There were active research groups in each range country but little pooling or synthesis of recent information. There was rising interest in collaboration between conservation groups in the region. The purpose of the workshop was to allow researchers and conservation managers to combine their

Project #	Workshop focus	Workshop background, purpose and conservation action already taken or in place
		information to produce country and region-level review and analysis of the species, and to make recommendations for future conservation action.
	Main organisers	*IUCN SSC CPSG; Breeding Centre for Endangered Arabian Wildlife, Sharjah.*
33	**Riverine Rabbit** (*Bunolagus monticularis*)	The species had been classified as endangered since 1981 due to loss of burrowing habitat on which it relies due to conversion for agriculture. Prior to the workshop, a conservation-directed captive breeding programme had been established and attempts had been made to introduce the animals bred into an area of Karoo National Park. Surveys had been completed to assess population locations and numbers and to evaluate the location of remaining suitable habitat range-wide. The aims of the workshop were: to bring together all the groups responsible for the conservation and management of the species, to synthesise and evaluate the available information and, using PVA models to inform conclusions drawn, identify management and research priorities for the species.
	Main organisers	*IUCN SSC Lagomorph SG; and IUCN SSC CPSG.*
34	**Magellanic Penguin** (*Spheniscus magellanicus*)	The aims of the workshop were to agree with stakeholders the priorities for research and management for the species, as part of a broader penguin workshop that was built on the outputs of previous PHVAs for other *Spheniscus* species. Priority threats to this species were oil pollution, fisheries and climate change. Pre-existing action included no-fish zones (though these were recommended to be extended).
	Main organisers	*IUCN SSC CPSG; and Universidad Católica del Norte, Chile.*
35	**Giant Jumping Rat** (*Hypogeomys antimena*)	A conservation-directed *ex situ* programme had been in place for ten years prior to the workshop and surveys had been carried out to increase understanding of ecology, distribution, and threats. At the time of the workshop the species was suffering from its naturally restricted range, habitat destruction, predation by feral dogs and cats and disease. Workshop aims were to bring stakeholders together, assemble available information on the species and its threats, build PVA models to evaluate the information and agree priorities for future action.
	Main organisers	*Ministère de l'Environnement de Madagascar; Office National pour l'Environnement; Association Nationale pour la Gestion des Aires Protégé; Direction Générale des Eaux et Forêts; Faculté des Sciences, Département de Biologie Animale, Université d'Antananarivo; Madagascar Fauna Group; IUCN SSC Conservation Planning Specialist Group; Institute for the Conservation of Tropical Environments; Conservation International; IUCN SSC Primate Specialist Group.*
36	**Galapagos Penguin** (*Spheniscus mendiculus*)	The aims of the workshop were to agree with stakeholders the priorities for research and management for the species, as part of a broader penguin workshop that was built on the outputs of previous PHVAs for other *Spheniscus* species. Priority threats to this species were fisheries, climate change and small population related vulnerabilities. Pre-existing action included legal protection and patrols, wildlife monitoring, monitoring of feral cats on nearby islands. A dedicated PHVA workshop was carried out in 2005 but the 2000 workshop is considered the first science-based, inclusive and participatory workshop treatment.
	Main organisers	*IUCN SSC CPSG; and Universidad Católica del Norte, Chile.*
37	**Blue Swallow** (*Hirundo atrocaerulea*)	The species has a range of ten African countries but the range was fragmented and migratory or dispersal behaviour was unclear. Prior to the workshop the species was categorised as threatened in several countries and a Blue Swallow Working Group had been working to conserve the species. The aim of the workshop was to assess threats and prioritise required actions in a cohesive conservation plan.
	Main organisers	*BirdLife International – African Species Working Group; IUCN SSC CPSG South Africa; The Endangered Wildlife Trust.*

Project #	Workshop focus	Workshop background, purpose and conservation action already taken or in place
38	**Horned Guan** (*Oreophasis derbianu*)	The species occurs in Mexico and Guatemala but was reduced to small and fragmented populations. Pre-existing conservation action included area protection and legal protections afforded to threatened species including listing on CITES Appendix I. There was also a captive population. The purpose of the workshop was to develop a conservation action plan.
	Main organisers	*IUCN SSC CPSG-Mexico; and Africam Safari, CONAP.*
39	**Malayan Tapir** (*Tapirus indicus*)	The species range had continued to contract over decades, leaving small fragments in some places. At the time of the workshop the species was legally protected at national levels and listed on CITES Appendix I to prevent international trade. It was present in several protected areas and a captive programme was in place. It had been subjected to less *in situ* research and activity than South American tapir species but 18 members (25%) of the IUCN SSC Tapir Specialist Group were dealing directly with this species. The purpose of the workshop was to bring people together to review available knowledge and make recommendations for action, to update a 1997 IUCN SSC Action Plan for all tapir species. Though nominally an action plan, these early SSC documents are considered primarily status assessments. Therefore, this project was not excluded from the current study.
	Main organisers	*IUCN SSC Tapir Specialist Group (TSG); European Association of Zoos and Aquaria (EAZA) Tapir Taxon Advisory Group (TAG); Department of Wildlife and National Parks (DWNP), Malaysia; IUCN SSC Conservation Planning Specialist Group. Sponsors: Copenhagen Zoo; Department of Wildlife and National Parks (DWNP), Malaysia; Wildlife Conservation Society Thailand.*
40	**Harpy Eagle** (*Harpia harpyja*)	Species declines were from hunting and habitat loss. Species range includes Argentina, Belize, Bolivia, Brazil, Colombia, Costa Rica, Ecuador, French Guiana, Guatemala, Guyana, Honduras, Mexico, Nicaragua, Panama, Paraguay, Peru, Suriname and Venezuela. Prior to the workshop it occurred in many protected areas across its range, was legally protected at national levels and was listed on CITES Appendix I to protect it from international trade. An *ex situ* population existed. The report is in Spanish only.
	Main organisers	*El Instituto de Historia Natural y Ecología de Chiapas; Zoológico Miguel Álvarez del Toro Chiapas, México; and IUCN SSC CPSG.*
41	**Mountain Tapir** (*Tapirus pinchaque*)	The species occupies higher elevations than other tapirs, it is naturally rare, its distribution is fragmented, and it is illegally hunted. Prior to the workshop the species was legally protected at national levels and listed on CITES Appendix I to prevent international trade. It had been the subject of *in situ* research projects. Attempts to establish a captive population had been largely unsuccessful. The purpose of the workshop was to bring people together to review available knowledge and make recommendations for action, to update a 1997 IUCN SSC Action Plan for all tapir species. Though nominally an action plan, these early SSC documents are considered primarily status assessments. Therefore, this project was not excluded from this study. The report is in Spanish only.
	Main organisers	*IUCN SSC Tapir SG; IUCN SSC CPSG; IUCN SSC CPSG-México; and the Red Danta.*
42	**Maned Wolf** (*Chrysocyon brachyurus*)	The species occurs mostly in Brazil but also in Argentina, Brazil, Uruguay, and Paraguay. Distribution is fragmented in many places. The species is threatened by habitat conversion, persecution, diseases transmitted from domestic dogs, and other factors. Prior to the workshop, the species was present in protected areas throughout its range, was legally protected from hunting in several countries, had been the subject of research, was being regularly assessed for the IUCN Global Red List and was listed on CITES Appendix II to protect it from international trade. A captive population of several hundred individuals existed. The workshop brought together representatives from all range countries to form a common plan of action. The report is in Portuguese only.

Project #	Workshop focus	Workshop background, purpose and conservation action already taken or in place
	Main organisers	*Ministério do Meio Ambiente; Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis; Instituto Chico Mendes de Conservação da Biodiversidade; Diretoria de Conservação da Biodiversidade; Coordenação Geral de Espécies Ameaçadas; Centro Nacional de Pesquisas para a Conservação de Predadores Naturais.*
43	**Okinawa Rail** (*Gallirallus okinawae*)	Restricted to one, small population on one island and under threat from introduced predators and habitat conversion. The report is in Japanese only (except for the modelling chapter). Prior to the workshop the species was legally protected, and parts of its range were designated as protected areas. A captive population was in place. The workshop brought together stakeholders to create a single plan of action.
	Main organisers	*Japanese Government, IUCN SSC Conservation Planning Specialist Group.*
44	**Lowland Tapir** (*Tapirus terrestris*)	The species occurs in Argentina, Bolivia, Brazil, Colombia, Ecuador, Guyana, French Guyana, Paraguay, Peru, Suriname, and Venezuela. Habitat destruction and fragmentation, with resulting population isolation, and intensive hunting, were the main factors implicated in observed declines. The workshop brought together representatives of all 11 lowland tapir range countries. At the time of the workshop the species was present in many protected areas and carried legal protection against hunting in many parts of its range. It was listed on CITES Appendix II and a large captive population was in place.
	Main organisers	*IUCN SSC Tapir Specialist Group (TSG); Sorocaba Zoo, Sorocaba São Paulo, Brazil; Houston Zoo Inc., United States; and Sorocaba Convention and Visitors Bureau, Sorocaba, São Paulo, Brazil.*
45	**Rio Grande Silvery Minnow** (*Hybognathus amarus*)	The species historically occupied approximately 3,862 river kms in New Mexico and Texas. At the time of the workshop this was reduced to one 280 km reach of the Rio Grande in New Mexico. Increasing demands for available water had altered the normal hydrologic and ecological processes in the Rio Grande. Ongoing drought in this area of the United States had exacerbated these problems further. The species had been listed as federally endangered since 1994. Water management and use is a contentious issue, made increasingly so by the escalating scarcity. Conservation efforts were hampered by a divergence of views about resource use priorities and impacts. The workshop was designed to broaden stakeholder involvement and enhance information sharing across scientific, social, and economic groups and interests. The product was a detailed action plan for future management of the silvery minnow within New Mexico and throughout its range.
	Main organisers	*Middle Rio Grande Endangered Species Collaborative Program. Funded by USFWS.*
46	**Mangrove Finch** (*Geospiza heliobates*)	Inherently vulnerable due to small, restricted, and fragmented distribution. A range of potential exacerbating factors at play though precise reasons for recent declines were not known with certainty. The species habitat is protected within the Galápagos National Park. At the time of the workshop, research was underway to assess the impact of rats and predator control was ongoing. A study of breeding biology had been running for several years. Research into the factors behind the species' decline was continuing. A new project aiming to clarify the need for *ex situ* breeding or translocation had been running for a year. The workshop aimed to bring all current efforts together towards a single but multi-dimensional plan of action.
	Main organisers	*Charles Darwin Research Institute, Galápagos; Durrell; Parque Nacional Galápagos; Darwin Initiative; IUCN SSC CPSG and CPSG Mesoamerica.*

Note that information on these characteristics was not always readily accessible. Some reports could not be located, some were in a language not easily read and in other cases the information on previous action was not documented explicitly. The difference between the main organisers and workshop sponsors was not clear in all cases. Some unintentional errors may have resulted from this. The IUCN SSC Conservation Planning Specialist Group was formerly the IUCN SSC Conservation Breeding Specialist Group and is referred to as such in older reports.

APPENDIX B. CHAPTER 4. SUPPLEMENTARY MATERIALS

VORTEX INPUT FILE (BASELINE, UNMANAGED)

VORTEX 10.5.5.0 -- simulation of population dynamics
Project: KAK_Baselines
Scenario: Base_WH_S1
01/12/2023
1 populations simulated for 100 years for 1000 iterations
Scenario Settings Notes: Base Model for Whenua Hou with no management (S1)
Sequence of events in each time cycle:
EV
Breed
Mortality
Age
Disperse
Harvest
Supplement
rCalc

Ktruncation
GSUpdate
PSUpdate
ISUpdate
Census
Extinction defined as no males or no females.
Inbreeding depression with a genetic load consisting of
6.29 total lethal equivalents per individual, of which
50% are due to recessive lethals, and the remainder are lethal equivalents not subjected to removal by selection.
Populations:
Population1
Reproductive System:
Polygyny, with new selection of mates each year
Females breed from age 10 to age 65
Males breed from age 10 to age 65
Maximum age of survival: 65
Sex ratio (percent males) at birth: 50
Correlation of EV between reproduction and survival = 0
EV sampled from binomial distributions.

Reproductive Rates Notes: couldn't get into fertility spreadsheet - instead used Dragonfly data - P(breeding/masting year) = 0.34 x P(female breeds when mast doesn't ripen)=0.385 = 0.1309. Then use catastrophes to emulate impact of masting year (using chance of masting year in which the fruit ripens, which varies among sites (WH=0.34 x 0.275) and elevation of reproductive rate when a masting year occurs in which the fruit ripens - shift from 0.13 to 0.95).

Offsprng = eggs in this, such that egg infertility rates need to be factored into first year mortality.

Mortality Notes: Mortality age 0-1 derived from composite of non-breeding years and breeding but non-ripening years. Made up of loss dur to infertility (34.3%) and losses from fertile eggs before fledging (76.2%) leaving 15.65% fledgelings per egg (or 84.36% first year mortality from egg to fledgeling)

Population specific rates for Population1

Percent of adult females breeding each year: 13
with EV(SD): 1.3
Percent of adult males in the pool of breeders: 25
Normal distribution of brood size with mean: 2.4 with SD: 0.1
Female annual mortality rates (as percents):
Age 0 to 1: 84 with EV(SD): 4
Age 1 to 2: 47 with EV(SD): 5
Age 2 to 3: 1.5 with EV(SD): 0.25
Age 3 to 4: 1.5 with EV(SD): 0.25
Age 4 to 5: 1.5 with EV(SD): 0.25
Age 5 to 6: 1.5 with EV(SD): 0.25
Age 6 to 7: 1.5 with EV(SD): 0.25
Age 7 to 8: 1.5 with EV(SD): 0.25
Age 8 to 9: 1.5 with EV(SD): 0.25
Age 9 to 10: 1.5 with EV(SD): 0.25
After age 10: 1.5 with EV(SD): 0.25

Male annual mortality rates (as percents):
Age 0 to 1: 84 with EV(SD): 4
Age 1 to 2: 50 with EV(SD): 5
Age 2 to 3: 1.5 with EV(SD): 0.25
Age 3 to 4: 1.5 with EV(SD): 0.25
Age 4 to 5: 1.5 with EV(SD): 0.25
Age 5 to 6: 1.5 with EV(SD): 0.25
Age 6 to 7: 1.5 with EV(SD): 0.25
Age 7 to 8: 1.5 with EV(SD): 0.25
Age 8 to 9: 1.5 with EV(SD): 0.25
Age 9 to 10: 1.5 with EV(SD): 0.25
After age 10: 1.5 with EV(SD): 0.25
Catastrophe 1: Ripening Year
Local impact
Frequency (%): 9
Reproduction reduced by severity multiplier: 10
Survival reduced by severity multiplier: =1-(0.25*(A<1))
Initial population size:

Age 0 1 2 3 4 5 6 7 8 9 10 11 12 13 14 15 16 17 18 19 20 21 22 23
24 25 26 27 28 29 30 31 32 33 34 35 36 37 38 39 40 41 42 43 44 45 46 47
48 49 50 51 52 53 54 55 56 57 58 59 60 61 62 63 64 65 Total

Females 0 4 2 2 2 2 2 2 2 1 2 2 2 2 2 2 1 2 2 2 1 2 2 1 2
2 1 2 1 2 2 1 2 1 2 1 2 1 2 1 2 1 2 1 1 2 1 1 2 1 1 2 1
1 2 1 1 1 2 1 1 1 1 1 2 1 103

Males 0 4 2 2 1 2 2 2 2 2 1 2 2 2 1 2 2 1 2 2 1 2 1 2 2
1 2 1 2 1 2 1 2 1 2 1 1 2 1 2 1 1 2 1 1 1 2 1 1 2 1 1 1
1 2 1 1 1 1 2 1 1 1 1 1 1 97

Carrying capacity: 200

with EV(SD): 0

Table S4.1 Additional notes relating to site-specific estimates of catastrophes supplementary information
These effects were estimated by the Kākāpō Recovery Group. Grey shading = risks already included as part of background age-specific mortality. Unshaded = added to site-specific models as catastrophes.

Sites	Risk	Description	Frequency of occurrence	Impact	Frequ.	Impact (survival)
Whenua Hou	Stoat incursion	Very unlikely: only by human interference. If stoat got to WH, would be quickly dealt with.	0.5%	Loss of 4% of kākāpō of any age	0.5%	0.96
	Rat incursion	May affect breeding: would kill eggs/chicks. Would be detected quickly, before widespread.	1/30 years	Loss of 5% of eggs/chicks	3.3%	0.95 for chicks
	Flooding	Nests flooded. Occurred on Anchor in 2016: 2/16 nests flooded, with loss of 3 chicks.	1/20 years	Loss 3/25 chicks	5%	0.88 for chicks
	Disease	E.g aspergillosis 2019: lost 2 adults + 5 chicks + 2 juvs	1/10 years	Loss of 9/130 birds (with intervention)	10%	0.93
	Fire	Fire could spread to 30% of island and nests. Many adults unlikely to escape	1/50 years	30% of chicks, 30% of adults	2%	0.7
Pukenui	Stoat breach	Moderately likely. Stoat seen in 2005: all females evacuated.	1/15 years	Loss of 5% of kākāpō of any age	6.6%	0.95
	Rat incursion	May affect breeding: would kill eggs/chicks. Would be detected quickly, before widespread.	1/30 years	Loss of 5% of eggs/chicks	3.3%	0.95 for chicks
	Flooding	Nests flooded. Occurred on Anchor in 2016: 2/16 nests flooded, with loss of 3 chicks.	1/5 years	Loss of 3/25 chicks	20%	0.88 for chicks
	Disease	E.g aspergillosis 2019: lost 2 adults + 5 chicks + 2 juvs	1/10 years	Loss of 9/130 birds (with intervention)	10%	0.93
	Fire	Fire could spread to 30% of island and nests. Many adults unlikely to escape	1/50 years	30% of chicks, 30% of adults	2%	0.7
Hauturu-o-Toi	Stoat introduction	As per WH	0.5%	Loss of 4% of kākāpō of any age	0.5%	0.96
	Rat incursion	May affect breeding: would kill eggs/chicks. Would be detected quickly, before widespread.	1/30 years	Loss of 5% of eggs/chicks	3.3%	0.95 for chicks

Sites	Risk	Description	Frequency of occurrence	Impact	Frequ.	Impact (survival)
	Flooding/slips	Nests flooded. Adults drown	1/5 years	Loss of 5% adults 10% chicks	20%	0.95 for adults 0.90 for chicks
	Disease	E.g aspergillosis 2019: lost 2 adults + 5 chicks + 2 juvs	1/10 years	Loss of 9/130 birds (with intervention)	10%	0.93
	Fire	Fire could spread to 30% of island and nests. Many adults unlikely to escape	1/50 years	30% of chicks, 30% of adults	2%	0.7
Rakiura	Stoat breach	As per WH, except lower monitoring means that detection would take longer.	0.5%	Loss of 20-30%	0.5%	0.75
	Rats	As per WH, but lower monitoring level.		Loss of 50% of eggs/chicks	3.3%	0.5 for chicks
	Flooding/slips	Nests flooded. Adults drown	1/5 years	Loss of 5% adults 10% chicks	20%	0.95 for adults 0.90 for chicks
	Disease	E.g aspergillosis 2019: lost 2 adults + 5 chicks + 2 juvs	1/10 years	Loss of 9/130 birds (with intervention)	10%	0.93
	Fire	Fire could spread to 30% of island and nests. Many adults unlikely to escape	1/50 years	30% of chicks, 30% of adults	2%	0.7
Te Kākahu-o-Tamatea	Stoat breach	Moderately likely. Stoat seen in 2005: all females evacuated.	1/15 years	Loss of 5% of kākāpō of any age	6.6%	0.95
	Rat incursion	May affect breeding: would kill eggs/chicks. Would be detected quickly, before widespread.	1/30 years	Loss of 5% of eggs/chicks	3.3%	0.95 for chicks
	Flooding	Nests flooded. Occurred on Anchor in 2016: 2/16 nests flooded, with loss of 3 chicks.	1/5 years	Loss of 3/25 chicks	20%	0.88 for chicks
Five Fingers Peninsula?	Disease	E.g aspergillosis 2019: lost 2 adults + 5 chicks + 2 juvs	1/10 years	Loss of 9/130 birds (with intervention)	10%	0.93
	Fire	Fire could spread to 30% of island and nests. Many adults unlikely to escape	1/50 years	30% of chicks, 30% of adults	2%	0.7

Sites	Risk	Description	Frequency of occurrence	Impact	Frequ.	Impact (survival)
Coal Island?	Stoat breach	Moderately likely. Stoat seen in 2005: all females evacuated.	1/15 years	Loss of 5% of kākāpō of any age	6.6%	0.95
	Rat incursion	May affect breeding: would kill eggs/chicks. Would be detected quickly, before widespread.	1/30 years	Loss of 5% of eggs/chicks	3.3%	0.95 for chicks
	Flooding	Nests flooded. Occurred on Anchor in 2016: 2/16 nests flooded, with loss of 3 chicks.	1/5 years	Loss of 3/25 chicks	20%	0.88 for chicks
	Disease	E.g aspergillosis 2019: lost 2 adults + 5 chicks + 2 juvs	1/10 years	Loss of 9/130 birds (with intervention)	10%	0.93
	Fire	Fire could spread to 30% of island and nests. Many adults unlikely to escape	1/50 years	30% of chicks, 30% of adults	2%	0.7
Wainuiomata	Stoat breach	-	-	-	-	-
	Other risks?	-	-	-	-	-
	Other risks?	-	-	-	-	-
Resolution.	Stoat breach	Frequent incursions,but should be picked up by monitoring. Larger scale, so harder to mount a quick response before more kākāpō die	1/7 years	Loss of ~15% of adults every 7 years and 80% chicks every 14 years (when breeding coincides with incursion)	14%	0.85 for adults 0.6 for chicks
	Rat incursion	May affect breeding: would kill eggs/chicks. Would be detected quickly, before widespread.	1/30 years	Loss of 5% of eggs/chicks	3.3%	0.95 for chicks
	Flooding	Nests flooded. Occurred on Anchor in 2016: 2/16 nests flooded, with loss of 3 chicks.	1/5 years	Loss of 3/25 chicks	20%	0.88 for chicks
	Disease	E.g aspergillosis 2019: lost 2 adults + 5 chicks + 2 juvs	1/10 years	Loss of 9/130 birds (with intervention)	10%	0.93
	Fire	Fire could spread to 30% of island and nests. Many adults unlikely to escape	1/50 years	30% of chicks, 30% of adults	2%	0.7

Appendix C. Chapter 5. Supplementary Materials

Chapter 5. *Population and Habitat Viability Assessment: a One Plan Approach to Saving the Devil*, is included in the thesis because it helps to answer the question "What can applying recommended principles and tools look like in reality?" However, it was initially published as a book chapter and adheres to the brief and word limit for that publication. The following text is added here to increase its relevance to the thesis topic through a further, more detailed account of workshop preparation and delivery.

<u>Preparation for the workshop</u>

Six months before the workshop, the core team was assembled with representation from the main organising institutions. The team led the following preparatory steps for the workshop:

1) **Agreement on workshop goals.** Through open discussion it was agreed that the primary goal was to turn the meta-population concept into an achievable program of supported by all key implementers. Agreed supporting goals were: to review the meta-population strategy in light of current knowledge; to assess the feasibility of its component parts and, from the result; to agree a plan of action identifying what needs to be considered before action is taken, what needs to be done, when, by whom and with what resources.
2) **DRAFT workshop design.** The team agreed the basic building blocks of the workshop and CPSG drafted an initial agenda around this.
3) **Stakeholder analysis.** Of a larger number of potential stakeholders initially identified, the team prioritised approximately forty individuals who could offer knowledge, insights or influence over one or more of the following: 1) decisions about the future of the metapopulation, including its resourcing; 2) one or more of the devil management systems proposed; 3) devil biosecurity and health; 4) the status and prognosis for wild devils and/or of DFTD; 5) of local community views and concerns; or 6) recent, high-profile issues related to devil conservation including molecular genetic analyses and vaccine development.
4) **An invitation and briefing materials.** The invitation summarised the DFTD issue, the meta-population concept, the goals and intended outputs of the workshop and a draft agenda. Additional briefing materials were provided to help bring all potential participants to a more

comprehensive, shared understanding of the situation for devils and the options being considered for its future.

5) **Technical tool development.** Population Viability Analysis models were built to emulate each of the management systems to be discussed. Reliable demographic and genetic data were available for intensively managed devils in the form of a region-wide studbook database. The parameters derived from this informed the captive models. For wild and FRE models, captive parameters were used as a starting point and modified using some field data as well as expert opinion elicited through a questionnaire circulated to individual participants before the meeting. Models incorporated the predicted genetically effective population size of each system to allow genetic as well as demographic performance to be compared across systems, under different management scenarios. The parameters and assumptions of the models were presented on the first day of the workshop for further comment and input. In preparation for the Disease Risk Analysis working group, a literature review was carried out, of diseases relevant to devils in all three management systems including but not limited to DFTD.
6) **Participant communication.** Regular communication with those invited to the workshop, as well as with a larger group of interested parties unable to attend, was maintained throughout to avoid unnecessary rumour and speculation about the workshop and to reassure those not attending that the outputs would be shared widely and would be considered advisory pending further consultation. Contact numbers were provided for those with concerns not otherwise addressed.

Workshop delivery

The workshop assembled 40 stakeholders and specialists, from 19 organisations or community groups, and lasted four days. Most participants stayed on-site, and evening functions were organised to maximise social interaction and relationship building.

The workshop was opened by a Tasmanian Government Minister and a Welcome to Country by a representative of Tasmania's Traditional Owner community. The first two hours were spent in formal presentations followed by questions and brief discussion aimed at bringing everyone to the same understanding of the situation for devils and of the workshop goals and process. Participants introduced themselves and were invited to share the issues that they would most like to see addressed in workshop discussions. These issues were subsequently assigned to the relevant working groups in addition to pre-agreed tasks.

Following this session, five working groups were formed: one for each of the three management systems proposed; one comprising PVA modellers who ran model scenarios on request, to emulate scenarios put forward for testing by the management system-based groups; and one running a formal Disease Risk Analysis to inform disease management protocols. A further “synthesis” group was formed later, to address cross-cutting issues and to integrate emerging recommendations.

Wild, Captive and Free-range Enclosure groups worked through the following steps: 1) discuss and agree the purpose of the management system within the wider program; 2) identify, describe and prioritise the issues that must be addressed to ensure the management system fulfils that purpose (including those raised by participants in the first session); 3) evaluate the possible strategies through which priority issues could be addressed; 4) agree the strategy that should be taken in each case; and 5) document detailed action steps for implementing each agreed strategy, clearly identifying next steps and who should take them.

The Disease Risk Analysis working group worked through the following steps: 1) discuss and agree what constitutes an acceptable risk; 2) confirm priority disease hazards for discussion; 3) assess the risk posed by each in terms of the likelihood of occurrence and the consequences of it; 4) for high risk disease hazards identify critical control points; 5) discuss and agree mitigation strategies for each critical control point; 5) discuss and agree a communication strategy (who should be informed of what and when); 6) recommend next steps and who should take them.

The Synthesis group was charged with recommending an achievable pathway to a meta-population-wide genetically effective population size of Ne=500, while at the same time minimising loss of the gene diversity already captured, through the following steps: 1) confirm or alter previously estimated Ne contribution per devil for each management system; 2) estimate annual cost per devil for each management system; 3) estimate population size limits for each system; 4) estimate time-to-establishment for Wild and Free-range Enclosure systems; 5) recommend target population size and time-lines for deployment of each system.

Facilitators worked to maintain the overall integrity of the concurrent workshop processes and supported working groups through the assigned tasks. Regular report-back sessions prevented silos developing and promoted cross-fertilisation of ideas among groups. Progress was by consensus and where not reached dissenting views were documented.

Following the workshop, participants reviewed iterative revisions of the report and all comments were addressed before finalisation. An oversight committee for implementation was established. Trust, relationships and good working practices created during these four days continued in the weeks and months following the workshop allowing work to progress quickly on the re-organisation, re-direction and expansion of the metapopulation. This would not have been possible prior to the workshop.