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Decision-support for marine spatial prioritisation

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Abstract

Human demands on natural resources result in habitat degradation, resource exploitation, prey competition, pollution, and threats to species viability. Halting the decline of biodiversity and alleviating human threats requires investing in conservation actions such as restoration, protection, and management of species and ecosystems. As funding for conservation is often limited, strategies are needed to ensure investments are allocated to places that will best deliver conservation outcomes. However, the development and execution of such strategies are confounded by our limited knowledge of natural systems and uncertainty about both the state of the systems we aim to conserve, and the costs and feasibility of our conservation efforts. Tools to inform decision-making exist, having emerged from fields such as economics, operations research, and mathematics, but are not often tailored and applied to solve conservation challenges. In this thesis, I examine a broad spectrum of applications around problem-based conservation prioritisations to illustrate their utility for decision-making. I chose a diversity of problems linked by this common theme deliberately.

In ***chapters 2*** and ***3***, I investigated common approaches to collecting and collating data for conservation decision-making. In ***chapter 2***, I challenged the notion that all data are useful for conservation and argued that unless new data changes a decision about an action in space or time, it is of limited utility for conservation decision-making. I present a conceptual framework of the types of impacts new data delivers to conservation. My primary message was to urge scientists collecting data with the motivation of informing conservation to examine which uncertainties (e.g. uncertainty about habitat condition or aspects of species demography) are the most important to reduce. I drew from the emerging field of animal telemetry to argue my case. Finally, I provide a decision tree to illustrate when new information should be pursued as opposed to managing with uncertainty and suggest value of information analysis as a tool to address this challenge.

In ***chapter 3***, I evaluated one of the most comprehensive species-focused spatial datasets available for the global oceans, BirdLife International's Important Bird and Biodiversity Areas (IBAs). IBAs are intended to delineate the most important marine habitats for conservation. Using Australia's Exclusive Economic Zone as a case-study, I first tested the

ability of the Australian IBA inventory to act as effective surrogates for other known biodiversity (e.g. known seabird ranges, ecoregions and benthic habitats) and then examined how various treatments of IBAs influenced the cost-efficiency of marine reserve networks. Based on my findings, I present the first “best practice” guidelines for including IBAs into systematic conservation planning processes.

C***hapters 4 and 5*** focused on the development of decision-support tools for two different aspects of marine conservation, spatial planning policy and global prioritisation strategies, respectively. In ***chapter 4***, I developed a system model that optimizes marine zoning allocations for three actions; establishing: no-take marine reserves, managed fishing areas and open-access fishing zones. My aim was to develop a simple, purpose driven model to inform decisions about how to optimally partition marine systems into different zones that maximise a conservation benefit (e.g. standing stock biomass) given a fixed budget, while maintaining a minimum level of fisheries catch. I found that when management budgets are small, investing the entire budget into no-take protected areas is the optimal strategy. As the management budget increases, growing the size of the management zone enters into the optimal zoning allocation. This rule of thumb was robust to changes in parameters and provides a starting point for managers overseeing coastal resources in countries where over-fishing and exploitation are concerns, to better tailor policy around proportional zoning allocations.

In ***chapter 5***, I developed a novel, flexible decision-support tool to inform a specific conservation financing mechanism called “Debt for Nature” swaps – the conversion of country’s debt in exchange for a commitment to protect nature. Small Island Developing States (SIDS) often have significant financial constraints (high debt ratios) that make it difficult to finance conservation. Thus, there is a need to prioritise future Debt for Nature swaps in those countries that can achieve the greatest return on investment. The tool leads users through a prioritisation protocol of enabling factors, the consideration of abatable (e.g. fishing) and unabatable (e.g. sea level rise) threats, benefits, weightings, costs and the likelihood of success to rank countries on their cost-effectiveness to inform global investment strategies. I provide a proof-of-concept to demonstrate the tool using Caribbean SIDS and coastal nations.

I conclude that solving complex conservation challenges requires clearly defined problems and user-inspired decision support tools that link actions (e.g. what can be done) to objectives, threats, and costs. Translating the hopes, dreams and fears of end-users is a time-consuming and challenging task. But in the end, it is only through a deep and detailed understanding of end-user needs that we can deliver the best decision-support tools for conservation.

Declaration by author

This thesis is composed of my original work, and contains no material previously published or written by another person except where due reference has been made in the text. I have clearly stated the contribution by others to jointly-authored works that I have included in my thesis.

I have clearly stated the contribution of others to my thesis as a whole, including statistical assistance, survey design, data analysis, significant technical procedures, professional editorial advice, financial support and any other original research work used or reported in my thesis. The content of my thesis is the result of work I have carried out since the commencement of my higher degree by research candidature and does not include a substantial part of work that has been submitted to qualify for the award of any other degree or diploma in any university or other tertiary institution. I have clearly stated which parts of my thesis, if any, have been submitted to qualify for another award.

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<u>Publications included in this thesis</u>

McGowan J, Beger M, Lewison R, Harcourt R, Campbell H, Priest M, Dwyer R, Lin H, Lentini P, Dudgeon C, McMahon C & HP Possingham. 2016. Linking research using animal-borne telemetry with the needs of conservation management. *Journal of Applied Ecology*. doi: 10.1111/1365-2664.12755. - incorporated as Chapter 2.

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McGowan J & Possingham HP. 2016. Commentary: Linking Movement Ecology with Wildlife Management and Conservation. *Frontiers in Ecology and Evolution*. doi: 10.3389/fevo.2016.00030.

Brown C, Bode M, Venter O, Barnes MD, **McGowan J**, Runge C, Watson JEM, & Possingham HP. 2015. Effective conservation requires clear objectives and prioritising actions, not places or species. *Proceedings of the National Academy of Sciences of the United States of America*. 112 (32) E4342. doi. 10.1073/pnas.1509189112.**Conference Abstracts:**

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McGowan J, Bode M, Beger M, Davis K, Giakoumi S, Krueck N, Mazor T, Yates K, & Possingham HP. 2014. Sea sharing or sparing: Should we focus on more habitat protection or no-take marine reserves. *Ecological Society of America conference,* Sacramento, CA, USA.

McGowan J, Beger M, Treml E, Green A, White AT, Heron S, van Hooidonk R, Wolff N, Klein C, Hock K, Mumby PJ, & Possingham HP. 2014. Identifying Regional Ecological Gaps to inform CTMPAS. *5th Coral Triangle Initiative on Conservation, Fisheries and Food Security. MPA Regional Exchange (REX5)*, Cebu, Philippines.

McGowan J, Beger M, Treml E, Green A, White AT, Heron S, van Hooidonk R, Wolff N, Klein C, Hock K, Mumby PJ, & Possingham HP. 2014. Conservation Gaps in the Coral Triangle: Towards a representative, connected and climate proof MPA system. *Society for Conservation Biology Oceania Regional Meeting*, Suva, Fiji.

Contributions by others to the thesis

This thesis consists of four manuscripts, with myself as lead author, which are either published or intend to be submitted for publication. Chapters 2, 3, and 4 have been published during my candidature with the assistance of collaborators. This is reflected in the use of the plural first-person pronoun “we” or “our.” Chapters 1 and 5 were written entirely by me with editorial assistance and use the singular first-person pronoun, “I” or “my.”

Chapter 1

This chapter was written by myself, with editorial input from Maria Beger, Carissa Klein and Hugh P. Possingham.

Chapter 2

Contributions to this chapter are detailed in the preceding “Publications included in this thesis” section.

Chapter 3

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This chapter is being prepared for submission to *Methods in Ecology and Evolution*. The idea for this manuscript was conceptualised by Hugh P. Possingham, Eddie T. Game, Rob Weary and myself. All data analysis was conducted by myself with the input of Hugh P. Possingham. The chapter was written by myself with editorial input from Hugh P. Possingham.

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Abbreviations

DOM - Dynamic Ocean Management

IBA - Important Bird and Biodiversity Area

EBSA - Ecologically and Biologically Significant Area

EEZ - Exclusive Economic Zone

GDP - Gross Domestic Product

IUCN - International Union for the Conservation of Nature

KBA - Key Biodiversity Area

MPA - Marine Protected Area

SIDS - Small Island Developing States

SMART – Specific, Measurable, Achievable, Realistic and Time bound

SST- Sea surface temperature

TNC - The Nature Conservancy

PE - Protection Equality

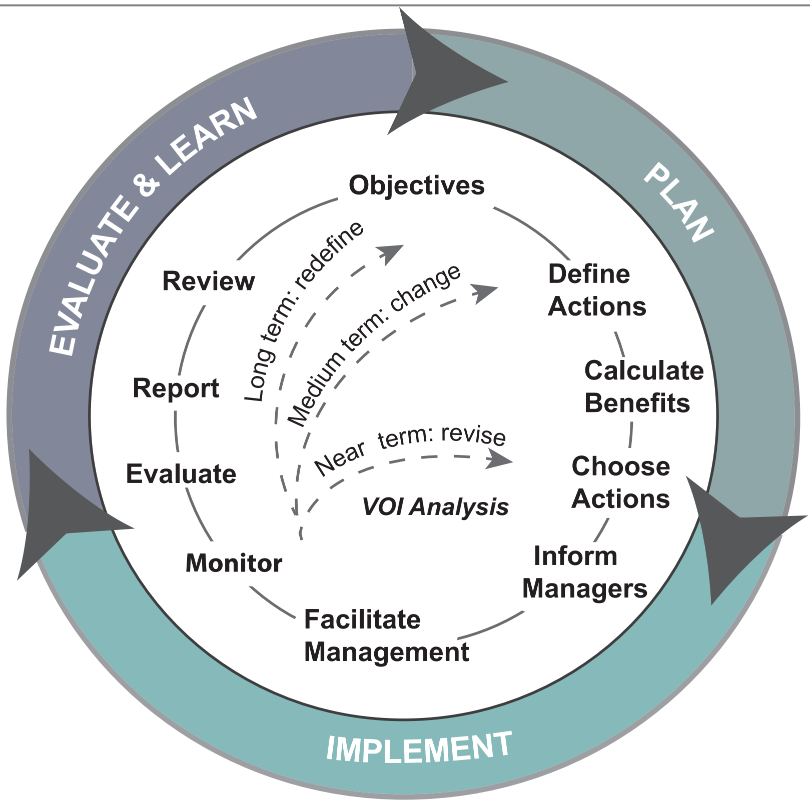
UV - Ultraviolet radiation

VoI - Value of Information

1 INTRODUCTION

Increasing human population coupled with increasing human dependencies on nature continue to threaten global biodiversity, degrade ecosystems, extirpate species and deplete renewable and non-renewable resources on the land and sea (Halpern *et al.* 2008; Butchart *et al.* 2015wilson; Venter *et al.* 2016; Ceballos *et al.* 2017). There are many promising management actions to ameliorate these trends, such as changing policies to promote food security and ecosystem services (Pauly *et al.* 2002; Palumbi *et al.* 2008), mitigating threats to ecosystem functioning (Worm *et al.* 2006) and/or protecting and restoring biodiversity (Watson *et al.* 2014; Bayraktarov *et al.* 2015; Davidson & Dulvy 2017). While much progress has been made to move nations towards protecting and managing their biodiversity under the international commitments set forth in the Convention on Biological Diversity (2011), substantial shortfalls in achieving our global and national conservation targets remain (Butchart *et al.* 2015; Klein *et al.* 2015). To meet these targets, practitioners and governments must pursue conservation planning techniques that inform decisions on the best course of action to take for a given conservation context (Groves & Game 2015). When these planning decisions are placed within the adaptive management cycle (Figure 1.1) (Holling 1978; McFadden *et al.* 2011), the process of identifying and selecting actions follows an iterative, well-structured and socially-engaged process (Groves & Game 2015; Schwartz *et al.* 2017) that accounts for the presence of uncertainty (Grantham *et al.* 2009b; Westgate *et al.* 2013).

The adaptive management cycle has three general stages: planning, implementation, and evaluation and learning (Fig 1.1). Whether "passive", which involves reviewing the performance of past or current actions to alter future actions, or "active", where there is a direct effort to the balance between knowledge acquisition and conservation action, adaptive management capitalises on opportunities to improve the effectiveness of management strategies as new knowledge is gained (Walters and Hilborn 1978; McCarthy & Possingham 2007; Grantham *et al.* 2009b), or conditions in the social, political or ecological system change.



Planning: Setting objectives, choosing actions and calculating benefits
Conservation plans can have many goals, for example improving the viability of a population or increasing the ecosystem services provided by that population. Often those goals rely on imperfect knowledge of the systems, such as the movement patterns of species, in which case the spatial and temporal scale of that movement is critical to forming specific and quantifiable objectives. Choosing which actions to take when and where requires knowledge of costs, feasibility and the likelihood that the action will successfully deliver the desired outcome. Value of information (VOI) analysis assists with identifying the appropriate action to take given uncertainties may exist. Identifying appropriate, quantifiable performance metrics that are consistent with the established objectives is a critical step in the planning stage, as monitoring and evaluating progress relies on these metrics.

Implementation: Facilitation and Monitoring
Implementing a conservation plan is the active phase of adaptive management. This stage involves dissemination of the plan and engaging in the outlined actions and monitoring of performance metrics. Social and political processes dominate the implementation stage and make communication and engagement with a broad audience of managers, policy-makers, and stakeholders a necessity. As such, the success of this stage is strongly influenced by the values and perceptions of stakeholders in support of the proposed action/s.

Evaluation and Learning
Learning is an integral component of the adaptive management process, particularly when it comes to evaluating and refining conservation actions. Iterative improvements can occur at different post-implementation stages as new information enables adjustments within the cycle. Evaluation of how well performance metrics are met determines weather or not the stated actions need refining, new actions need to be implemented, or if the objectives need to be changed in order to meet the conservation goal.

Figure 1.1 A schematic of the key phases of the adaptive management cycle in a conservation planning context (adapted from Grantham et al. (2009) to reflect aspects of this thesis, such as where value of information (VoI) analysis exists within this structured, higher- level process).

Within an adaptive management framework, conservation planning is inherently a value-driven process that begins with setting broad goals, such as: "conserve a region's biodiversity and ecosystem processes whilst retaining opportunities for development" (Tear *et al.* 2005). These goals are then transformed into more specific objectives and constraints such as: secure 20% of every kind of habitat, minimise opportunity costs for other users of the land or sea, maximise the number of species secured from extinction, and stay within a given budget. Given objectives and constraints, the next step is to elicit a plausible suite of strategies or actions from the relevant stakeholders and decision-makers. For conservation, some of the actions can be very straightforward such as: choose a number of locations to protect from human exploitation, or decide where to restore seagrass beds within a marine protected area (MPA). Others can be more general such as: how should we allocate funds across different national parks, how do we balance investment between seagrass restoration and seagrass protection (Saunders *et al.* 2017), and/or public education about the benefits of mangroves for local communities. Once the plausible actions or strategies have been chosen, they need to be assigned a cost (Naidoo *et al.* 2006; Bayraktarov *et al.* 2015) and connected to potential outcomes, such as the benefits they might deliver to help achieve the stated goals (Maxwell *et al.* 2014a; Brown *et al.* 2015b).

It is within the initial planning stage where decision-support tools and approaches play a vital role in prioritising between conservation actions (Runge *et al.* 2011) and establishing the appropriate evaluation metrics. Many frameworks have been developed to assist with decision-making for conservation planning (Schwartz *et al.* 2017), and many planning processes claim to enact adaptive management. Given our shortfalls in protecting biodiversity, and the state of species declines globally (Ceballos *et al.* 2017), more effort is needed to ensure decision-making frameworks and tools embedded in the adaptive management cycle, can be more effectively put into practice (Sutherland *et al.* 2004; Westgate *et al.* 2013).

This thesis focuses on a subset of conservation decision-making, spatial conservation prioritisation. Spatial conservation prioritisation is employed around the world (Wilson *et al.* 2009; Sinclair *et al.* 2018) and is the "biogeographic-economic activity of identifying important areas for biodiversity; where, when and how we might efficiently achieve conservation goals" (Kukkala & Moilanen 2013). I conceptualise spatial conservation prioritisations to date into three broad categories of decision-support approaches: 1) single attribute-based; 2) multiple attribute-based; and 3) problem-based (Figure 1.2). The following introductory review and synthesis is divided into three sections, the first of which describes these three approaches and how they have been operationalised within conservation prioritisation. This conceptualisation is supported with a toy-problem to illustrate how these different approaches vary. The section on "Limitations" discusses the advantages and disadvantages of these approaches. The final section outlines gaps in the research around decision-support for spatial prioritisation and defines the key objectives of this thesis addressed in each of the following research chapters.

1.1 Spatial Conservation Prioritisation

Attribute-based approaches that give each site a score based on a single attribute

Many examples of spatial prioritisation focus on producing two kinds of maps, one of assets and one of threats. The first set of maps describe the distributions of *assets*, such as various types of biodiversity (e.g. number of endemic species (Orme *et al.* 2005), habitat or species diversity (Roberts *et al.* 2002) or ecosystem services (e.g. pollination services, carbon sequestration (Turner *et al.* 2007; Atwood *et al.* 2017), or places with a number of species above a threshold (Myers *et al.* 2000)). The other set of maps shows us the distributions of threats. These could be maps showing the current rate of forest destruction, or the level of poaching or a measure of overgrazing (Halpern *et al.* 2009; Tulloch *et al.* 2015; Venter *et al.* 2016). These maps build on decades of research in scientific discovery and are critical to conservation and management. In many instances, these maps are promoted by individuals or organisations as "priority areas" for conservation (e.g. biodiversity hotspots (Myers *et al.* 2000)) and used to raise awareness and leverage funding towards these locations (Brooks *et al.* 2006) (see Figure 1.2A; Appendix A).

Approaches that combine multiple attributes but ignore dependencies between sites

Within this conceptualisation, "attribute-based" refers to the gathering of opinions about what combination of factors should drive priorities from a wide group of stakeholders and/or experts (Moilanen *et al.* 2009c). These factors could be any number of attributes related to issues such as: habitat quality, threatened species present, vulnerability, population sizes, environmental attributes (e.g. rainfall, or bathymetric complexity). There are two primary attribute-based approaches: scoring and criteria.

Scoring approaches

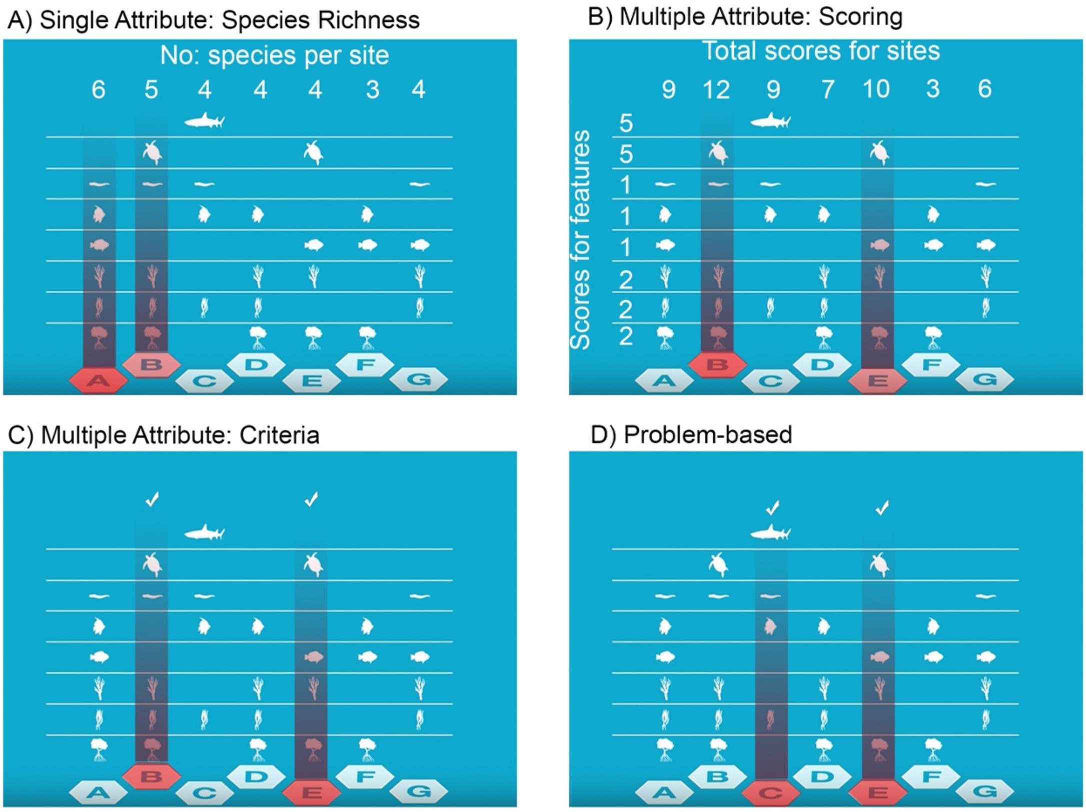
Scoring approaches have been used for decades as a way to integrate data into decision-making and priority setting. Scoring is carried out on a site-by-site basis, where each candidate site receives a score according to predetermined factor(s). In some cases, relative qualitative assessments (e.g. sites of low, medium, high condition) are converted into numbers and added together. In other instances, scores can be rescaled, weighted and added or averaged to provide a final score for a site (Kaufman *et al.* 2011). Typically, the site/s achieving the highest scores become the priority sites for conservation action (Ferrier & Wintle 2009) (see Figure 1.2B).

Criteria approaches

Similar to scoring, criteria-based approaches also rely on expert consensus to determine which attributes of a sites should guide the process of differentiating them from their surroundings. Criteria-based approaches play an important role in driving current global conservation policies. Many international organisations use this approach to identify global priorities for biodiversity conservation such as Ecologically or Biologically Significant Areas (EBSA) (Dunstan *et al.* 2016), or Key Biodiversity Areas (KBAs) (Eken *et al.* 2004). Species focused interest groups also use criteria to identify outstanding sites for biodiversity, such as BirdLife International's Marine Important Bird and Biodiversity Areas (IBAs) (BirdLife International 2010c). The primary difference between criteria and scoring approaches is that for the former, all sites meeting the criteria are considered equally important for conservation and there is no differentiation between the relative priority of different sites once criteria are met (See Figure 1.2B "scoring" compared to 1.2C "criteria").

Problem-based prioritisation

When a conservation problem is well-defined, it eventually leads to a mathematical representation with clearly expressed objective functions (e.g. to be minimized, maximised or set as constraints), defined actions, and models (or expert opinions) that connect the actions to the objectives. Importantly, within problem-based prioritisation, the value of a site is often dependent on the status of other sites selected. In spatially-explicit conservation planning, finding solutions to problems can be very complex and often need to be solved with the help of decision-support algorithms. For example, the popular spatial planning software Marxan (Ball *et al.* 2009) solves a mathematical problem formulated to meet targets for biodiversity whilst minimizing costs to users (See Figure 1.2D). This problem formulation appeals to a broad range of marine prioritization applications and has helped deliver spatial plans worldwide (Sinclair *et al.* 2018) including in the Coral Triangle (Beger *et al.* 2015; Jumin *et al.* 2017), the Mediterranean Sea (Giakoumi *et al.* 2011), and the Great Barrier Reef Marine Park (Fernandes *et al.* 2005). For other contexts, different algorithms (e.g. integer linear programming (Beyer *et al.* 2016)) and/or other economic approaches (e.g. cost-effectiveness analysis (Joseph *et al.* 2009; Klein *et al.* 2016)) can be used to solve well-defined problems.



Approaches to prioritization: a two-site problem

Approach:	Priority- driven by:	Priority site(s):
A) Single Attribute	Species richness	Sites A, B
B) Multiple Attribute: Scoring	Additive scores	Sites B, E
C) Multiple Attribute: Criteria	Presence of important habitats for: sea turtles mangroves	Sites B, E
D) Problem-based	Representation of every species once	Sites C, E

Figure 1.2 A hypothetical reserve design problem consisting in 7 sites and 8 biodiversity features showing how different approaches to selecting two sites will deliver different answers.

A toy problem

If we examine the two-site toy problem in Figure 1.2, which has seven sites and eight features across a hypothetical planning region, we can see how these approaches differ in terms of which two sites are selected as priorities.

- A single attribute-based approach (panel 1.2A) focused on identifying sites with the greatest species richness will deliver site A first as it has the most species, followed by Site B.
- A multiple attribute-based approach using scoring (panel 1.2B) will assign values to each feature, with important mobile species receiving the highest score. These scores are then

added together (without weightings in this instance) so sites site B and C become the two priority sites.

- A multiple attribute-based approach using criteria (panel 1.2C) for the presence of sea turtles and mangroves will select both sites B and E as equal priorities.
- A problem-based approach (panel 1.2D) approach aiming to represent all species will select sites C and E.

1.2 Limitations

The prevalence of both single and multiple attribute-based approaches in the conservation literature has prompted discussion about their methodological pros and cons (Ferrier & Wintle 2009; Game *et al.* 2013a; Klein *et al.* 2014). Mapping the distribution of biodiversity assets and/or threats is essential to conservation planning. Yet, while these maps appear to be useful in and of themselves for decision-making, and more so in combination with each other, they are ambiguous with respect to actions. Thus, attribute-based approaches are not equivalent to prioritisations. Attribute-based approaches often involve arbitrariness and hidden value judgements when constructing scoring rules, making repeatable and quantitative evaluations of how priorities are identified difficult. Further, criteria often rely on firm thresholds (e.g. a site must contain >1% of a population) that exclude many desirable sites for aspects of biodiversity or feasibility, and are ambiguous with respect to actions (Knight *et al.* 2007). However, we do note that both approaches can be used within more robust structured decision-making processes, such as multi-criteria decision analysis (MCDA) which uses criteria and scores to evaluate across scenarios in order to select the best course of action (Gregory *et al.* 2012; Runge *et al.* 2015). However, in the absence of a structured process, none of these approaches explicitly consider dependencies between sites, or the costs or feasibility of the actions when identifying sites for conservation - essential components of any prioritisation process (Game *et al.* 2013a).

While it is true that the single and multiple-attribute based approaches can be written as a well-defined mathematical problem, they rarely, if ever, are. If they were then the user would soon realise that the problem does not always reflect what objectives set out to achieve. For example, if we revert back to the two-site toy problem, if the objective is to maximise the number of species conserved, picking the two sites with the most species, or picking two sites that meet a set of criteria may satisfy the conditions of the attribute-based approach, but doing so ignores the overall objective of protecting every species by continuously missing site C (Figure 1.2).

Problem-based approaches, where the suite of actions delivers an outcome that is very different from the sum of the benefits of the individual action(s) taken in isolation, drive the fields of engineering, military operations, economics and transportation, but have yet to achieve widespread uptake throughout the field of conservation. There are myriad explanations for this gap including: scepticism due to uncertainties in data and ecological models (Keith *et al.* 2011), the costs in time and money required to construct and/or learn new decision-support tools, a fundamental resistance to the concept of "prioritising" nature that persists in public perceptions (Bottrill *et al.* 2008; Vucetich *et al.* 2017), and the failure of researchers to deliver science that is useful for decision-makers (Knight *et al.* 2008). Despite these limitations, problem-based prioritisations reflect decades of evolution in ecological theory and conservation practice and provide a promising future for global conservation driven by objective, transparent and efficient decision-making (Brown *et al.* 2015b).

1.3 Objectives and Significance

The primary aim of this thesis is to illustrate how problem-based approaches, grounded in decision-science, can deliver efficient and effective conservation priorities. Under the common theme of explicit problem definition, I address the following objectives:

1. To argue that clearly defined problems and decision-support approaches, such as value of information analysis (VOI), can assist with reducing key uncertainties (e.g. species movement patterns or demographic parameters) for conservation decision-making (***chapter 2***).
2. To evaluate the ability of a globally prominent "criteria-based" approach, BirdLife International's marine Important Bird and Biodiversity Areas (IBAs) (BirdLife International 2010b) to deliver benefits to biodiversity. Then, to provide best practice guidance on how to utilise such data in problem-based spatial prioritisation for protected areas (***chapter 3*****).**
3. To develop and test simple problem-based decision support tools to underpin decisions about 1) management policies for spatial zoning of marine resources (***chapter 4)*** and 2) strategies to prioritise conservation finance investments via Debt for Nature swaps for Small Island Developing States and coastal countries (***chapter 5***).

To achieve these objectives, I have focused on research gaps that can be classified under two streams related to problem-based spatial conservation prioritisation: one is focused on uncertainty and data, and the other is focused on tool development. These objectives and research streams are supported with four original research chapters (Figure 1.3).

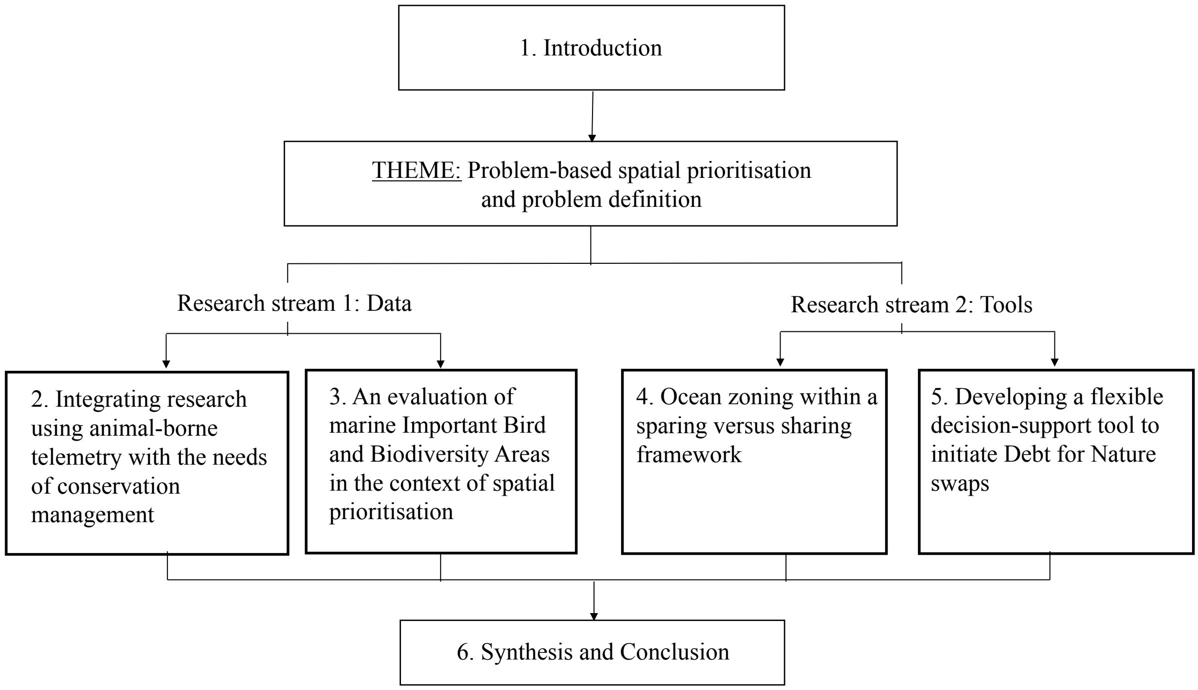


Figure 1.3 Schematic of the thesis organisation including the major theme, research streams and supporting chapters.

Stream 1: Data

Conservation decision-making is plagued with uncertainty, and the prevailing notion of many scientists is that more data are needed to reduce uncertainties before decisions can be made. Given data are costly to collect, and can risk delaying implementation to the detriment of the species or systems in need of conserving (Grantham *et al.* 2009a), differentiating between resolving those uncertainties that will influence a decision, versus those that will not, is a critical step for management (Runting *et al.* 2012; Maxwell *et al.* 2014). Value of Information (VoI) analysis can assist with this task (Figure 1.1) but the case for VoI has not been clearly presented to those collecting data with the motivation of informing conservation and management. My first research chapter focuses on dispelling the perception that more data will always lead to better decisions (objective 1, ***chapter 2***). I frame this argument towards the burgeoning field of animal telemetry where there is a noted trend of justifying the collection of animal movement and demographic data to inform conservation management, but where research studies are rarely aligned with management objectives (Campbell *et al.* 2015) (Figure 1.1). To address this, I construct a decision-tree which explicitly links data to actions, and emphasises Value of Information theory as an analysis tool to employ prior to collecting more data. While framed in the context of animal telemetry, the decision tree and argument can be applied to any field where data is collected to reduce uncertainties for conservation applications.

There are several prominent attribute-based approaches driving global conservation policy today. These include BirdLife International's marine Important Bird and Biodiversity Areas (IBAs)

(BirdLife International 2010b), and the International Union for the Conservation of Nature's (IUCN) Key Biodiversity Areas (KBAs), both of which use criteria-based approaches to identify priorities for biodiversity. While there has been some investigation into the utility of these datasets to represent important biodiversity in terrestrial systems (Di Marco *et al.* 2015), no one has evaluated their use within the marine realm. This is a critical research gap as countries race to expand their MPA estates. To address this, I evaluate how well marine IBAs, which meet a set of expert-based standardised criteria, represent other aspect of marine biodiversity and how different treatments of these data affect the efficiency of spatial conservation plans **(**objective 2**, *chapter 3***). I then develop best practice guidelines on how to use these data in problem-based spatial prioritisations and identify future research needs for IBAs and other criteria-based approaches delivering spatial data for use in conservation planning.

Stream 2: Tools

Decision-support tools exist in many forms: from participatory mapping (Brown & Weber 2011; Merrifield *et al.* 2013), to algorithmic software (Ball *et al.* 2009; Moilanen *et al.* 2009a), to complex-models that can simulate entire ecosystem level responses to changes in management (Pauly *et al.* 2000; Fulton *et al.* 2011; Metcalfe *et al.* 2015). Decision-support tools vary in their ability to inform the full spectrum of conservation challenges and several excellent synthesis papers exist that describe the roles and attributes of different decision-support tools for conservation (Sarkar *et al.* 2006; Stamoulis & Delevaux 2015; Schwartz *et al.* 2017). Many of these reviews focus on evaluating the ability of tools to incorporate important elements of conservation planning (e.g. connectivity or stakeholder processes). A key research gap is a comprehensive guide to matching appropriate tools to appropriate conservation problems. An important consequence of this gap is that decision-makers may construct a conservation challenge around the tool, rather than fitting the tool to the challenge. While constructing such a guide requires more than this aspect of my thesis can provide, I begin to fill this gap by developing two decision-support tools that solve very different, well-defined problems.

The first tool cost-effectively allocates budgets across different management actions in order to deliver the largest benefit. To accomplish this, I first develop a framework for optimal ocean zoning where three common management actions can be taken: biodiversity protection, fisheries management, and doing nothing (objective 3, ***chapter 4)***. The problem formulation aims to maximise biodiversity (defined as standing stock fish biomass) while meeting a minimum fisheries harvest constraint for a fixed budget. The decision-support tool is a simple process-model that produces an optimal proportional allocation of the seascape for different management budgets. The primary focus of this tool is to understand when the optimal management strategy shifts in relation to financial constraints. The second tool moves beyond prioritising between actions (Possingham *et al.* 2015; Saunders *et al.* 2017), to answer the question of where to invest between places to take a specific action (Wilson *et al.* 2006). To do this, I developed a bespoke cost-effectiveness calculator for a specific end-user, The Nature Conservancy, who is interested in a particular type of conservation finance mechanism called “debt for nature” swaps. I develop a four-part prioritization protocol that returns a ranked order of countries based on their ability to cost-effectively deliver biodiversity benefits whilst accounting for threats, costs, and the probability the action will succeed (objective 3, ***chapter 5***).

What follows are four partly dependent chapters illustrating different parts of my thesis on the formulation of applied nature conservation problems. While they are best read in order, they can also be read independently.

2 Integrating research using animal-borne telemetry with the needs of conservation management

This chapter is reproduced from the following publication with some alterations to format and structure:

McGowan J, Beger M, Lewison R, Harcourt R, Campbell H, Priest M, Dwyer R, Lin H, Lentini P, Dudgeon C, McMahon C & HP Possingham. 2016. Integrating research using animal-borne telemetry with the needs of conservation management. *Journal of Applied Ecology*. doi: 10.1111/1365-2664.12755.

2.1 ABSTRACT

Animal-borne telemetry has revolutionised our ability to study animal movement, species physiology, demography and social structures, changing environments and the threats that animals are experiencing. While there will always be a need for basic ecological research and discovery, the current conservation crisis demands we look more pragmatically at the data required to make informed management decisions. Here, we define a framework that distinguishes how research using animal telemetry devices can influence conservation. We then discuss two critical questions which aim to directly connect telemetry-derived data to applied conservation decision-making: (i) Would my choice of action change if I had more data? (ii) Is the expected gain worth the money and time required to collect more data? To answer questions about integrating telemetry-derived data with applied conservation, we suggest the use of value of information (VoI) analysis to quantitatively assess the return-on-investment of animal telemetry-derived data for conservation decision-making.

2.2 INTRODUCTION

The rapid ascent of animal-borne telemetry research reflects the ability of this approach to improve our understanding of fundamental ecology, enhance monitoring of the planet's natural resources and inform conservation practices (Hussey *et al.* 2015; Kays *et al.* 2015). What is remarkable about animal-borne telemetry is its ability to illustrate how individuals, ranging from bees to whales, interact with each other and the natural environment and reveal information about species habitat use, movement patterns, behaviour, physiology and the environment they inhabit (Cooke *et al.* 2004). These studies have documented ocean-wide dispersal events (Block *et al.* 2011), identified the use of unexpected habitats (Raymond *et al.* 2014), fundamentally changed our understanding of physical processes in the natural environment (Roquet *et al.* 2013), and revealed unknown life history characteristics of threatened and cryptic species (Davidson-Watts *et al.* 2006). It is indisputable that animal-borne telemetry has enriched our understanding of the natural world and the animals that inhabit it.

With these advances, there comes an opportunity to use animal telemetry-derived data to combat global species declines (Ceballos *et al.* 2015). Much of the published literature using telemetry technologies claim conservation implications, yet the link between many of these studies to direct conservation actions remains tenuous (Campbell *et al.* 2015; Jeffers & Godley 2016). Here, we challenge the assumption by many scientists that more data will invariably lead to better management and suggest an evaluation of the return-on-investment from research using animal-borne telemetry devices (Runge *et al.* 2011; Maxwell *et al.* 2014a).

Given the potential of telemetry-derived data to inform resource management and conservation, and the various costs involved in collecting these data (e.g. financial costs of equipment and salaries, impact on mortality and reproduction of animals involved (Cooke *et al.* 2004; McMahon *et al.* 2012), it is essential to evaluate the conservation benefit of these research techniques. As conservation science is an explicitly applied field, our aim is to differentiate between telemetry-derived data that improves ecological knowledge with implications for broad conservation efforts versus data that have direct impact on conservation decision-making. Our objective is to encourage researchers utilising telemetry technology with an underlying conservation rationale to target their research towards gathering information that is more likely to change actions and maximise species persistence.

2.3 DIFFERENTIATING CONSERVATION IMPACTS

The use of telemetry devices to monitor free-ranging animals can affect species conservation in many ways. To differentiate these impacts according to conservation specificity and time-scale of impact, we draw from a conceptual model developed for ecological monitoring activities (Possingham *et al.* 2012). We present this framework to distinguish how animal-borne telemetry studies, specifically, can influence conservation. We frame this discussion around the distinctions made among six types of graduated impact, ranging from long-term and diffuse to short-term and direct (Fig 2.1).

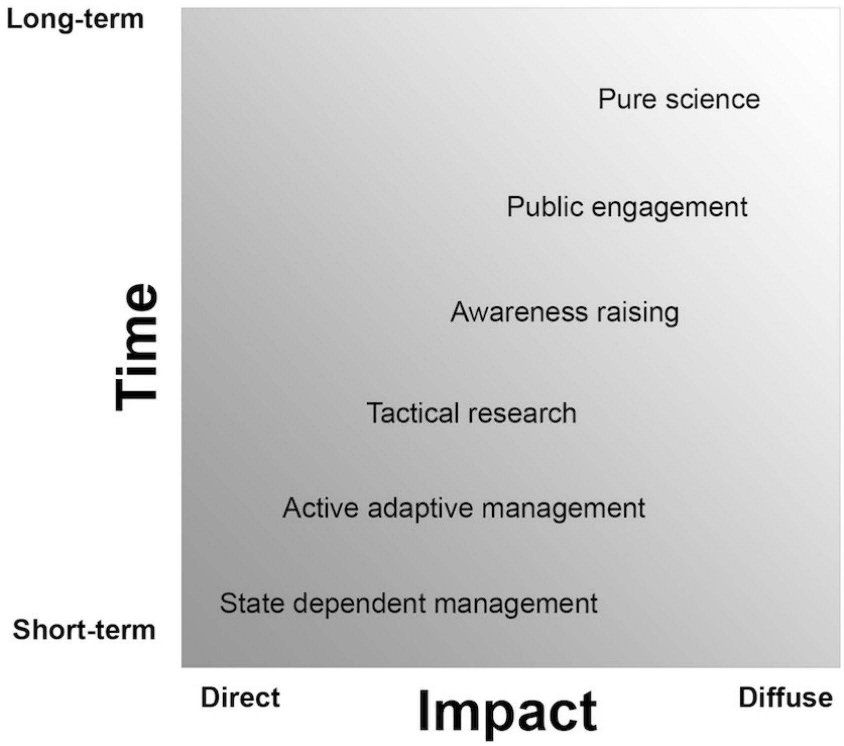


Figure 2.1 A categorization of research and monitoring activities in terms of their ability to deliver conservation outcomes. The impacts can be visualized along a gradient from direct to diffuse and occurring in the near- or long-term time-scales.

Pure scientific research

Discovering new facets of life history, biology or ecology motivates many scientists conducting animal-borne telemetry research. The driver of this work is often pure ecological enquiry (Hart & Hyrenbach 2009; Donaldson *et al.* 2014). Through exploratory science, telemetry-derived data can generate novel findings or improve existing knowledge. It is possible that this knowledge will indeed influence conservation actions at some point. For example, radio-tracking studies in the UK revealed that protected species of *Pipistrellus* bats, which cannot be distinguished through observational studies, actually exploit distinct species-specific habitats and thus require individually tailored conservation measures (Davidson-Watts *et al.* 2006). New insights of this nature will certainly change conservation goals and thinking, yet the impact is often serendipitous, diffuse and over long time scales.

Engaging the public and leveraging effort

Unlike other forms of monitoring, where members of the public can easily participate and volunteer in the data collection process (i.e. citizen science), the tagging and tracking of individuals requires special expertise and can limit the role of the public to be intimately involved in data acquisition. Although public engagement would rarely be the sole purpose of a telemetry-based animal study, the application is exciting and often engages and captivates a broad public audience through social media campaigns (http://www.ocearch.org) and cultural events. The astonishing behaviours revealed through tracking individuals, such as the recent discovery of the near 2,500 km long-distance American eel (*Anguilla rostrate)* migration (Beguer-Pon *et al.* 2015), can raise species profiles and promote public awareness of conservation issues. Although changing perceptions and improving commitment to nature is an important component of a society's willingness to commit resources to species conservation, the process can be unpredictable.

Raising awareness for the public and policy makers

Visual aids, such as maps, can be vital knowledge brokering tools for issues of conservation concern (Hebblewhite & Haydon 2010). Maps of animal movements and habitat use provide evidence of the ecological connectivity between disparate geographies. These findings provide visual support to unify politically diverse regions or groups towards a common conservation goal and encourage cross-boundary collaboration. For example, telemetry-derived data reveal the movements of long-distance migrants that connect countries, continents and hemispheres. These studies underpin multi-lateral initiatives such as the East Asian Australasian Flyway (http://www.eaaflyway.net/), the Convention for Migratory Species (www.cms.int), as well as species focused initiatives such as sea turtle conservation under the Coral Triangle Initiative for Coral Reefs, Fisheries, and Food Security (Beger *et al.* 2015).

Tactical research

Tactical research is research that is not of immediate use to solve a management problem, but is prioritized because a researcher uses their experience to determine that it is likely to be important in the near future. For example, we know that many animals experience different and varied magnitudes of threats across migration routes. Therefore, the success of an action taken in a nesting site may prove futile if threats at important stopover, bottleneck or refugia sites are not identified and mitigated. Committing resources to monitor and learn about unknown spatial processes using telemetry technologies, such as identifying migratory pathways, can determine what state- and time-dependent actions will deliver the greatest benefit to the population's viability (Runge *et al.* 2014; Cooke *et al.* 2016). However, there is a point where investing in tactical research returns marginal benefits to conservation decision-making relative to solving urgent problems (Possingham *et al.* 2012).

Active adaptive management

Telemetry-derived data can also identify which conservation actions to take -or not take- within the adaptive management framework (Holling 1978; McFadden *et al.* 2011). Adaptive management capitalises on opportunities to improve the effectiveness of management strategies as new knowledge is gained (McCarthy & Possingham 2007; Grantham *et al.* 2009b). This may be a "passive" process, which involves reviewing the performance of past or current actions to alter future actions, or "active", where there is a conscious effort to balance knowledge acquisition and conservation action. These management programs maintain well-established monitoring protocols and are capable of responding to observed changes in populations. For example, biotelemetry research on anadromous salmon has led to an improved understanding of mortality events from catch and release fishing interactions, and physiological factors influencing spawning failure, which in turn justify restrictions on fished populations (Cooke *et al.* 2012).

State-dependent management

State-dependent management requires monitoring the state of a system or population to determine how best to manage it. State-dependent management, such as quota setting for harvestable species is the most direct way for telemetry derived-data to influence species conservation. These research techniques are already powering new approaches that integrate individual-based movement information and decision theory. For instance, Dynamic Ocean Management (DOM) is an approach that changes in space and time in response to the shifting nature of the ocean, the animals in it, and its users based on the integration of current biological, oceanographic, social and/or economic data (Maxwell *et al.* 2015). Some of these applications use telemetry-derived data to alter spatial management over short timeframes (Lewison *et al.* 2015). This has benefits for mitigating dynamic threats such as bycatch from seasonal fishing effort (Hobday *et al.* 2010).

2.4 THE VALUE OF INFORMATION TO DECISION-MAKING

It is clear that many studies using animal-borne telemetry have the potential to inform conservation. We have discussed several classes of impacts delivering important benefits to society and species. As with all research efforts, one would want to know both the quantifiable costs and expected benefits from the research. Here, we present a framework that can allow researchers to ask: "If that effort could have been placed directly into management and implementation, would the species be better off?"

We focus the remaining discussion on how to improve the conservation return-on-investment in research using animal-borne telemetry and argue that to do so, the ecological knowledge derived from these studies needs to inform and guide management actions (McDonald-Madden *et al.* 2010). Several excellent reviews discuss the potential of using telemetry technology for species management (Cooke 2008; Godley *et al.* 2008; Metcalfe *et al.* 2012; Hays *et al.* 2016) and policy (Barton *et al.* 2015). Yet, these reviews underemphasise the importance of defining clear links from research to actions. Similarly, Allen and Singh (2016) recently developed the Movement Management Framework - a first attempt to formally integrate movement information into a decision-making process. However, the authors overlooked critical aspects of modern decision science, namely the importance of setting explicit quantitative objectives, and how movement data can help screen and select actions at the beginning of the planning process based on their associated costs, social and economic acceptability and likelihood of success (McGowan & Possingham 2016). Figure 2.2 highlights two key questions that serve to directly connect research using animal-borne telemetry to applied conservation decision-making.

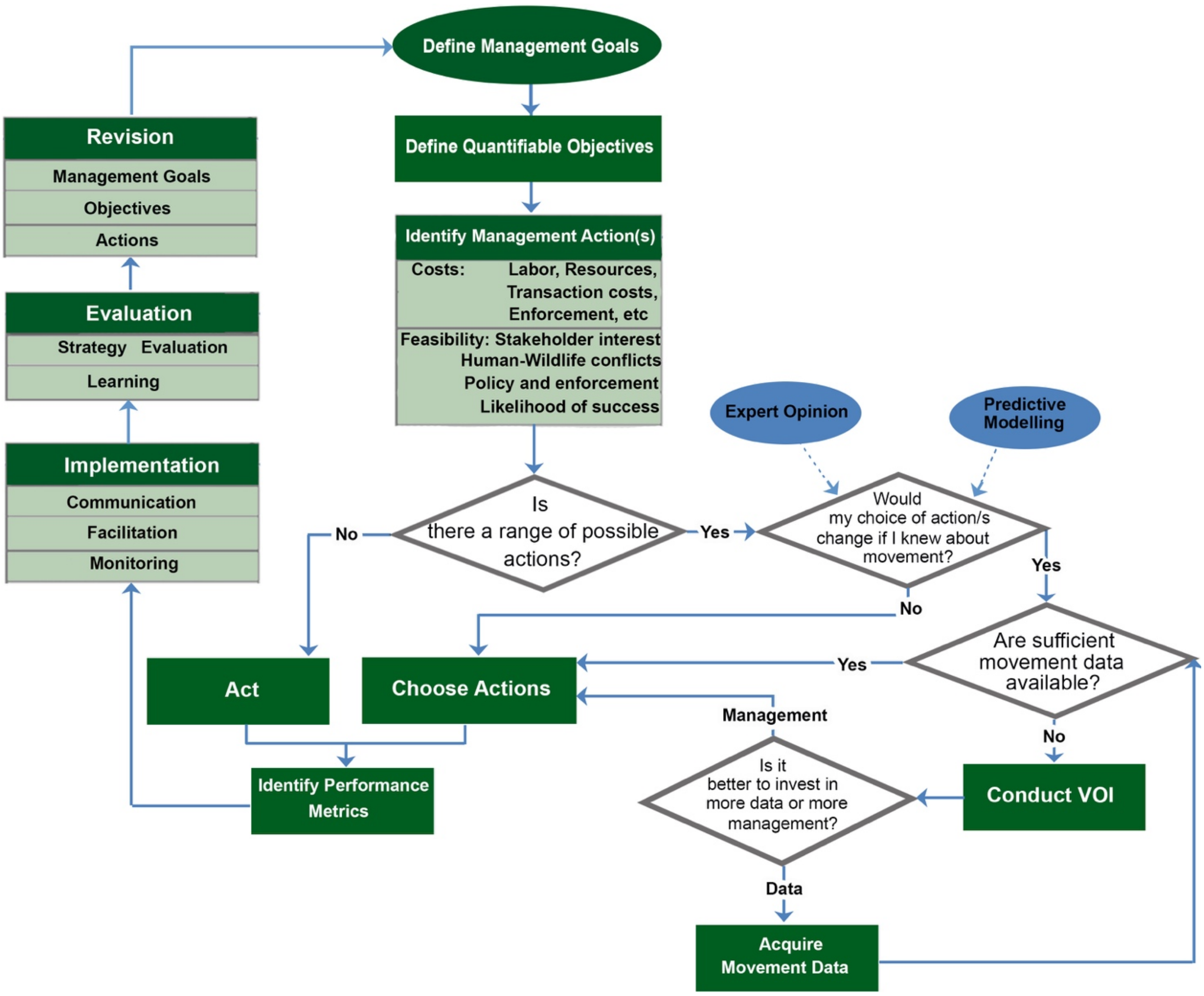


Figure 2.2 A decision tree to assist with evaluating the conservation return-on-investment from acquiring telemetry-derived data for decision-making. After conservation goals, objectives and possible actions are defined, one must ask if certain types of data, such as animal movement, will affect the selection of management action(s). If yes, then one should evaluate existing data for quality and new data should only be pursued after a value of information (VOI) analysis reveals that the benefit of that new data (e.g. reducing uncertainty) outweighs the benefit relative to more management. Adapted from previous versions of the Movement Management Framework of Allen and Singh (2016) and McGowan & Possingham (2016).

Would my choice of action change if I had more data?

To know this, high level goals must first be translated into quantifiable (also termed "operational" objectives (Katsanevakis *et al.* 2011)) so that actions can be evaluated based on their ability to improve the overall benefit of the conservation intervention (Tear *et al.* 2005). Such objectives should be specific, measurable, achievable, realistic and time bound, or "SMART" (Katsanevakis *et al.* 2011; Stelzenmuller *et al.* 2013). Table 2.1 provides some examples of how the results from animal research using telemetry technology enables managers to choose between conservation actions that abate threats to population growth rates, habitat quantity, quality, connectivity, and deliver outcomes for specific objectives. We also note that telemetry techniques can play a major role in reducing uncertainty about threats themselves, which may be a necessary step before mitigating actions can be prescribed. However, we stress that just because there is uncertainty in an ecological variable, parameter, or threatening process, it does not mean that reducing that uncertainty facilitates better decisions or leads to better management (Runge *et al.* 2011).

We draw from a trend in the movement ecology literature to track individual occupancy within and around established protected areas to illustrate this point. The rationale underlying these studies is often to inform protected area design, as data reveal that changes are needed to better capture the movements and habitat-use of the tracked population. A fundamental yet often ignored aspect of these studies is that once established, protected area boundaries are very slow to change. Given that planning horizons can be decades long (Grantham *et al.* 2009b), these findings likely fall within the diffuse impact category of raising public concern and awareness about protection deficiencies rather than delivering direct benefits in the near-term.

While telemetry-derived data may reveal major gaps in contemporary conservation practices, a mechanism to take the recommended action is also required to achieve direct influence over conservation. For example, if the objective is to maximize the population size of a marine species, money spent on tracking individuals around a protected area could be more optimally spent on threat mitigation, such as fisheries regulations outside the boundaries, nesting/breeding site patrols, or bycatch reduction strategies. From a decision science perspective, we don't necessarily need to know the movements of individuals to best achieve the objective.

Table 2.1 Examples of linkages between classes of threats, conservation objectives and action informed by animal telemetry-derived data

Threat	Class	Objective	Actions	Animal telemetry-derived data tell us:
Linear infrastructure e.g. road, rail, power lines	a) Demographic, animals are killed by collisions b) Connectivity, animals avoid crossing linear features	a) Reduce collisions b) Improve colonization or genetic exchange	a) Fence entire road segments or increase visibility b) Build crossing structures	a) Which linear feature segments are most frequently crossed b) Where animals are more likely to cross
Anthropogenic barriers in rivers e.g. dams, and weirs	a) Connectivity, animals need to move between feeding and breeding grounds b) Habitat, altered flow decreases suitable breeding habitat	a) Increase the fraction of individuals able to reach their breeding grounds b) Increase the area of suitable breeding habitat	a) Prioritise the location of fish passage options b) Regulate flow regime upstream of barriers to increase habitat availability and quality	a) Which barriers prevent the most fish from passing b) Which habitats are most used for breeding
Point infrastructure (e.g. electricity pylons, communication towers, or wind farms)	Demographic, structures kill threatened species (vultures, orange-bellied parrot, migratory microbats)	a) Not cause unacceptable harm to a population b) Reduce the likelihood of threats at an existing site	a) Approve location of point infrastructure b) Modify timing of operations (e.g. wind turbines)	a) The number of individuals passing through and residency time at a site for key species b) The time at which individuals pass through a site
Mortality from extractive industry (i.e. fisheries)	Demographic, interactions result in harm or death	Reduce incidental mortality (e.g. bycatch rates)	Gear restrictions or spatial closures	When and where non-target individuals forage
Human-wildlife conflict	a) Demographic; persecution and culling impact on survival b) Habitat exclusion from key breeding or foraging areas	a) Reduce frequency of negative interactions with humans b) Maximise area of important habitats which species can access	a) Install barriers to protect communities b) Introduce compensatory schemes to encourage coexistence	a) Frequency of wildlife encroachments b) When and where important breeding and feeding areas are
Disease	Demographic; mortality from pathogen transfer	Understand how disease spreads through population	Restrict the movement of disease vectors	Where and when carrier individuals move
Illegal harvest or poaching	Demographic; interactions result in harm or death	Decrease poaching rates	Optimise patrol routes	Spatial and temporal distribution of poaching-related mortality
Invasive species	a) Demographic, mortality from invasive predators b) Habitat, exclusion by introduced competitor	a) Increase probability of persistence of prey species b) Reduce area of occupancy of competitor	a) Control of invasive predator population b) Control of invasive competitor	a) Location and timing for culling operations to have greatest impact b) Home range and encounter probability of traps or bait

Is it better to invest in more data or more management?

There are several taxonomies of uncertainty affecting how management decisions are made however we focus our discussion on epistemic, and structural and/or parametric uncertainty in the models we use to understand how natural systems work (Regan *et al.* 2002; Runge *et al.* 2011). Our imperfect knowledge of natural systems often leads to the assertion that a greater understanding of ecological processes, spatial data and/or detailed parameters will always improve decisions. However, from a conservation decision-making perspective, investments in advancing basic ecological science to aid conservation can redirect resources away from management. Given this quandary, how does one decide whether or not to invest in more data collection? We can resolve this using an approach relatively new to ecology and conservation – value of information analysis (VoI), a quantitative tool for incorporating uncertainty into decision making (Canessa *et al.* 2015; Williams & Johnson 2015). Value of information analysis can be used to examine the trade-off between the ability of new information to reduce decision uncertainty and the costs of collecting more data; which uncertainties may be most important to reduce in order to improve gains in management outcomes (Runge *et al.* 2011); or what the financial value of gaining new information is worth to management (Maxwell *et al.* 2014).

Maxwell et al. (2014) provide an excellent example of using value of information analysis for wildlife conservation. In this study, the authors considered several possible actions that can be taken to maximize the growth rate of a declining koala *Phascolarctos cinereus* population. These include building wildlife passages to avoid vehicle collisions, allocating resources to dog owners to prevent attacks, and securing koala habitat. The management decision relied on uncertain information about demography and movement so one could easily have argued for a tracking study to inform the decision. However, investing in telemetry devices for research *a priori* would have been misguided as the value of information analysis showed optimal management decisions were not sensitive to these uncertainties, but were primarily driven by the cost-efficiency of the actions and the management budget (Maxwell et al. 2014).

Improving the return-on-investment of animal-borne telemetry for conservation decision-making

To date, there are only a few examples of using value of information analysis to inform management decisions, and even fewer using telemetry–derived data. The potential benefits from this field are rarely being systematically incorporated into conservation decision-making or spatial prioritisation (Mazor *et al.* 2016a). While there will always be a need for basic ecological research and discovery, the extent of the current conservation crisis demands we look more pragmatically at the data required to make decisions. Given the global investment in telemetry devices for threatened species, we have an ethical and practical obligation to maximise this investment's benefit to conservation. To improve the conservation return-on-investment in these techniques, we need new tools and frameworks to effectively link the growing catalogue of animal telemetry-derived data to conservation and management, especially when species persistence depends on complex decision contexts with multiple trade-offs. Uncertainty will never be fully removed from conservation and management, but value of information and other approaches that explicitly evaluate the value of science to decisions should play an increasingly important role.

3 An evaluation of marine Important Bird and Biodiversity Areas (IBAs) in the context of spatial prioritization

This chapter is reproduced from the following publication with some alterations to format and structure:

McGowan J, Smith R, di Marco M, Clarke R & HP Possingham. 2017. An evaluation of marine Important Bird and Biodiversity Areas in the context of spatial prioritization. *Conservation Letters*. doi: 10.1111/conl.12399.

3.1 ABSTRACT

Important Bird and Biodiversity Areas (IBAs) are sites identified as globally important for bird species conservation. Marine IBAs are one of the few comprehensive multi-species datasets available for the marine environment, and their use in conservation planning will likely increase as countries race to protect 10% of their territorial waters by 2020. We tested 15 planning scenarios for Australia's Exclusive Economic Zone to guide best practice on integrating marine IBAs into spatial conservation prioritization. We found prioritizations based solely on habitat protection failed to protect IBAs, and prioritizations based solely on marine IBAs similarly failed to meet basic levels of marine habitat representation. Further, treating all marine IBAs as irreplaceable sites produced the most inefficient plans in terms of ecological representativeness and protection equality. Our analyses suggest that marine spatial planners who wish to use IBAs treat them like any other conservation feature by assigning them a specific protection target.

3.2 INTRODUCTION

Spatial conservation prioritization is the process of identifying priority sites for conservation actions in space and time (Moilanen *et al.* 2009b). When designing marine protected area (MPA) networks, priority areas are selected based on several core principles: Representation- ensuring all aspects of biodiversity receive protection (e.g. setting targets for species distributions, abundances, ecological processes, habitats, unique features and/or cultural sites); Adequacy – ensuring what is protected is sufficient to help biodiversity persist through time; and Cost-efficiency – ensuring the feasibility of the conservation action has been accounted for and the social, economic and/or political impacts minimized (Ban & Klein 2009; Brown *et al.* 2015). Two additional concepts aid spatial prioritization: 1) Complementarity - selecting suites of sites that collectively ensure all conservation features receive protection, a concept that underpins the principle of representation (Moilanen *et al.* 2009b); and 2) Irreplaceability - the contribution of a site to meet a pre-established set of biodiversity conservation targets, or the extent to which target achievement is compromised if the site is lost (Pressey *et al.* 1994; Ferrier *et al.* 2000). Spatial prioritization is typically performed with freely available software, such as Marxan and Zonation, which operationalize the principles and concepts described above (Moilanen *et al.* 2009a).

A practical challenge for marine spatial prioritization is the paucity of data available at relevant planning scales. Time and resource constraints often hinder collating comprehensive spatial biodiversity inventories across a planning region. It is now considered common practice to use habitat types as broad biodiversity surrogates (Dalleau *et al.* 2010; Sutcliffe *et al.* 2015) in lieu of more detailed ecological and biophysical data (Ward *et al.* 1999). Practitioners routinely rely on publicly available spatial datasets of habitats and species ranges when conducting prioritizations of where to place protected areas (e.g. IUCN Red List Spatial Data; UNEP-WCMC).

Seabirds are believed to be important indicators of marine ecosystem function (Furness & Camphuysen 1997; Zacharias & Roff 2001) and seabird distributions play an important role in identifying priority areas for marine conservation (Nur *et al.* 2011; Lascelles *et al.* 2012; Ronconi *et al.* 2012; McGowan *et al.* 2013; Bax *et al.* 2016). The seabird conservation community is a prominent and well-organized collective who, driven in large part by the efforts of BirdLife International (http://www.birdlife.org), strive to make comprehensive global seabird data available. One such dataset comes from BirdLife International's Important Bird and Biodiversity Area (IBA) program which uses a threshold-based approach to identify priority sites on land and at-sea based on fulfilling one or more criteria related to the presence of: (A1) threatened species, (A2) restricted-range species, (A3) biome-restricted species, and (A4) large congregations of individuals from one or more species (BirdLife International 2010a) (Table 3.1).

Table 3.1 Standardized Criteria for Important Bird and Biodiversity Areas

BirdLife International IBA Criteria
A1- Globally Threatened Species: the site regularly holds significant numbers of a globally threatened species, or other species of global conservation concern;
A2- Restricted-range Species: the site is known or thought to hold a significant component of the group of species whose breeding distributions define an Endemic Bird Area or Secondary Area;
A3- Biome-restricted Assemblages- The site is known or thought to hold a significant component of the group of species whose distributions are largely or wholly confined to one biome;
A4- Congregations: **i) Site known or thought to hold, on a regular basis,>= 1% of a biogeographic population of a congregatory waterbird species.** **ii) Site known or thought to hold, on a regular basis, >= 1% of the global population of a congregatory seabird or terrestrial species.**

iii)	**Site known or thought to hold, on a regular basis, >= 20,000 waterbirds or >+ 10,000 pairs of seabirds of one or more species**
iv)	**Site known or thought to exceed thresholds set for migratory species at bottlenecks sites**

IBAs are intended to delineate sites that are essential for the survival of the birds (O'Dea *et al.* 2006) and subsequently, the biodiversity they represent (BirdLife International 2010b). Globally, more than 12,000 IBAs have been identified on the land and sea, and an additional 2,000 candidate sites have been proposed for the global oceans (BirdLife International 2010c; Lascelles *et al.* 2016). Candidate marine IBAs consist of seaward extensions of seabird breeding colonies, non-breeding coastal congregations, migration bottlenecks and pelagic distributions. The IBA dataset provides spatially- explicit ecological knowledge beyond species ranges, and is one of the most comprehensive species-specific datasets available for the oceans.

As countries race to meet their commitments to conserve 10% of their Exclusive Economic Zones (EEZs) in MPAs by 2020, marine IBAs are expected to play a significant role in achieving the Convention on Biological Diversity's protected area network goals (BirdLife International 2010b). Hence there is an urgent need to understand how best to use these sites in future planning for our global protected area estate. To date, there are no specific guidelines on how best to use marine IBAs for spatial conservation prioritization. Here we use a planning exercise for Australia's EEZ, an area of 6.0 million km^2, as a way to examine 15 different planning approaches based on using habitat data with three different treatments of marine IBAs. We evaluate the resulting spatial plans with respect to their cost-effectiveness and how equally they distribute protection across biodiversity features. We ask the following questions: (1) Are marine IBAs (including candidate sites) effective surrogates for benthic and pelagic marine habitats and to what extent does selecting sites for those habitats also represent IBAs? (2) How does treating marine IBAs as irreplaceable sites influence spatial planning outcomes? (3) What is the best way to integrate IBAs with other biodiversity features when identifying MPA networks? Thus, our analysis aims to identify the best ways to include marine IBAs in spatial prioritization, rather than identify where new MPAs in Australia should be located.

3.3 METHODS

Spatial Data

Marine ecoregions provide a spatial framework for planning that captures unique biogeographic assemblages, including biophysical and oceanographic processes (Spalding *et al.* 2007). We use provincial and meso-scale marine bioregions of Australia (Commonwealth of Australia 2006) to stratify seafloor geomorphic features (Harris *et al.* 2014) across seven depth classes and create a dataset of over 1600 conservation features covering Australia's EEZ. We also included the Australian marine IBA inventory provided by BirdLife International (see https://maps.birdlife.org/marineIBAs/default.html). This inventory consists of 69 marine IBAs (mean size = 34 km2), triggered by 27 seabird species, most of which are bird congregation sites that fulfil Criterion A4 (Table 3.1). In addition, there are 67 candidate IBAs (mean size = 13,000 km2) triggered by 25 seabird species that fulfil Criterion A1 and A4 (Figure 3.1). While provisional, we included candidate IBAs in our analyses because they are published online and distributed for use in conservation planning.

Spatial analysis

MPA planners commonly rely on computational algorithms to identify priority areas for protection (Sarkar *et al.* 2006). We used the decision-support tool Marxan (Ball *et al.* 2009a) to identify groups of 10km x 10km planning units that meet pre-established conservation targets, while minimizing the overall cost of the identified network. We use a proxy of opportunity cost for the lost revenue from fishing or other industries that would be displaced should a site become protected (Ban & Klein 2009). This proxy cost decreases with the distance of each planning unit to the nearest Australian port (available from www.data.gov.au)(Mazor *et al.* 2014). We generated 100 Marxan solutions, based on 1 million iterations each, and selected the solution with the best score for comparisons across scenarios. We assess a suite of 15 typical approaches for setting marine conservation priorities based on three treatments of the IBA data (Table 3.2).

Table 3.2 Scenario matrix to derive 15 spatial planning approaches for habitats and IBAs.

		Treatment 1: IBAs as features (N=136)			
		100%	Sliding scale	20%	0%
Habitats (features N=1659)	20%	1b	1d	1c	1a
	0%	0	1f	1e	-
		Treatment 2: IBAs as core habitats (N=33)			
	20%	-	2a	2c	-
	0%	-	2b	2d	-
		Treatment 3: IBAs as abundances (N=33)			
	20%	-	3a	3c	-
	0%	-	3b	3d	-

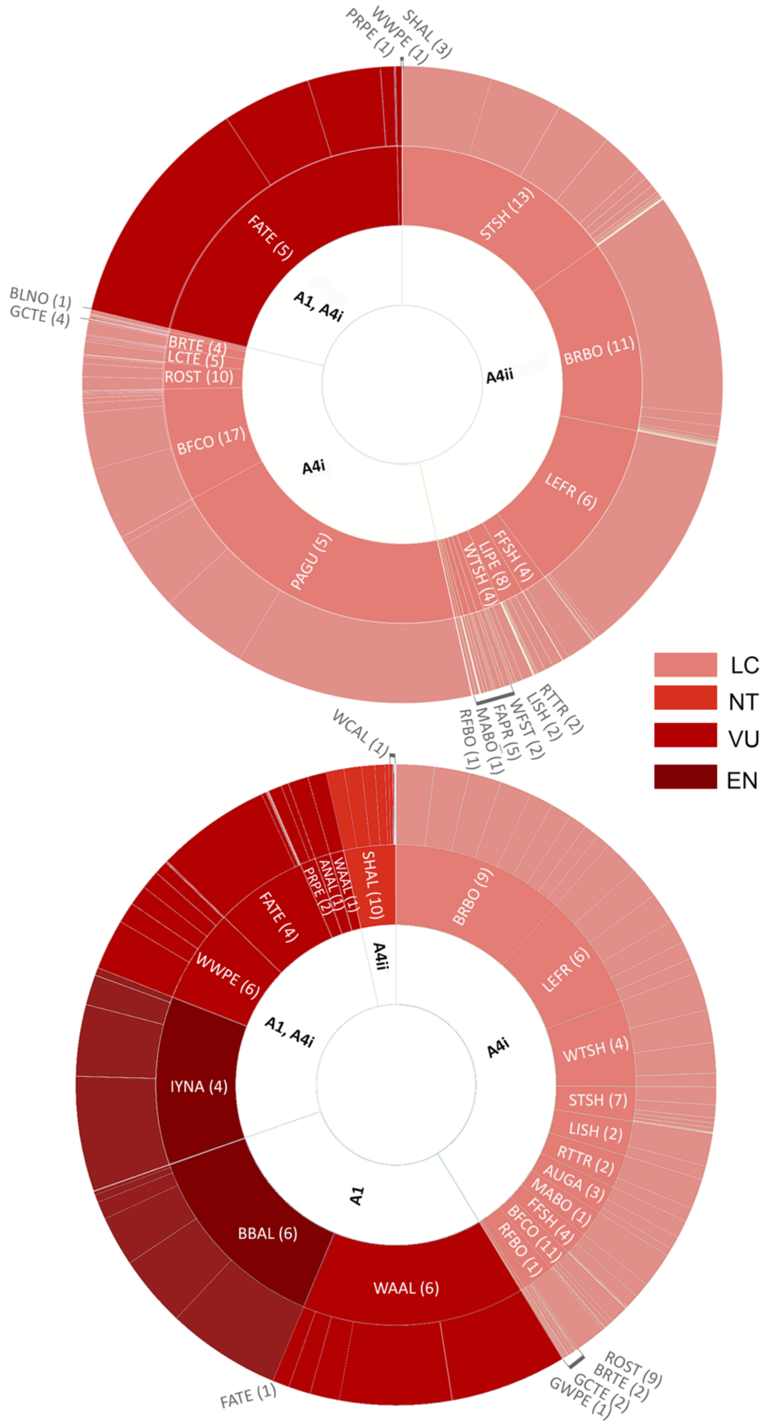


Figure 3.1 A hierarchical visualization of the marine IBA inventory by area for a) confirmed and b) candidate IBAs. The inner circle categorizes IBAs by their underlying criteria. The middle circle shows the species meeting these criteria and the outer circle shows the number of IBAs (also in parentheses next to species name) and the relative size of each IBA per species and criteria combination. The graduated colours correspond to the species IUCN red-list category to illustrate the threat level for trigger species. For example, in a) there are 5 IBAs for the Fairy Tern. Species abbreviations as follows: AUGA-Australasian Gannet (*Morus serrator*) ; ANAL-Antipodean Albatross *(Diomedea antipodensis)*; BBAL-Black-browed Albatross *(Thalassarche melanophrys*); BFCO-Black-faced Cormorant *(Phalacrocorax fuscescens)*; BLNO-Black Noddy *(Anous minutus)*; BRBO-Brown Booby *(Sula leucogaster)*; BRTE-Bridled tern *(Onychoprion anaethetus)*; CDPE-Common Diving Petrel *(Pelecanoides urinatrix)*; FAPR-Fairy Prion *(Pachyptila turtur)*; FATE-Fairy Tern *(Sternula nereis);* FFSH-Flesh-footed Shearwater *(Ardenna carneipes)*; GCTE-Greater Crested Tern *(Thalasseus bergii)*; GWPE-Great-winged Petrel *(Pterodroma macroptera)*; IYNA-Indian Yellow-nosed Albatross *(Thalassarche carteri)*; LCTE-Lesser Crested Tern *(Thalasseus bengalensis)*; LEFR-Lesser Frigatebird *(Fregata ariel)*; LIPE-Little Penguin *(Eudyptula minor)*; LISH-Little Shearwater *(Puffinus assimilis)*; MABO-Masked Booby *(Sula dactylatra)*; PAGU-Pacific Gull *(Larus pacificus)*; PRPE-Providence Petrel *(Pterodroma solandri)*; RFBO-Red-footed Booby *(Sula sula)*; ROST-Roseate Tern *(Sterna dougallii)*; RTTR-Red-tailed Tropicbird *(Phaethon rubricauda)*; SHAL-Shy Albatross *(Thalassarche cauta)*; STSH-Short-tailed Shearwater *(Ardenna*

tenuirostris); WAAL-Wandering Albatross *(Diomedea exulans)*; WFST-White-faced Storm Petrel *(Pelagodroma marina)*; WTSH-Wedge-tailed Shearwater *(Ardenna pacificus)*; WWPE-White-winged Petrel *(Pterodroma leucoptera)*.

Marine IBA Treatments

We treated marine IBA polygons in three ways that are typical of how species data are used in spatial prioritizations (Figure 3.2). First, we considered each individual IBA as a unique conservation feature (Treatment 1: IBAs as features) noting that there can be many IBAs identified for the same species, and an individual IBA can be designated because of more than one species. Second, we assumed that IBAs associated with individual species represent the most important parts of their distribution throughout the EEZ, and treated the species as the conservation feature for which we set a target (Treatment 2; IBAs as core habitats). Third, we created an abundance map for every species using the maximum population size recorded for each IBA location and assuming this population is evenly distributed in the planning units found within each IBA area (Treatment 3: IBAs as abundances). In this case the conservation feature is also the species, but its value is weighted by its local abundance in each planning unit.

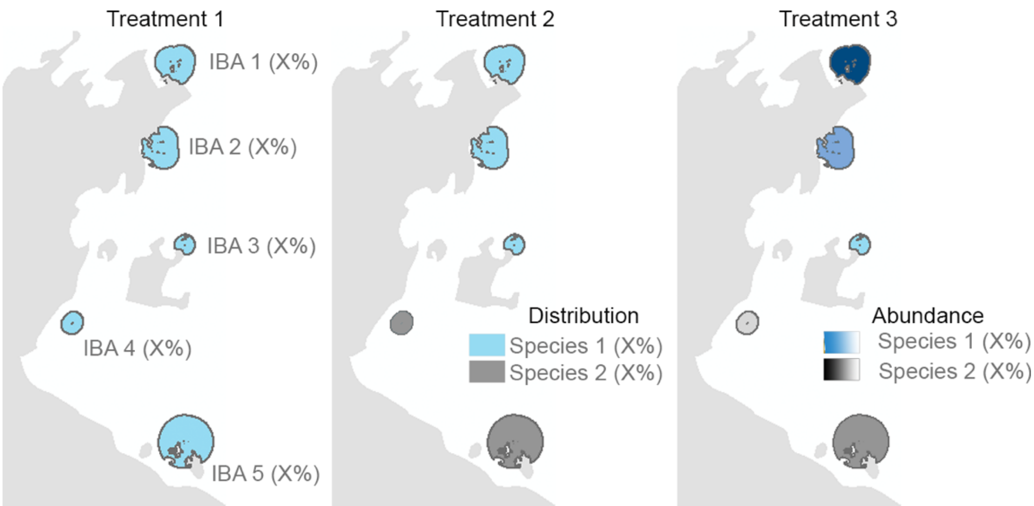


Figure 3.2 A visualization of how IBAs and targets are considered under our three treatments: 1) as individual features; 2) as the core habitats for the associated trigger species; and 3) as the abundances of associated trigger species where the distribution is weighted by the local population estimates. Note: for clarity in this example each IBA is only triggered by one species but many IBAs are triggered by more than one species.

Conservation targets

Targets for conservation features are often set based on characteristics such as endemism, rarity, or risk of extinction or threat. However, targets can also be assigned based on policy commitments (e.g. Convention on Biological Diversity Aichi Target 11) and planning precedent (Carwardine *et al.* 2007). We constructed planning scenarios by varying the targets for habitats and marine IBAs (Table 3.2). For the habitat and bioregional features, we used a constant 20% target as was used in the zoning plan for Australia's Great Barrier Reef (Fernandes *et al.* 2005). We varied the targets for marine IBAs across treatments in three ways. First, we considered them as irreplaceable sites and set a target of 100% for each IBA (Scenario 0 and 1b). Second, we used a 'sliding scale' approach (Figure 3.2) based on the IUCN Red List status of the species used to identify the IBAs (the 'trigger' species). Following Australian terrestrial reserve policies (Commonwealth of Australia 1997), we set targets as: Least Concern= 10%; Near Threatened = 20%, Vulnerable=60%, Endangered= 90%. When more than one trigger species was used to identify an IBA, we set the target based on the species with the highest threat category. Third, we set targets for IBAs equal to the other conservation features at 20%. We note that while setting different targets based on the criteria for which an IBA is established may seem logical, a defensible, systematic and appropriate way to justify this approach does not exist yet.

Evaluation metrics

We defined surrogacy as the incidental coverage of features captured in a spatial plan when no specific targets were set for those features (Grantham *et al.* 2010). Sutcliffe *et al.* (2015) introduced the percentage gap metric as an evaluation of surrogate effectiveness in spatial prioritization. This metric measures the average target shortfall for a given scenario when surrogates drive the prioritization. For example, if all features miss their target by 20%, or if only a third miss their target by 60%, then the percentage gap is 20%. If all targets are met the percentage gap = 0, and the surrogates are considered effective (Jantke *et al.* 2018). We used this metric to evaluate the surrogate effectiveness of plans based on marine IBAs (Scenario 0) and habitat features (Scenario 1a) to meet 20% targets for each other, as well as the geographic ranges of 58 seabird species found within Australia's EEZ (BirdLife International and the Handbook of the Birds of the World 2016).

We also used the proportional protection equality metric (*PE*) described in detail by Chauvenet *et al.* (2017) for scenario comparisons. *PE* is based on a modified version of the Gini coefficient which ranges between 0 and 1 (Barr *et al.* 2011; Chauvenet *et al.* 2017). In conservation planning, *PE* evaluates how equal a network of protected areas is for feature representation. For example, if a network protects the same proportion of every feature's distribution, *PE* would be equal to 1. The more disparity there is in protection across features, the more unequal the network is and the lower the PE value. We evaluated scenarios in terms of the trade-off between representation (measured in *PE*) and cost-effectiveness with respect to two kinds of conservation features – habitats (measured in proportions of bioregions protected) and species (measured in proportions of total abundances protected derived from IBA Treatment 3) and the average of these two values. We defined the upper and lower bounds of this trade-off as efficiency and inefficiency frontiers, respectively.

3.4 RESULTS

The spatial results for all scenarios can be viewed in Figure 3.3.

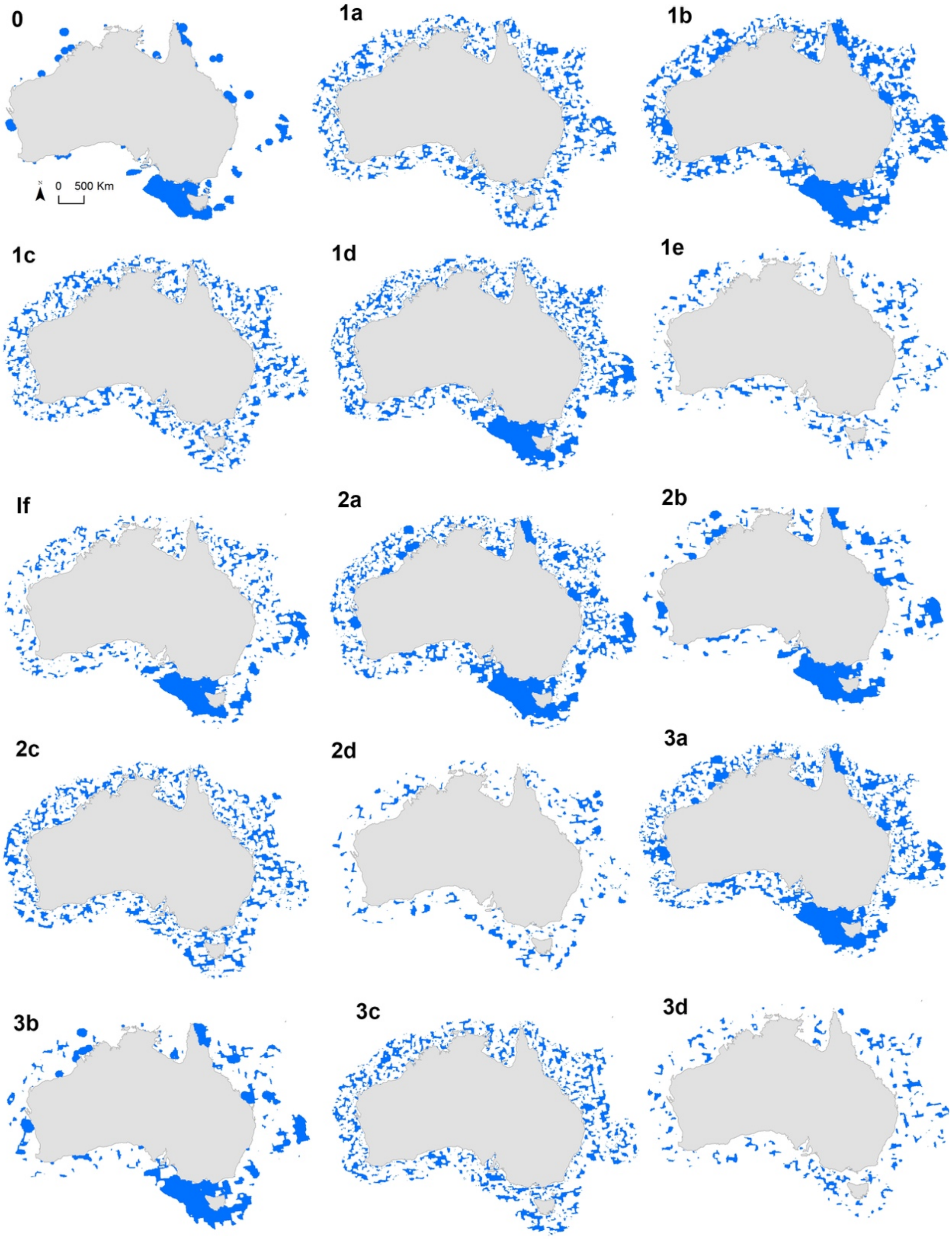


Figure 3.3 Best solutions for all 15 planning scenarios (Refer to Table 3.2 for feature targets and IBA treatments).

Surrogate performance to achieve representation targets

The effectiveness of individual marine IBAs and habitats (including bioregions) to act as surrogates for each other was poor. A conservation prioritization based solely on IBAs (Scenario 0) adequately met targets for only 519 of the 1,659 habitat features, and offered no protection to 828 of the smaller range features with a percentage gap of 62%. Similarly, a conservation prioritization based solely on habitats (Scenario 1a) met targets for only 55 of 136 marine IBAs, with a percentage gap of 49% (Figure B1).

Interestingly, spatial prioritizations driven solely by IBAs (Scenario 0) performed poorly as surrogates for trigger species' ranges when we set 20% targets (Figure 3.4A). IBAs biased the protection of seabird ranges away from non-trigger species (seven of which are listed as Vulnerable or higher on the IUCN Red-List), with large discrepancies in the amount of species-level protection provided (percentage gap = 19%; Figure 3.4B). In contrast habitats were excellent surrogates for representing seabird ranges at the 20% level, meeting targets for all IBA trigger species (Figure 3.4C) and with only a negligible percentage gap (0.02%) for non-trigger species (Figure 3.4D).

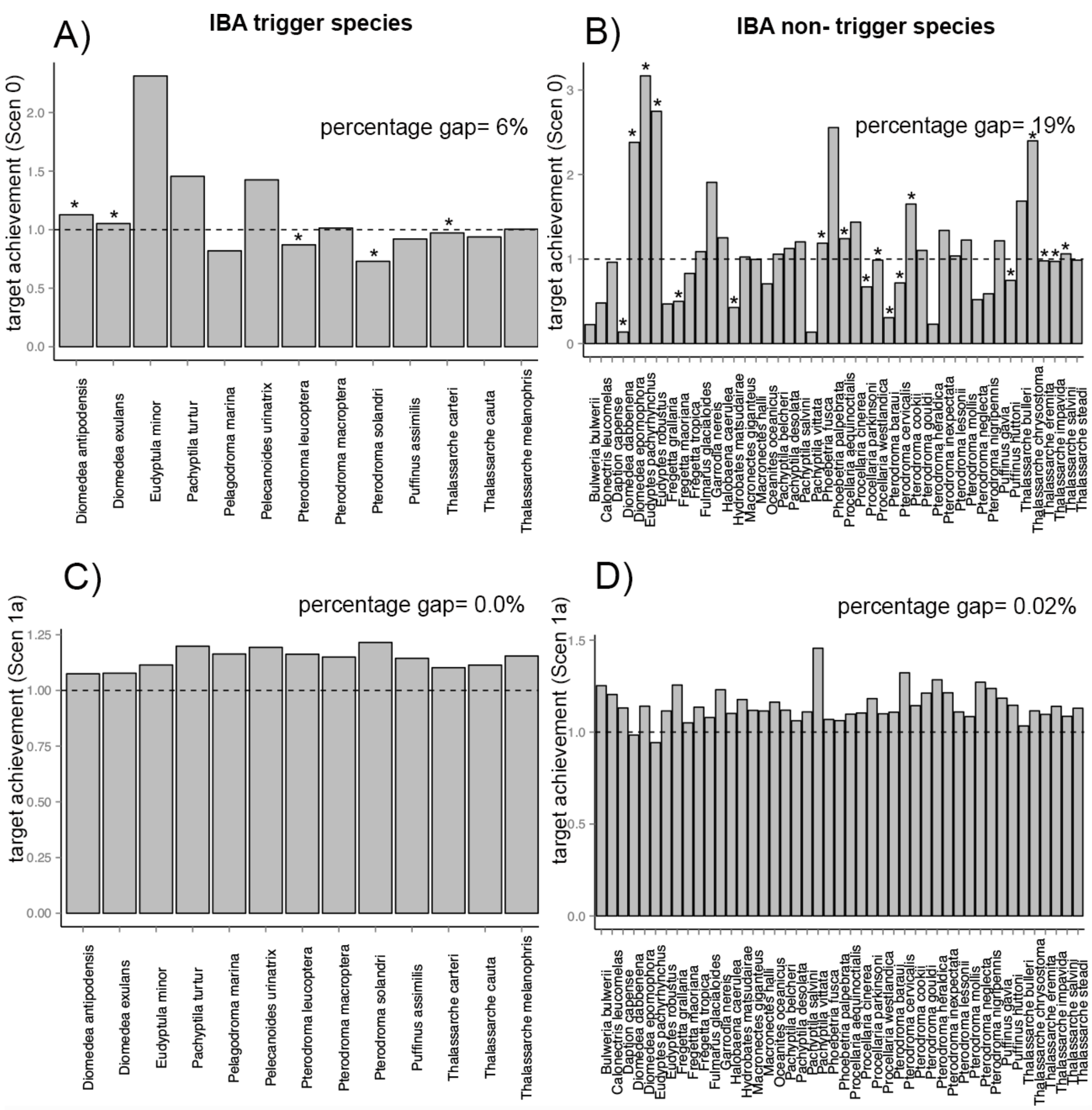


Figure 3.4 The surrogate effectiveness of Australia's marine IBAs (A-B), and habitats/bioregions (C-D) to achieve 20% conservation targets for the pelagic distributions of Australian seabird species. Results are reported as the percentage gap metric of each scenario's performance for species that do and not trigger an IBA. Asterisks note species listed as Vulnerable or higher according to the IUCN Red-List.

Protection Equality, cost-efficiency and IBA treatments

Given IBAs and habitats were poor surrogates for one another, we focus the following results only on those scenarios where all features were included in the analysis (See Figure B2 for Lorenz curves for each scenario). Setting targets for individual IBAs (Treatment 1) at 100% (e.g. treating them as irreplaceable sites) and habitats targets of 20% (Scenario 1b) was the worst scenario in terms of protection equality, *PE* (Figure 3.5a-c). When we instead assigned a 20% target to both IBAs and habitats (Scenario 1c), the bioregional *PE* improved by 28% and the cost-efficiency of the prioritization improved by more than 50% (Figure 3.5a; Table 3.3). While this treatment did not perform as well as other treatments for *PE* across species (Figure 3.5b), it formed the upper bound of the efficiency frontier for the average *PE* values (Figure 3.5c).

Table 3.3 Results of the best solutions for each scenario and associated *PE*. The best performing scenarios are based on the trade-off between PE and cost-effectiveness (in bold) derived from Figure 3.4. Scenario 0 has NAs because *PE* = 1 by default due to the inclusion of 100% of each IBA.

Scenario	Cost of Network	No. Planning Units	PE of best solution (bioregions)	PE of best solution (species abundance)	Average PE
Surrogacy scenarios					
0	89043	10095	0.36	NA+	NA
1a	**81510**	**16189**	**0.94**	**0.68**	**0.81**
Treatment 1: IBAs as features					
1b	180087	27439	0.71	0.98	0.85
1c	**80549**	**15864**	**0.90**	**0.74**	**0.82**
1d	125904	21553	0.73	0.77	0.75
1e	37605	8017	0.56	0.72	0.64
1f	84047	15178	0.43	0.73	0.58
Treatment 2: IBAs as core habitats					
2a	152920	23346	0.69	0.93	0.81
2b	116572	16266	0.46	0.97	0.71
2c	**84951**	**16416**	**0.94**	**0.62**	**0.78**
2d	27235	6713	0.46	0.52	0.49
Treatment 3: IBAs as abundances					
3a	152008	23667	0.70	0.97	0.84
3b	116415	16601	0.49	0.97	0.73
3c	**84988**	**16326**	**0.93**	**0.73**	**0.83**
3d	27182	6977	0.40	0.75	0.58

Treating IBAs as species distributions (Treatment 2) produced mixed results. Setting sliding scale targets for the species and 20% targets for habitats (Scenario 2a) returned relatively inefficient results compared to setting a flat 20% target for each species distribution (Scenario 2c). While Scenario 2c provided the most equal protection for bioregions of all scenarios (*PE* = 0.94, Table 3.3), the network performed worse than average for species protection equality ($PE_{,}$ = 0.62) (Fig 3.5b).

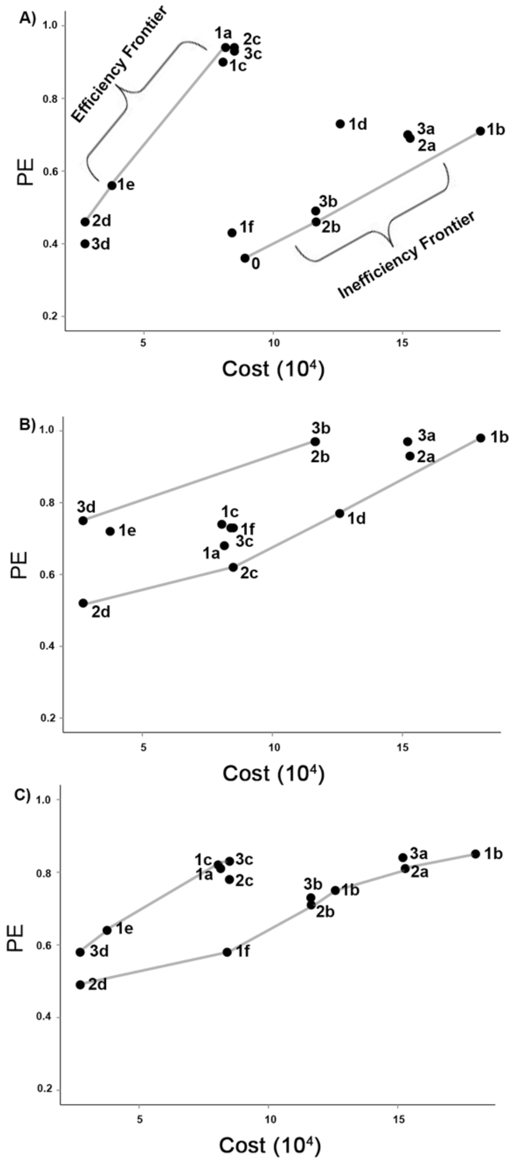


Figure 3.5 The network costs and protection equality (*PE*) for each scenario's best solution (see Table 3.3). *PE* is calculated for a) Australia's bioregions; b) IBA trigger species abundances (derived from Treatment 3); and c) the average of the two. To assist with evaluating the trade-off between costs and *PE*, we consider those scenarios on the upper and lower bounds of the graphs as efficiency and inefficiency frontiers. Point labels correspond to the scenarios described in Table 3.1.

3.5 DISCUSSION

Marine IBAs provide species-level data to inform spatial planning for expanding the global MPA estate. However, in the absence of best practice guidelines on how to use the marine IBA dataset, spatial planners are left with several options: not use the IBAs; treat them as irreplaceable sites (i.e. protect 100% of every IBA); or treat them like any other conservation feature and assign them a protection target. Our analysis explores these options by first assessing IBAs as surrogates for benthic and pelagic biodiversity, followed by an evaluation of different IBA target-setting strategies and the influence they hold in providing cost-effective and equitable MPA networks.

Ignoring IBAs overlooks unique sites for seabirds

Neither marine IBAs nor habitat features were effective surrogates for each other. IBAs were also poor surrogates for the pelagic ranges of Australian seabirds, but much of the range of these species consist of low-quality habitat that is not likely essential for species persistence. Thus, there is evidence that IBAs help address the principle of adequacy by identifying areas that support these critical processes rather than representing patterns of biodiversity. However, we cannot attest to the merits of marine IBAs to capture biodiversity in other regions of the world. Having an IBA dataset based on a comprehensive evaluation of Australia's entire EEZ, would likely increase the surrogate performance of IBAs for other biodiversity. Similarly, targeting a higher percentage of habitats would likely improve their performance as surrogates for IBAs. Importantly, our analysis suggests that it is prudent to include IBAs as an additional data layer despite them not representing broader marine biodiversity because IBAs contain unique spatial information on the processes that drive seabird abundances and distributions.

Treating marine IBAs as irreplaceable is very inefficient

BirdLife International's IBA program considers all IBAs to be equally important for marine biodiversity (BirdLife International 2010b; Butchart *et al.* 2012). The strictest translation of these guidelines suggests that every IBA should therefore be conserved in its entirety and treated as irreplaceable. When we treated marine IBAs as irreplaceable sites (e.g. 100% of every IBA is conserved), we arrived at the worst performing scenario for habitat-based protection equality (Fig. 3.5a; Scenario 0). Treating IBAs as irreplaceable sites and setting additional targets for biodiversity features resulted in the most inefficient network across all 15 scenarios (Fig. 3.5a-c; Scenario 1b). When only considering habitat features, we observed far better performance for cost-efficiency (~ 50% cost reduction) for a small reduction in the averaged protection equality of species abundances and habitats (Fig. 3.5c-Scenario 1a). These results demonstrate that treating marine IBAs as irreplaceable by conserving the full extent of every marine IBA, particularly large IBAs, is not a practical way to build MPA networks that are ecologically representative and cost-efficient.

Recommendations for integrating IBAs into spatial prioritization as conservation features

Spatial conservation prioritization delivers spatial plans that cost-effectively represent biodiversity within a complementary network of sites. This is best accomplished using information on well-stratified habitats and ecoregions, coupled with data on individual species or the distributions of other important biodiversity elements. IBAs can be an important source of such biodiversity data and our results suggest marine IBAs should be treated as conservation features for which a target is set. When we set 20% targets for both IBAs and habitats, we produced plans that are reasonably efficient and representative - defining the upper bounds of the efficiency frontier for average *PE* and cost (Fig. 3.5c: Scenarios 1c, 2c, 3c). However, these targets were somewhat arbitrary and proportional representation does not guarantee the long-term viability of species.

Setting targets for IBAs based on species threat status is a rational approach, as sites triggered by highly threatened species should be more influential in establishing a protected area than sites triggered by a Least Concern species. However, we found large inefficiencies with this approach principally due to the size of the candidate IBAs driven by Endangered and highly mobile species. For example, the Endangered Black-browed albatross defines six candidate IBAs covering approximately 248,000 km^2 of Australia's Southern Ocean and for which we set a 90% target according to Australian policy (see Fig 3.1). This exposes the challenge of defining pelagic IBAs where no clear biogeographic or habitat boundary exists and which are identified using seabird tracking datasets (Lascelles *et al.* 2016). While we encourage improving the conservation return on investment from seabird tracking (McGowan *et al.* 2017a, Mason *et al.* 2018), protecting large pelagic IBAs for highly mobile threatened species such as albatross and petrels is probably not politically or socially feasible. Such IBAs may be best used to inform broad policy not MPAs.

While planners may consider setting differential targets based on species attributes, such as range size (Rodrigues *et al.* 2004), the characteristics of individual IBAs could also influence target setting. For example, planners could set higher conservation targets for IBAs capturing seaward extensions of nesting colonies or migration bottlenecks for declining populations, than for large pelagic IBAs. Appropriate targets will depend on the objectives of the spatial plan, the size of the planning region, the number of marine IBAs identified for the region and what other species-level data are available.

The future of IBAs in marine spatial planning

Using criteria to identify sites that are globally significant for biodiversity is not equivalent to identifying priority conservation areas for action (Knight *et al.* 2007; Di Marco *et al.* 2015; IUCN 2016). Criteria-based delineations often do not explicitly state what actions should be taken to ensure species persist within those sites, nor do they routinely account for the financial, social and political constraints associated with implementing conservation actions. Further, these sites do not account for complementarity in site identification (Brown *et al.* 2015b). Di Marco et al. (2016)

stressed the importance of complementing the threshold-based identification of terrestrial IBAs with the systematic identification of irreplaceable sites. Our results support the same claim for the marine realm. While we found little support for treating marine IBAs as universally irreplaceable (100% targets for all IBAs), our recommendations do not preclude setting 100% targets for particular IBAs when appropriate. Following the terrestrial analysis of (Di Marco *et al.* 2015), evaluating how different IBA criteria reflect the irreplaceability values of marine IBAs would be a valuable next step towards establishing a systematic method for setting targets based on underpinning criteria.

The integration of marine IBAs into spatial conservation prioritization demands planners be equipped with more specialized knowledge of how and why individual IBAs exist. Attributes of the IBA trigger species and criteria, as well as the method of establishment (e.g. whether through telemetry tracking (Lascelles *et al.* 2016), at-sea surveys (Smith *et al.* 2014), expert opinion, or identifying foraging hotspots (Arcos *et al.* 2012) should be provided. While some IBAs may be too large for strict protection they could inform specific spatial policies such as modified fishing activities that reduce bycatch, defining places where new marine activities must go through more rigorous impact assessment, or national and regional fisheries policy on gear and catch. Releasing prescriptive actions associated with these spatial data will further advance the utility of IBAs in spatial conservation prioritization. We believe these findings are relevant to other threshold-based approaches, such as Key Biodiversity Areas (IUCN 2016) and Ecologically or Biologically Significant Areas (Bax *et al.* 2016), where sites are delineated with the intent of influencing global conservation priority-setting.

4 Optimal ocean zoning within a sparing versus sharing framework

This chapter is reproduced from the following publication with some alterations to format and structure:

McGowan J, Bode M, Davis K, Krueck N, Beger M, Yates K & HP Possingham. 2018. Ocean zoning within a sparing vs. sharing framework. *Theoretical Ecology.* doi.org/10.1007/s12080-017-0364-x

4.1 ABSTRACT

The land-sparing versus land-sharing debate centres around how different intensities of habitat use can be coordinated to satisfy competing demands for biodiversity persistence and food production in agricultural landscapes. We apply the broad concepts from this debate to the sea, and propose it as a framework to inform marine zoning based on three possible management strategies, establishing: no-take marine reserves, regulated fishing zones, and unregulated open access areas. We develop a general model that maximizes standing fish biomass, given a fixed management budget while maintaining a minimum level of catch. We find that when management budgets are small, sea-sparing is the optimal management strategy. As the management budget increases, the optimal strategy switches to sea-sharing. Our intention is to illustrate how general rules of thumb derived from plausible, single-purpose models can help guide marine protected area policy under our novel sparing and sharing framework. This work is the beginning of a basic theory for optimal zoning allocations and should be considered complementary to the more specific spatial planning literature for marine reserve as nations expand their marine protected area estates.

4.2 INTRODUCTION

The land-sparing versus land-sharing (sparing vs sharing) debate emerged from contrasting views about how to balance the competing demands for biodiversity persistence and food production in agricultural landscapes (Green *et al.* 2005; Fischer *et al.* 2014). Land sparing involves spatial consolidation and intensification of agricultural activities. This approach is based on the idea that concentrated agricultural activity can achieve equal or higher yields in a smaller land area than low intensity usage. More land is available for biodiversity protection thereby providing a net conservation benefit. The counter-argument in support of sharing argues that wildlife-friendly farming produces lower yields per unit area, but supports biodiversity conservation by using less intensive production techniques across larger portions of the landscape (Fischer *et al.* 2008). Studies typically investigate the sparing vs. sharing dichotomy to identify the most appropriate strategy for a given context, because how well species or populations fare alongside increasing agricultural yields depends upon local production methods (Balmford *et al.* 2005; Green *et al.* 2005; Phalan *et al.* 2011), species traits and agricultural yields (Grau *et al.* 2013). Although much of the debate centers around semantic issues (Tscharntke *et al.* 2012; Fischer *et al.* 2014), more recent empirical research supports the discussion with quantitative data (Lee *et al.* 2014; Butsic & Kuemmerle 2015; Kremen 2015b; Law & Wilson 2015) particularly in plantation and livestock production (Grau *et al.* 2013).

While not framed as sparing vs. sharing per se, equivalent discussions in ocean management debate the benefits of either prohibiting fishing in some parts of the seascape or constraining fishing through management (Hilborn 2016). Marine reserves that exclude all extractive activities are a popular tool for conserving marine biodiversity. Efforts are underway to increase the number of reserves globally, particularly in developing countries where inshore fisheries experience heavy exploitation (White *et al.* 2014). In contrast, it is argued that traditional fisheries management, such as catch and size regulations, are more effective mechanisms to maintain healthy fish stocks and productive fisheries (Hilborn *et al.* 2004). In this context, quantitative investigations about sparing vs sharing in the sea traditionally argue whether or not marine reserves will provide greater fish biomass and environmental benefits than fishery regulations (Hastings & Botsford 1999; Hilborn *et al.* 2006; White & Kendall 2007) – a typically either/or argument. These studies identify whether a fraction of the system in marine reserves – sparing - or regulation across the entire area – sharing - maximizes fishery yields or profits (Sanchirico & Wilen 2001; Gerber *et al.* 2003; Hastings & Botsford 2003; Sanchirico *et al.* 2006; White *et al.* 2008). We note, however, there is a body of literature that considers and tests the utility of marine reserves as part of a mixed management strategy to achieve fisheries objectives, rather than an either/or argument (Holland & Brazee 1996; Mangel 2000; White *et al.* 2010).

Valid concerns remain regarding the socioeconomic impacts of marine reserves on communities and countries. Indeed, most studies modeling the use of reserves for fisheries management have found that the addition of reserves will reduce yields whenever fisheries are already well managed (Tuck & Possingham 2000; Hilborn *et al.* 2006), or suggest reserves are an effective secondary management option in cases where fisheries are heavily-exploited and policy or effort reductions are unlikely to succeed (Holland & Brazee 1996). The establishment of marine reserves can lead to a redistribution of fishing effort within a region, potentially negating any net benefit of the reserve through increased fishing pressure elsewhere (Agardy *et al.* 2011). Other studies have identified scenarios in which reserves could be essential for maintaining high yields in spite of otherwise effective management regulations. These include, for example, the potentially critical function of reserves as a buffer against environmental stochasticity (Mangel 2000; West *et al.* 2009), and the positive impact of reserves on the density-dependent survival of young fish (White 2009) which could increase the net productivity of fished populations adjacent to reserves (but see White *et al.* 2008; Hart & Sissenwine 2009; Russ & Alcala 2011).

Similar to the terrestrial debate, there is no standard solution to protecting biodiversity and meeting human needs from the sea. Equipping decision makers with a variety of tools to inform policy will enable better and more flexible management strategies as to which zoning allocation should be pursued in a given context. Australia's Great Barrier Reef Marine Park, for example, represents one of the first systematically designed networks of marine protected areas in the world whose shared seascape consists of roughly equal proportions of marine reserves, managed fisheries and general use areas (Fernandes *et al.* 2005). While successful in Australia (McCook *et al.* 2010), encouraging other countries to adopt the exact same allocation would be unfounded given the diverse ecological, socio-economic and governance structures across marine jurisdictions globally. Yet general ecological and socio-economic principles apply everywhere, and rules of thumb based on plausible, single-purpose models can help guide policy (Starfield 1997; Gerber *et al.* 2003) in a time of rapid marine protected area expansion (Klein *et al.* 2015).

Here, we transfer the land sparing vs. sharing debate to the sea using three common zoning types: fully protected no-take marine reserves, managed fishing zones, and unregulated and/or unmanaged fishing zones, hereafter called "open-access". We choose to characterize an allocation with only marine reserves and open-access areas as a "pure" sea sparing strategy. In the sea, we translate sharing to be any strategy that incorporates managed fishing zones, which can manifest as regulations on spatial or temporal effort, or gear restrictions that minimize impact to the benthos or non-target species. We characterize sharing along a continuum where some proportion of the seascape is managed, but consider a "pure" sharing strategy when the entire seascape is managed and no reserves or open-access zones exist (Figure 4.1). When defined in this manner, we move beyond the sparing vs sharing dichotomy that prevails in the terrestrial debate (Kremen 2015), to develop a framework that includes seven potential spared and/or shared seascapes. We then illustrate how to operationalize the framework using a simple modeling approach whose optimally zoned seascapes secure a minimum biomass yield while maximizing standing stock biomass (the environmental benefit) for a given management budget. This approach considers a single habitat-dependent fished species whose harvest methods exert different levels of pressure on the benthos. We are interested in the circumstances in which the optimal seascape is either a sparing strategy, defined here when the case study area is allocated amongst no-take reserves and open access zones, and when that changes to a sharing strategy, defined when the case study includes a managed fishery zone, and potentially the addition of either or both no-take and/or open access zones (Figure 4.1).

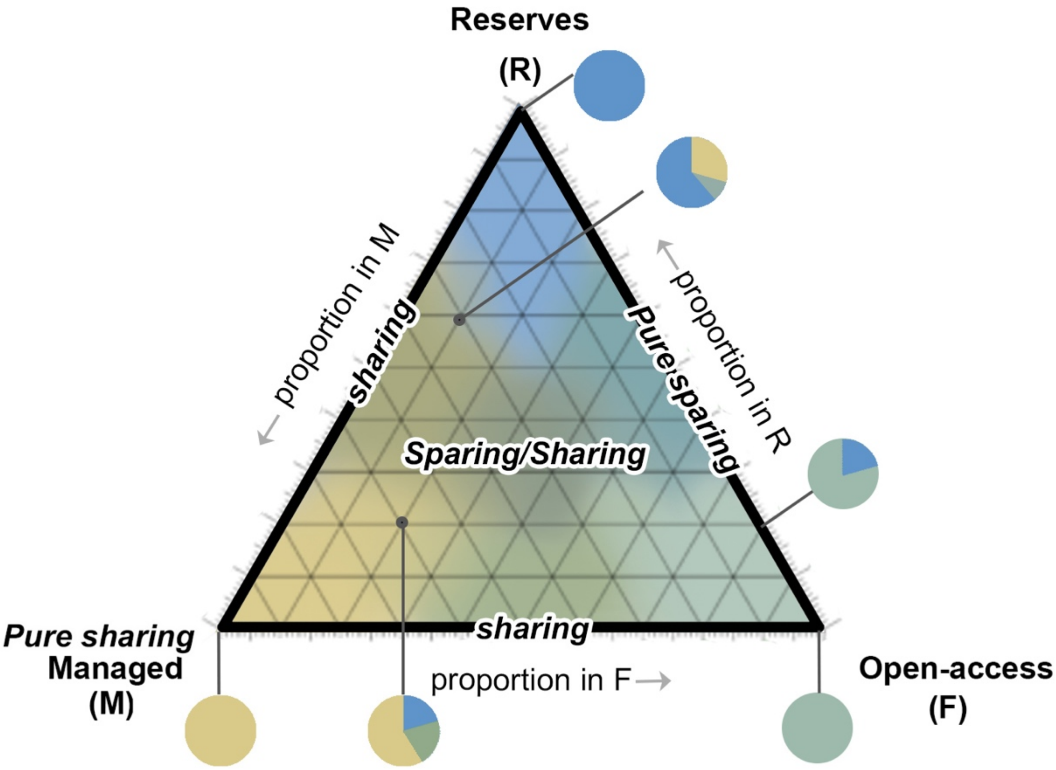


Figure 4.1 Classes of sparing and sharing seascapes derived from our three-zone framework. Pure spared seascapes are those defined by no-take reserves (*R*) and open-access areas (*F*) and defined in these plots as any point on the line between *F* and *R* (excluding apex points where the zoning allocation would be 100%). Shared seascapes are defined by any allocation with managed fishing zones (*M*), with a pure shared seascape defined by apex M (100% managed). Pie charts offer illustrative examples to help interpret the zone allocation at given points on the graph.

4.3 METHODS

Model Description

Our model assumes we are managing a single habitat-dependent fished species that reproduces with a pelagic larval phase leading to evenly distributed recruitment in all parts of the seascape. The seascape is divided into three management zones: protected marine reserves (fraction *R*), managed fishing zones (fraction *M*), and open-access fishing zones (fraction *F*; so every part of the system is in one of the zones, $R + M + F = 1$). There is a financial cost to reserving (C_R) and managing (C_M) habitat, the sum total of which must not exceed an allotted total management budget (*B*), $R * C_R +$

$M^*C_M \leq B$. We assume there is no management cost incurred in the open-access zone. Our objective is to maximise the total population of our fishery species subject to the budget constraint and a minimum biomass yield. Our model identifies the optimum proportional allocation of a seascape amongst the three zones.

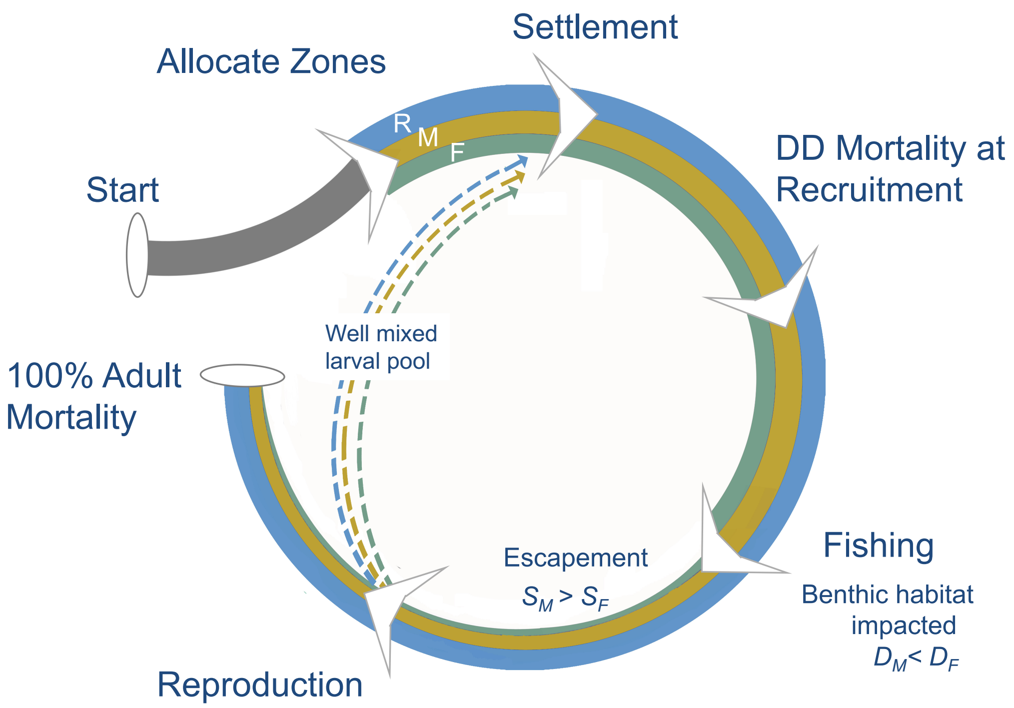


Figure 4.2 Schematic of the steps in the process model used in the simulated optimization and general rules that apply to the assumptions about fishing impacts and escapement across the two managed (M) and open-access fishery (F) zones.

To link the decisions about seascape zoning allocation to our objectives and constraints, we use a simple population model tracking adult post-harvest biomass, A_t, at time t. We start by noting that if l larva attempt to settle uniformly, at random, across n sites, with only 1- d proportion of sites viable, and with all viable sites only able to support one individual, recruitment is given by the Beverton-Holt function (Duncan *et al.* 2009):

$$R(l) = \frac{(1-d)l}{1+l/n} \tag{4.1}$$

Let L and K, be fecundity and the total number of potential sites available for larval settlement (i.e. larval carrying capacity), respectively. Fishing mortality in the managed and open access zones are (1-S_M) and (1-S_F), respectively. We assume habitat damage temporarily reduces the proportion of available sites for settlement in zone type i, by D_i, for i in $\{M, F, R\}$, at time t. We assume the damage is more severe in the open-access zone ($D_M < D_F$), and that no habitat damage occurs in the no-take reserves, ($D_R = 0$). Assuming fish reproduce post-harvest and contribute larva to a common pool, which are then allocated to the three zone types proportionally based on area, we obtain the following difference equation for total post-harvest population size,

$$A_{t+1} = \frac{[S_M(1-D_M)M+S_F(1-D_F)F+R]LA_t}{1+LA_t/K}, \tag{4.2}$$

which has a stable equilibrium at

$$A^* = \left[S_M(1-D_M)M + S_F(1-D_F)F + R - \frac{1}{L}\right]K, \tag{4.3}$$

with equilibrium harvest

$$H^* = \frac{[(1-S_M)(1-D_M)M+(1-S_F)(1-D_F)F]LA^*}{1+LA^*/K}. \tag{4.4}$$

For simplicity, we assume 100% adult mortality after harvest and reproduction, but acknowledge the lifecycle for many short-lived species may not be annual. We then search through all financially possible zoning configurations to find the optimal seascape at equilibrium. The optimal solution is the seascape allocation that delivers the largest environmental benefit (total equilibrium post-harvest adult population size), while meeting the minimum food production and budget constraints. Ignoring the catch constraint, we obtain an analytic solution for this optimal zoning allocation, which produces a general rule of thumb which holds true for small budgets (see Results). However, to account for the nonlinear catch constraint, we solved for the optimal allocation using simulations conducted in Matlab (MathWorks, Natick Massachusetts, USA; Appendix C).

Case study parameterization

For our case study, we apply our model to derive an optimum zone allocation based on the conditions of tiger prawn fisheries (O'Neill & Turnbull 2006) using the parameters outlined in Table 4.1. Damage caused by benthic fishing is difficult to quantify and depends on the type of gear, and the frequency and distribution of effort (Thrush *et al.* 1998; Collie *et al.* 2000). Impacts to coastal habitats range from diminished structural complexity (Auster 1998), changes to community composition (Thrush *et al.* 1998) and altered ecological processes (e.g. reduced primary production from macrofauna depletion; enhanced nutrient cycling via suspended sediment loads (Auster & Langton 1999).

For the purpose of this exercise, we make several necessary simplifying assumptions about benthic impacts from fishing activities. We recognize benthic habitat condition is case-specific. In cases where more detailed data exist, this information can easily be incorporated into our modeling framework. We assume that previously unregulated trawling has impacted the benthic community in the open-access zone. We define "significant" impact as the mean mortality (20-50%) of benthic invertebrates reported in Collie *et al.* (2017) for towed benthic fishing gears. We assume perfectly enforced restrictions in the managed zone reduce the fishing impacts on the benthos by half so that $D_M = 0.5*D_{F.}$ (Chuenpagdee *et al.* (2003). We set S_M to be the survival proportion that will yield MSY in a fully managed seascape and S_F to be the survival that leads to an equilibrium of 10% of virgin biomass when the fishery is completely unregulated, open access. We assume that fishers will not tolerate a level of catch lower than the pre-managed open access yield therefore the catch threshold (*CT*) is set to the open access harvest.

Costs

Despite being critical to decision-making about natural resource management (Naidoo *et al.* 2006, Iacona *et al.* 2018), costs associated with establishing and managing protected areas are often poorly reported, difficult to quantify (Balmford *et al.* 2004; Ban *et al.* 2011), and highly contextual (Rojas-Nazar *et al.* 2015). As a flexible way to integrate the amalgam of costs (e.g. stock assessments, ecological monitoring, staffing, enforcement etc.) associated with the different zones (Ban *et al.* 2011) and across regions, we parameterize the relative costs between protected and managed areas. One key factor driving the cost of management interventions, be they marine reserves, gear restrictions, effort reduction, or sustainable harvesting, is the cost of enforcing compliance. The costs associated with surveillance and enforcement depend on both the size of the zones and the social and economic characteristics of the resource users. Only a few studies have explicitly quantified these costs (Ban & Klein 2009; Davis *et al.* 2015). Ban et al. (2011) compared the enforcement costs for staffing an entirely no-take protected area versus a mixed zone seascape (protected and fished) and found that compliance staffing was doubled when mixed zoning occurred.

As a starting point for our case study, we assume the cost of enforcing fisheries management is twice that of protecting area, $C_M = 2C_R$ but we test the sensitivity of the outcome to variations in the relative costs to protect and manage when $C_R = C_{M\,\text{and}}$ when the cost of enforcing reserves is double the cost of enforcing managed fishing areas $C_R = 2C_{M.}$ We also examine the case of additional fixed costs of reserves and managed areas, costs that do not scale with area, in the appendix. Management budgets can vary enormously between regions and in time, therefore, we are most interested in identifying the circumstances under which optimal management strategy shifts between sparing and sharing as the management budget changes. We investigate the optimal strategy under different budgets to variations in several parameters of interest: habitat damage in the open-access fishing zone (D_F), escapement in the open-access fishing zone (S_F), fecundity (L) and the catch threshold (CT).

Table 4.1 Case study parameters based on population conditions for *Penaeus esculentus* (tiger prawn).

Parameter	Description	Value	Source
s	Intrinsic survival	1	O'Neill and Turnbull 2006
K	Carrying capacity of whole environment	30	O'Neill and Turnbull 2006
L*	Fecundity of adults	5	O'Neill and Turnbull 2006
D_F *	Habitat damage in the open-access fishing zone	0.35	Collie et al. 2017
D_M	Habitat damage in the managed fishing zone	0.175	(derived as $0.5*D_F$) Chuenpagdee et al. 2003
S_F *	Survivorship in Fished Zones	0.48	To achieve 10% virgin biomass at equilibrium. See formula in code, Appendix A
S_M	Survivorship in Managed Zones	0.65	To achieve MSY at equilibrium. See formula in code, Appendix A
CT*	Catch threshold	1.85	Open Access Equilibrium
C_M to C_P *	Cost ratio between managing and protection	2:1	Ban et al. 2011
* sensitivity tested (see Figure 4.4 and Appendix C)			

4.4 RESULTS

Case study

In our case study, we find that when management budgets are low (Figure 4.3 If there is no management budget, then fishing must occur under open-access conditions throughout the seascape, regardless of the fishery being considered because managed areas and reserves require investment. In our case study, we find that when management budgets are low (Figure 4.3 where $B \leq 0.61$), the optimal choice is to allocate the entire budget to establishing no-take zones and have no managed areas. With the budget exhausted the rest of the seascape remains in open-access fishing – considered here as a sea sparing strategy where the portions of the seascape not under protection are intensively harvested. As the budget increases, so does the fraction of the protected seascape. During this stage, initially, the catch increases because additional reserves increase larvae production, which is then mostly distributed to unregulated zones for fishing. However, after a critical reserve threshold, catch declines because additional reserves do not provide sufficient larval export to the open-access zones to compensate the fishery for the population now excluded from harvesting. Eventually the optimal seascape switches from sparing (reserves and open access) to include all three zones – a version of sea sharing (Figure 4.1). This occurs when there is already so many reserves that additional reserves prevent the fishery from satisfying the catch constraint. In this case biomass can be increased further with the addition of managed zones while still satisfying the catch constraint.

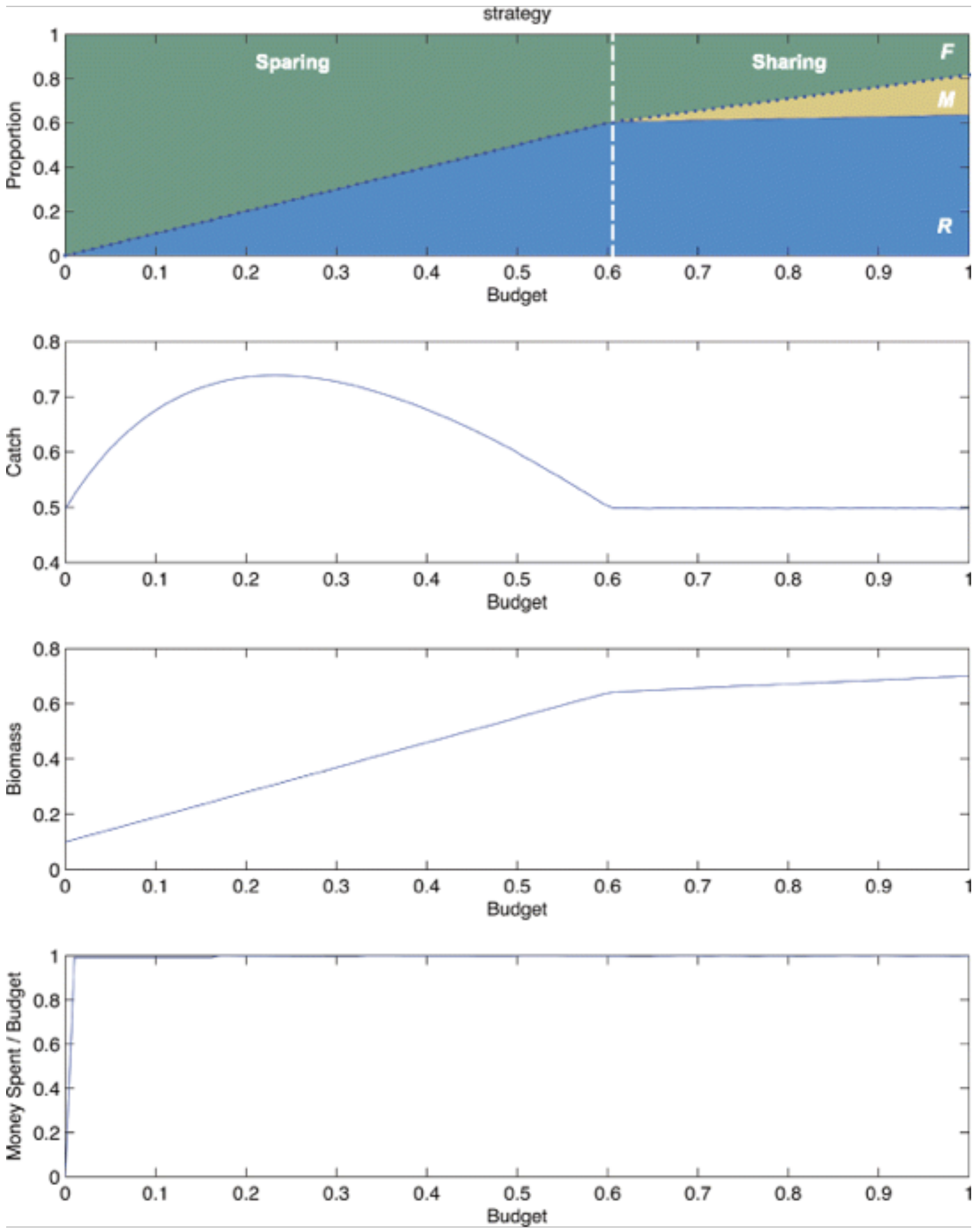


Figure 4.3 The optimal sparing versus sharing strategy (top) showing the fraction of the seascape allocated to each of the three zones with an increasing budget for our case study. No-take marine reserves in blue (R); open-access fishing in green (F); and managed zones in yellow (M). The white dashed line is the departure point between sparing and sharing. When there is no budget we can neither reserve nor manage. As the budget increases, first marine reserves and then managed fisheries, enter the optimal zoning allocation. Also shown are catch, biomass, and the spending regime.

Figure 4.4 shows how the optimal zoning allocation changes as a function of the budget for our parameters of interest: D_F, S_F, L and CT. Beginning with no budget, the seascape is completely open-access fishing (apex F). As the budget grows, the allocation moves along the "sparing" boundary, where the seascape consists of open-access and increasing proportions of no-take reserves. A point of departure, or switching point, finally moves the allocation away from sea sparing and into a configuration consisting of all three zones. We find this departure is most sensitive to changes in fecundity (L) and occurs when the reserve coverage is between 45 -70% of the seascape. When fecundity is greater, we switch to investing in management zones at lower proportions of reserves in the seascape.

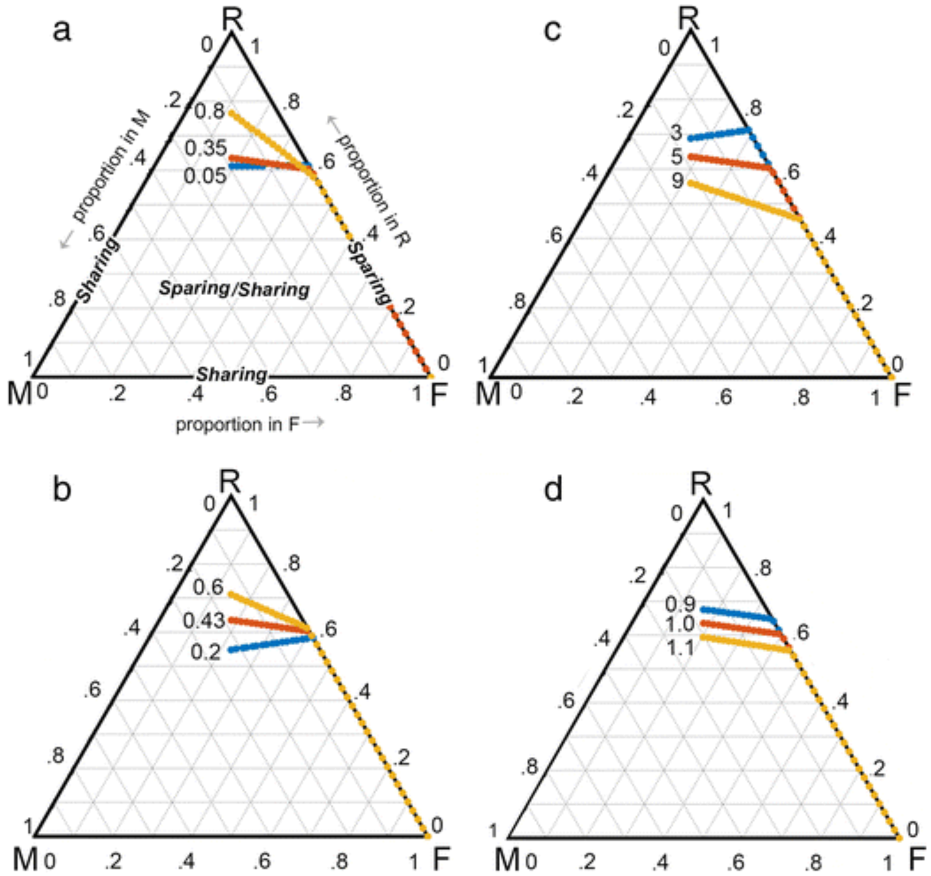


Figure 4.4 Ternary plots showing the fraction of the seascape in each of the three zones (R = no take reserves, M = managed fishing zones, F = open access) for a given budget, where $R + M + F = 1$. When no budget exists, $B = 0$, the entire seascape is open access, 100% F in the bottom right corner. Coloured lines show the sensitivity of the seascape allocation under several values for each parameter of interest: **a** habitat damage caused by fishing in the open-access fishing zone (D_F), **b** escapement in the open-access fishing zone (S_F), **c** fecundity of adults (L), and **d** the catch threshold (CT). The departure from the sparing strategy (line F–R) indicates the transition point from sparing to sharing as the budget increases

Regardless of the parameter tested, we consistently observe the phenomenon of sea sparing when budgets are small, as well as the switch to the three-zone version of sharing as budgets increase. This trend is robust to changes in the cost ratio as well as when we eliminate the influence of habitat damage caused by fishing in each zone ($D_M = 0$ and $D_F = 0$) (Figure C1) for further sensitivity analyses). Sensitivity manifests in two possible ways that affect the optimal seascape as the budget grows: 1) the point of departure from sparing to sharing, and 2) the proportion allocated to each zone (Figure 4.4). Interestingly, the proportion of area protected, R, at the point of departure from sparing to sharing remains fairly constant irrespective of the cost ratio for our case-study (Figures C2 - C4; about 60% of the seascape). When the cost of protection is double the cost of management, $C_R = 2C_M$ (Figure C3), the point of departure is substantially delayed as the budget grows large enough to share the seascape but ultimately follows the same investment strategy.

Optimal Rule of Thumb for Small Budgets

Our approach also allows us to derive an analytic rule of thumb to assist decision-makers about what the optimal investment strategy may be for their given context. With no catch constraint, the optimal zoning solution is to allocate the entire budget to marine reserves (sparing) if the benefit of adding a reserve (relative to open access fishing), per unit cost, is greater than the cost-benefit of adding a managed area. Otherwise the decision maker should spend their entire budget on managed areas. This rule can be simplified mathematically as: spend the entire budget on reserves if

$$1 - \frac{1 - S_M(1 - D_M)}{1 - S_F(1 - D_F)} > \frac{C_M}{C_R}. \qquad (4.5)$$

To derive this rule, let x be the amount of money allocated to reserves and, $B - x$, the amount of money allocated to managed areas. Then $R = x/C_R$ and $M = (B - x)/C_M$, and $F = 1 - R - M$. One can solve for the x that maximizes A^* by substituting these quantities into Equation 4.3 which produces condition 4.5.

Based on our numerical simulations, the rule of thumb held for all tested cases until so many reserves had been purchased that the catch constraint would no longer be satisfied if the decision maker continued adding reserves. For our baseline parameterization, we found that reserves were favoured over managed areas unless the cost of reserves was nearly 5 times that of managed areas. Even for the combination of parameters most favourable for managed areas in the sensitivity analysis, managed areas were not selected for low budgets unless the cost of reserves was over three times higher than the cost of managed areas.

4.5 DISCUSSION

A sea sparing and sharing framework

Seven seascape allocations emerge from our sea sparing and sharing framework (Figure 4.1). A seascape allocated entirely to one zone is highly unlikely as: (1) an entirely reserved no-take system (R=1) cannot meet the harvest constraint; (2) an unmanaged open-access system (F=1) likely results in over exploitation and potential fishery collapse (Hutchings 2000); finally, while (3) a purely shared system is possible (e.g. M=1 with no reserves or unmanaged fisheries), the reality of limited management budgets and global commitments to MPAs reduce the likelihood of this option persisting through time. Mixed zoning under our framework consists of (4) a pure spared seascape with both no-take reserves and open-access zones, (5) shared seascapes with managed and open-access zones, and two zoning configurations that allow "sparing and sharing." The first of these last two zoning configurations includes (6) no-take reserves and managed fisheries; and (7) no-take reserves, managed fisheries and open-access zones. With this conceptual starting point, a useful next step for the future would be to classify existing management plans within this framework to see what the most dominant strategies are in practice, and to create a typology of spared and shared seascapes that enable moving beyond the dichotomous view of the sparing vs sharing debate. Building on this idea, our framing also exposes the need for a more refined classification system, as "sparing", "sharing" and "sparing and sharing" are too vague to encompass the nuanced management practices governing marine systems (White *et al.* 2010; Kremen 2015b).

Only the rich can afford to share

When budgets are small, sea sparing is always the optimal allocation. As the budget grows, we arrive at a point where increasing the amount of the reserves any further will compromise our ability to achieve the minimum biomass yield. If budgets increase beyond this point, the optimal strategy is to start sharing. However, the optimal strategy under our framework will likely differ for a different set of objectives and constraints (White *et al.* 2017). For example, we approached this problem by identifying a single conservation objective (maximize standing biomass), while acknowledging a natural resource dependency (expressed by the minimum harvest constraint) and a fixed management budget (cost constraint). It is important to note there are many alternate ways to define a problem by treating each of these outcome variables either as objectives to be maximized or minimized, and/or constraints. Defining a different objective for ocean management (e.g.

maximising larval connectivity, protecting species climate refugia (Beger *et al.* 2015) or building near-pristine fish biomass (McClanahan *et al.* 2007), or evaluating trade-offs for multi-objective problems would also be valid approaches.

We strategically simplify many assumptions in order to develop a model that can begin to inform policy (Hastings & Botsford 1999). Opportunities to add complexity into our approach include incorporating a spatially-realistic modeling environment (Polasky *et al.* 2008; Metcalfe *et al.* 2015), alternative assumptions of density dependence before and after settlement (e.g. Ricker models), age structure, overcompensation (e.g. White and Kendall (2007), integrating more complex dispersal processes, accounting for variable distributions of fishing effort and displacement, socio economics (Sanchirico & Wilen 2002; Halpern *et al.* 2004; Armstrong & Skonhoft 2006; Costello & Polasky 2008) and developing multi-species models.

For some of these limitations, we can foresee how the model will respond. For example, adding age structure would allow biomass to accumulate in reserves, likely achieving our objectives with less reserved area. In instances where overcompensation is justified (Cury *et al.* 2014) we would expect to see higher reserve coverage (White & Kendall 2007). We acknowledge that our approach also depends on some degree of overfishing for this framework to apply. This assumption influences the point of departure, in that, the time at which managed areas are added will depend on the assumptions of overfishing. However, the general trend of sparing first and moving to the 3-zone version of sharing is robust and highlights that mixed management approaches have merit where substantial management capacity exists (Hilborn 2016).

The species and associated fishery we chose to represent in the model are intentionally responsive to reserves, because we believe that it is these types of species and fisheries that drive decisions on coastal management zone allocations. However, our findings may also apply to systems where common pool dispersal assumptions are not met. Recent studies on larval dispersal show that for several fishery species, dispersal distances are much more extensive than previously assumed (Green *et al.* 2015; Jones 2015; Williamson *et al.* 2016; Almany *et al.* 2017). In such cases reserve size and placement can be optimized with a high level of flexibility to provide for maximum fishery benefits (Krueck *et al.* 2017a; Krueck *et al.* 2017b).

Despite our stated limitations, our model goes beyond traditional management zone assessments by illustrating how fisheries management influences the optimal seascape allocation. Our approach is the first attempt to underpin the sharing and sparing debate with a process model. In doing so, we reveal a more nuanced and practical framework than the debate has produced to date (Kremen 2015b). Ocean management can benefit from applying this framework and devising simple rules of thumb to guide policy options, for example, investing in marine reserves when budgets are low with the addition of managed areas when budgets are high. Building additional complexity into this base exploration as well as developing the sea sparing vs sea sharing framework will help advance the debate and its relevance for marine policy. Additionally, as the costs associated with different aspects of fisheries management in shared seascapes become available (Iacona *et al.* 2018), estimates of how management budgets are allocated to different activities can become more realistic. This work is the beginning of a basic theory for optimal allocations within seascape zoning frameworks and should be considered complementary to the more specific spatial planning literature for marine reserve design and implementation, which addresses the size, shape and placement of individual MPAs within a seascape.

5 A flexible decision-support tool to initiate debt-for-nature swaps using conservation finance

5.1 ABSTRACT

Debt-for-nature swaps gained popularity in the 1980s and 1990s as a financial mechanism to address rapid declines in biodiversity within developing countries with high debt. Due to the high transaction cost of implementing debt-for-nature swaps and the large potential impact these arrangements hold for conservation, there is a need to prioritize future efforts. In this paper, we progress two broad goals. First, we develop a flexible new tool for answering the question of where an environmental NGO or philanthropic fund should invest in debt-for-nature swaps to generate the greatest expected return on investment. Our second goal is to understand how priorities for debt-for-nature swaps change according to: 1) the classification of different threats as abatable and unabatable; and 2) different benefits and ecological weightings. We use the case-study of Caribbean countries to provide a proof of concept for our approach and construct 12 planning scenarios to evaluate these differences. We found that the choice of benefit influenced the outcomes more than the classification of threats, but this may not be the case for other regions in the world which have more variability in the spatial distribution of threats, benefits, costs, and likelihoods of success. We believe this tool should inspire broad appeal as a strategy for prioritising conservation investments, and see its development as a first step towards making decisions about prioritising conservation finance approaches, such as debt-for-nature swaps, more robust, repeatable and transparent.

5.2 INTRODUCTION

Identifying priority areas for global conservation action helps to direct funding towards places that best deliver benefits to biodiversity (Brooks *et al.* 2006; Wilson *et al.* 2006). There is a tendency for environmental non-governmental organisations (NGOs) and scientists conducting conservation planning activities to create maps of conservation assets (e.g. species richness (Myers *et al.* 2000), centres of endemism (Orme *et al.* 2005), or the last remaining tracts of wilderness (Sanderson *et al.* 2002). While these maps are essential components of conservation decision-making, they are of limited utility as priorities in and of themselves because they do not tell us what actions to take in the places they identify. Proper prioritisation demands we make actions explicit, for example, establishing a protected area, lobbying the government, reducing poaching, or restoring a patch of degraded habitat (Game *et al.* 2013a; Brown *et al.* 2015b). Actions have probabilities of success and costs (Naidoo *et al.* 2006; Ban Natalie & Klein Carissa 2009), and importantly, deliver benefits to the conservation asset(s) via threat abatement (Ferraro 2009).

Understanding threats is essential when accounting for the expected benefits an action can deliver. For example, if there is no current or foreseeable threat to a site then protecting that site achieves little above and beyond having done nothing (Ferraro 2009; Devillers *et al.* 2014). Alternatively, if there is a substantive and unstoppable threat to a site with 100% displacement, then protection also achieves nothing. Efforts to map and understand the spatial and temporal distribution of threats to biodiversity (e.g. invasive species, ocean acidification, human development, etc.) continue to advance (Halpern *et al.* 2009; Venter *et al.* 2016; Stock *et al.* 2018), as do efforts to document the impact of threatening human activities to natural systems (Allan *et al.* 2017; Jones *et al.* 2018). Yet, the consideration of threats in spatial conservation prioritization remains largely inadequate, with many authors being unclear as to whether high or low threat areas are priorities (Game *et al.* 2013a; Tulloch *et al.* 2015). This lack of clarity emerges because authors are not prioritising actions. A high threat location is a high priority if the action being prioritised reduces or stops that threat, alternately a high threat location is a low priority if the action we are prioritising does little to abate the threat.

These inadequacies stem from analyses typically considering either single threats or cumulative threats. When considering single threats, such as run-off from land degradation (Tulloch *et al.* 2016), we risk narrowly focusing our management actions on addressing that particular threat (e.g. reducing land-clearing). In reality, multiple actions may be needed to safeguard biodiversity and meet conservation objectives (Tulloch *et al.* 2015). When more than one threat is considered, studies commonly rely on cumulative impact indices, which communicate threats as a single additive value (Halpern *et al.* 2008; Gissi *et al.* 2017). Using a single value to represent different kinds of threat likely masks those threats that will not be fully abated through any management actions (e.g. climate change or ocean acidification), potentially undermining the expected benefit of management interventions (Brown *et al.* 2013). Decision theoretic tools can help resolve these issues by explicitly linking actions to threats and benefits (Joseph *et al.* 2009; Klein *et al.* 2016; Di Fonzo *et al.* 2017). Here, we develop a strategic decision-support tool that enables the prioritisation of a particular type of conservation investment, debt-for-nature swaps.

Debt-for-nature swaps gained popularity in the 1980s and 1990s as a financial mechanism to address rapid declines in biodiversity within developing countries with high debt loads (Sheikh 2010). Debt-for-nature swaps are voluntary transactions whereby the donor(s) cancel, reduce or restructure the sovereign debt of a developing country and the savings are then invested into local conservation projects. Most debt-for-nature initiatives were focused on countries with tropical forests after the United States Congress authorised the "Tropical Forest Conservation Act" in 1998 specifically to enable debt restructuring programs for these threatened ecosystems (Sheikh 2010). The Nature Conservancy (TNC), a large environmental NGO was an early adopter of debt-for-nature swaps and are currently seeking to leverage funds for small Island Developing States (SIDS) and coastal nations which steward some of the world's most threatened fisheries and marine biodiversity (Roberts *et al.* 2002). Many of these countries also often have significant financial constraints (high debt ratios) that make it difficult to finance conservation and climate change adaptation measures to protect vulnerable communities and ecosystems. Setting up a debt-for-nature swap takes a long time and requires substantial finance, legal, political and environmental expertise. Due to the high transaction cost of implementing debt-for-nature swaps and the large potential impact these arrangements hold for conservation, there is a need to prioritize future efforts.

In this paper, we progress two broad goals. First, we develop a flexible new tool for answering the question of where an environmental NGO or philanthropic fund should invest in debt-for-nature swaps to generate the greatest expected return on investment. To do so, we use a four-step process (see Methods) beginning with a screening stage to narrow down the list of priority countries followed by a cost-effectiveness analysis or cost-utility analysis. Our second broad goal is to understand how priorities for debt-for-nature swaps change according to: 1) the classification of different threats as abatable and unabatable; and 2) different benefits and ecological weightings. We use the case-study of Caribbean countries to provide a proof of concept for our approach and construct 12 planning scenarios to evaluate these differences. We conclude with recommendations for future additions to the tools that will enable more flexibility in its conservation applications.

5.3 METHODS

Our approach is constructed around a four-part prioritisation protocol that considers: a) an initial screening stage where enabling factors and desirable conditions are compiled to evaluate whether a country should be considered as a candidate for a debt-for-nature swap, b) scenario construction, c) cost-effectiveness (or cost-utility) analysis, and d) post-hoc evaluation. Below we outline our four-part prioritization protocol, which is built into a spreadsheet (see Appendix D), followed by more descriptions of the proof of concept application in the Caribbean.

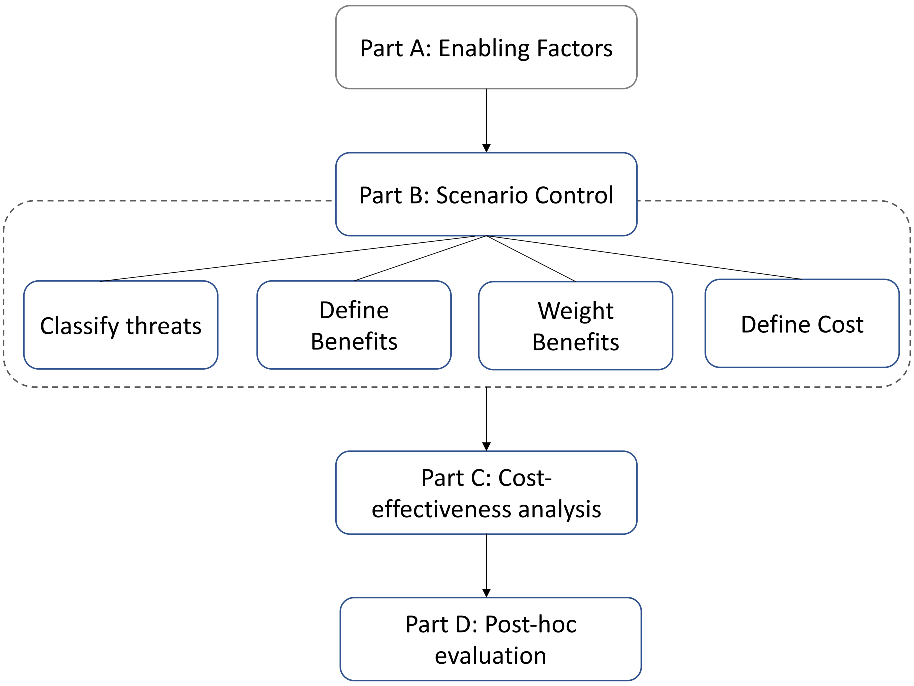


Figure 5.1 Schematic of the four-part workflow for the debt-for-nature swap prioritisation.

Part A: Enabling factors

This is the initial screening stage where users collate information that will determine whether a country will be included or excluded as a candidate for the debt-for-nature prioritisation process. Users will invariably have different perspectives on what constraints, or enabling factors, will be most relevant to their planning objectives. This could lead to attributes that consider socio-economic, governance or political factors related to the feasibility of conservation interventions (Brown *et al.* 2015b; Davidson & Dulvy 2017), reputational risk for the organization, or characteristics of the biodiversity present. Factors can be binary (yes-no), for example, does the country have debt available to purchase? Or, is the debt load within a desirable range? Enabling factors could also be relative values, such the country's debt to Gross Domestic Product (GDP) ratio. There is no rule of thumb for how to link enabling factors to the decision about a site's inclusion or exclusion. This decision could be based on expert judgement, thresholds (e.g. the debt load must be within a desirable range), or some combination of both determined by the user.

Part B: Scenario Construction

This stage of the process allows user to define what data and parameters should influence the prioritisation. There are four primary elements in this stage the user needs to consider. These include classifying threats, identifying benefits, assigning weightings, and incorporating costs

(Figure 5.1). The first component of this tool is that it allows users to classify threats as either abatable or unabatable depending on the intended conservation action. How threats are classified in this step influences the expected benefit delivered. The benefits considered are also defined and explored in this step and could include traditional measures such as the amount of important habitats available (e.g. total area of tropical forests or coral reefs existing in the country), or the size of the Exclusive Economic Zone (EEZ). The benefits can then be weighted by important biodiversity measures (e.g. species richness, endemic species, habitat intactness) or ecosystem services such as the potential for carbon sequestration, or fisheries revenue. Scenario development can include treating some parameters as optional, such as specifying some threats as "neutral" so they do not factor into the analysis, or choosing not to weight the benefit, or assign costs (in which case these values default to 1). Other aspects of the cost-effectiveness analysis described below can also be defined in this stage, including the alpha scaling factor, and probability of success values. In many instances, developing and evaluating multiple scenarios to test how priorities shift in response to changing parameters will be desirable. Having well-structured scenarios pre-defined is recommended (see "Case-study" below).

Part C: Cost-effectiveness analysis

Cost-effectiveness analysis identifies opportunities with the best value per dollar spent and our tool uses this to rank countries in terms of their priority. We tailored the cost-effectiveness (*CE*) formula to reflect an asset (e.g. a biodiversity benefit, *B*), abatable (I_a) and unabatable (I_u) threats to that asset, the probability of success (*P*) of the action (in this case, a debt-for-nature swap), and the total cost (*C*) of the action (e.g. transaction and implementation costs of entering into a debt-for-nature agreement) according to:

$$CE = \frac{BI_a(1-\alpha I_u)P}{C} \qquad (5.1)$$

where the biodiversity benefit *B,* is a weighted sum of conservation-related factor, and where (α) is a multiplier to ensure the effect of the unabatable threats does not return a negative value in the overall net benefit function. Therefore, alpha must be large enough to ensure $(1-\alpha Iu)$ is not negative. Subsequent to scenario construction, values are retrieved and stored from within the tool and organised next to each candidate site. The tool then applies Equation 5.1 to these stored values. The results of the analysis are then sorted and displayed as a bar graph from the most cost-effective to the least cost-effective sites.

Part D: Post-hoc analysis

While not the focus of this analysis, post-hoc evaluation of the results is recommended to assist with the ultimate decision of where to act first and in what order. In many instances, human decisions and opportunism will dictate how resources for conservation are distributed rather than the most optimal strategy (Game *et al.* 2010).

Case-study: **Caribbean**

TNC is actively pursuing the expansion of 3 million km^2 of secured marine areas globally, financed through debt-for-nature swaps in the next few years. The following proof-of concept has been tailored to TNC's preferences and available data. The Caribbean is a priority region for marine conservation due to increasing threats from fishing, coastal development and climate change, which continue to degrade important coral reef ecosystems and the species and communities they support (Roberts *et al.* 2002; Mumby & Harborne 2010).

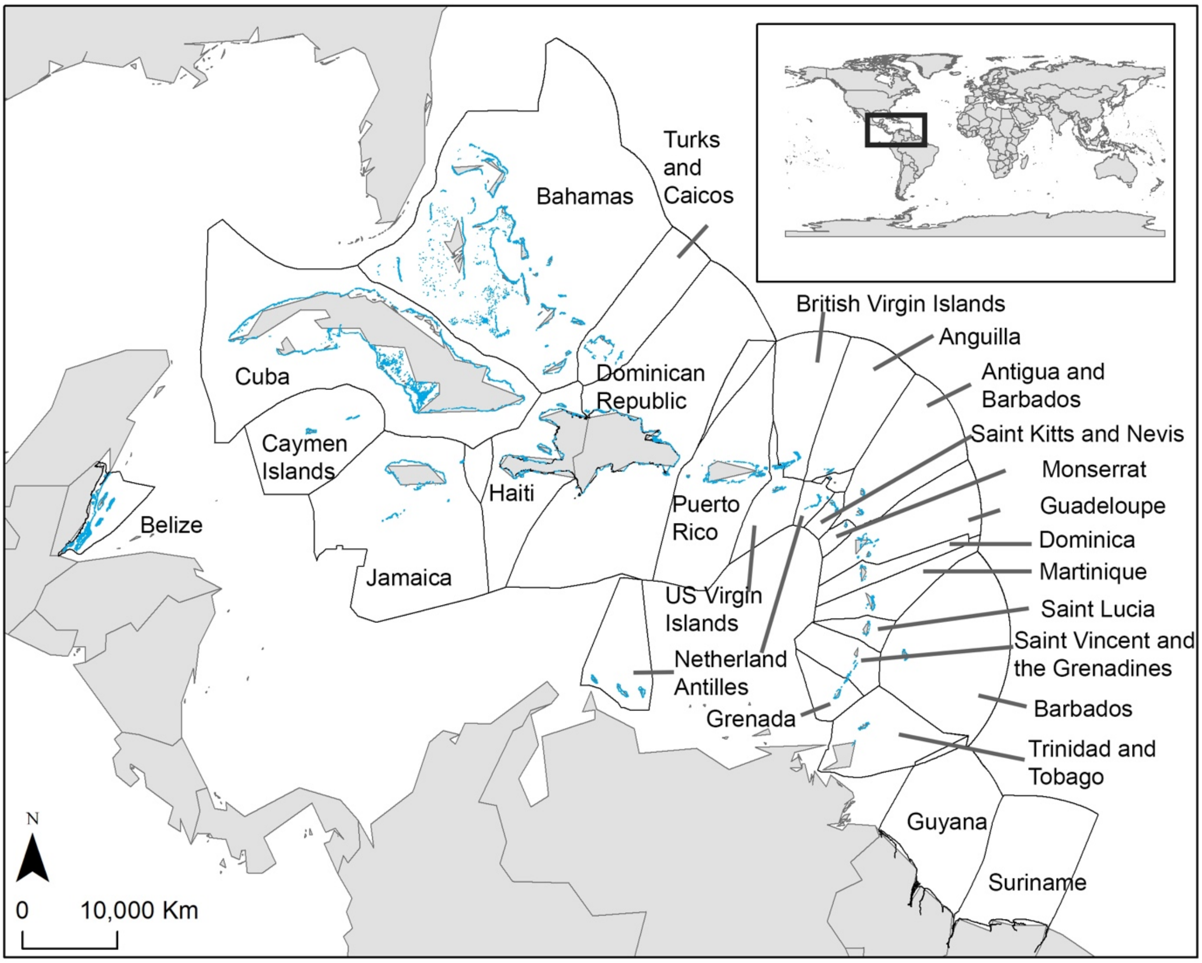


Figure 5.2 Map of the Exclusive Economic Zones (EEZs) of the candidate Caribbean countries and the distribution of coral reefs found within each (blue).

Enabling factors:

We compiled data for seven enabling factors based on existing TNC records for: the country's debt load, GDP, and debt to GDP ratio, as well as the presence of a TNC country program, the Official Development Assistance status of the country, if the county has expressed interest in debt conversion, and whether or not the country is currently part of a multi-lateral conservation initiative, such as The Micronesia Challenge (http://micronesiachallenge.org, last accessed June 4th 2018). Regardless of the values provided by these enabling factors for the Caribbean, we considered all countries as candidates for the proof-of concept in our analysis.

Scenario Construction

We considered two different benefits (B): the size of the country's EEZ (km^2) and the total area of coral reef habitat (km^2) found within each EEZ (UNEP-WCMC. et al. 2010). We used coral species richness per country from Roberts et al. (2002) as a weighting factor. Our primary aim for testing this proof of concept was to focus on the treatment of threats and the sensitivity of the prioritisation to changes in threat classification. Therefore, for cost (C) and the probability of success (P), we assigned values of 1.

Treatments of Threats

Knowledge of the spatial distribution and intensity of threats to marine ecosystems is required to improve ocean management and is an integral part of our strategic decision-support tool. Threats to marine systems stem from direct extractive activities (e.g. fisheries), land-based activities (e.g. sedimentation from coastal development), climate (e.g. acidification), and other streams of impact (e.g. oil rigs, commercial shipping, or invasive species) (Halpern *et al.* 2009). One of the primary conservation mechanisms advocated within TNC's debt-nature-swapping agreements is the establishment of marine protected areas (MPAs). The benefit of MPAs, which limit or exclude human activities, have been well-documented, particularly for fisheries recovery (McCook *et al.* 2010). Yet MPAs are not the panacea for all threatening marine processes and their ability to mitigate threats stemming from indirect activities, for example, heat stress from climate change, is limited (Game *et al.* 2008b). Our approach enables users to link threats to the conservation action(s) and expected benefits, therefore, we tested the influence of different classifications of threats according to the degree they can be mitigated by MPAs. We constructed three different treatments (Table 5.1) based on the classification of abatable and unabatable threats on Kuempel et al. (in review), which considers all fishing impacts, benthic structures and direct human impacts as abatable in relation to MPAs:

- Treatment 1: Abatable threats only (based on Kuempel et al (in review))

- Treatment 2: All threats considered (based on Kuempel et al (in review))
- Treatment 3: All threats considered, but where land-based threats could be abated with improved land management practices in addition to MPAs.

Our dataset also included commercial shipping, which we considered as an abatable threat given there is precedent for shipping lanes to change in response to conservation and zoning concerns (Dransfield *et al.* 2014). To assign threat estimates to each Caribbean country's EEZ, we used the mean values derived from 19 normalized data layers (values ranging between 0 and 1) available from the updated Human Impacts to Marine Ecosystems database (Halpern *et al.* 2008; Halpern *et al.* 2015) (Figure 5.3A).

Table 5.1 Threat data layers derived from Halpern et al. (2015) and three different treatments.

	Treatment 1 (abatable threats only)	**Treatment 2 (both considered)**	**Treatment 3 (land-based threats abatable)**
Artisanal fishing	✓	✓	✓
Demersal destructive fishing	✓	✓	✓
Demersal non-destructive fishing high bycatch	✓	✓	✓
Demersal non-destructive fishing low bycatch	✓	✓	✓
Oil rigs	✓	✓	✓
Pelagic high bycatch	✓	✓	✓
Pelagic low bycatch	✓	✓	✓
Shipping	✓	✓	✓
Inorganic nutrients	NA	✖	✖
Invasive species	NA	✖	✖
Night lights	NA	✖	✖
Ocean acidification	NA	✖	✖
Ocean pollution	NA	✖	✖
Plumes fertilizer	NA	✖	✓
Plumes pesticides	NA	✖	✓
Population	NA	✖	✖
Sea level rise	NA	✖	✖
Sea surface temperature	NA	✖	✖
UV	NA	✖	✖

✓ = abatable	✖ = unabatable	NA = not considered

By combining our benefits, weightings, and threat treatments, we derived 12 planning scenarios to prioritise Caribbean countries (Table 5.2). When unabatable threats were considered in treatments 2 and 3, we used the maximum scaling factor for alpha of 0.3.

Table 5.2 Scenario matrix derived from different combinations of benefits, weightings, and threat treatments.

	Treatment 1- Abatable threats only		
Benefits (km2)		No Weighting	Weighting-Coral Species Richness
	EEZ	1a	1b
	Coral	1c	1d
	Treatment 2- Both abatable and unabatable threats considered		
	EEZ	2a	2b
	Coral	2c	2d
	Treatment 3- land-based threats also abatable, unabatable threats considered		
	EEZ	3a	3b
	Coral	3c	3d

5.4 RESULTS

Classifying threats

Under threat Treatment 1 and 2, we classified eight threats as abatable and found that shipping is the dominant abatable threat in the Caribbean. The remaining abatable threats never rise above 0.1(Table 5.1, Figure 5.3A). In comparison, of the threats classified as unabatable under these treatments, ocean acidification poses the greatest risk but ultraviolet radiation (UV), sea surface temperature (SST) and sea level rise are also dominant (Figure 5.3B). Considering land-based threats as abatable (treatment 3) therefore makes minimal difference to the total threat scores as the average values for fertilizer and pesticides across countries is 0.0015 and 0.0003, respectively (Figure 5.3B). Due to this, we focus primarily on Treatment 2, which considers both abatable and unabatable threats as they relate to MPAs alone.

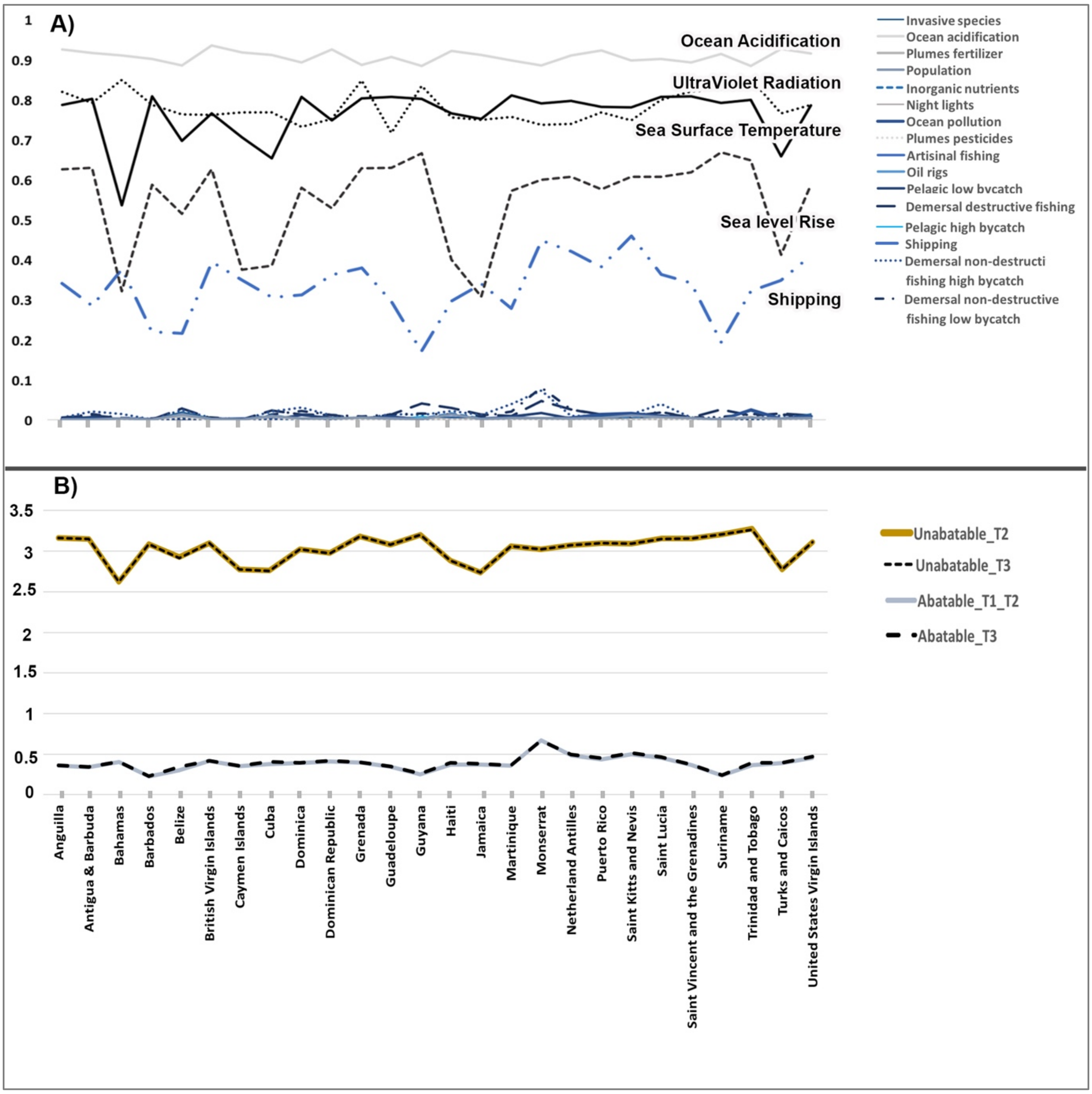


Figure 5.3 Relative threat values as derived from A) mean normalised values from Halpern et al, and B) additive threat values based on the threat treatments in Table 5.1.

Under Treatment 2, which considers both abatable and unabatable threats (Table 5.1), we found that abatable threats (aside from shipping), have significantly lower relative impacts than unabatable threats to marine ecosystems (Figure 5.3B). Those countries with the greatest cumulative abatable threats are Monserrat, Saint Kitts and Nevis, and Netherland Antilles. Those countries experiencing the most unabatable threats are Trinidad and Tobago, Guyana, and Grenada. Interestingly, Netherland Antilles is the only country with high threats that appears in the top 10 priorities across every scenario (Figure 5.4).

For each of our scenarios, we were interested in the countries falling within the top 10 priority rankings. Of the 25 Caribbean countries we considered in this analysis, half never fell within the top 10 priorities for any scenario (Fig 5.4). Countries that appeared in the top were Bahamas, Belize, Cuba, and Dominican Republic. Two countries, Antigua and Barbuda, and British Virgin Islands only appeared in the top 10 priorities when they came in tenth place. As expected, the scenario construction influenced which countries became top priorities and the benefits considered largely drove these differences. For example, when the size of the EEZ was the conservation benefit (Scenario 1-3a, b (Table 5.2)), regardless of the weighting, Bahamas was always the highest priority (Figure 5.5a). When the area of coral reefs was the conservation benefit (Scenario 1-3c, d (Table 4.2)), the highest priority was always Cuba (Figure 5.5b).

The priority ranking changes significantly for several countries as we alter the benefit chosen (EEZ or reef area). For example, Dominican Republic ranked 2^{nd} when the benefit was the size of the EEZ, but 6^{th} when the benefit was coral reefs. Similarly, Puerto Rico moved down from 5^{th} to 9^{th} (shown in green in Fig. 5.5c). Alternately, the Netherland Antilles moved from 9^{th} when the benefit was the size of the EEZ up to 5^{th} with coral reefs (shown in red in Fig 5.5c). For several countries, an appearance in the top 10 priority countries was linked entirely to a particular benefit. For example, Antigua and Barbuda, and Guyana, only appeared as priorities when the benefit was the size of the EEZ, and were never priorities for coral reefs. The opposite was true for Belize, and Turks and Caicos, which only appeared as priorities when the benefit was the area of coral reefs. Jamaica was the only country whose position never changed across scenarios, consistently appearing as the 4^{th} most cost-effective priority country across all scenarios.

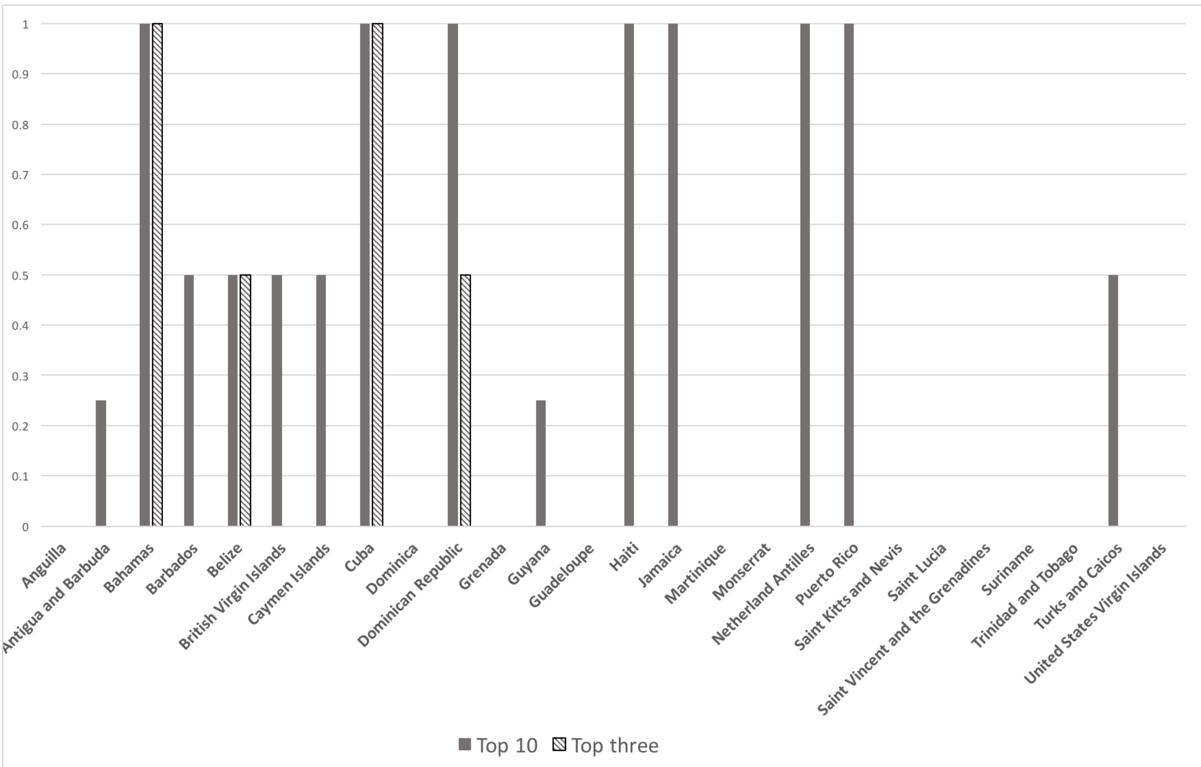


Figure 5.4 Proportion of times each country was found in the top 10 and top 3 solutions across all 12 scenarios.

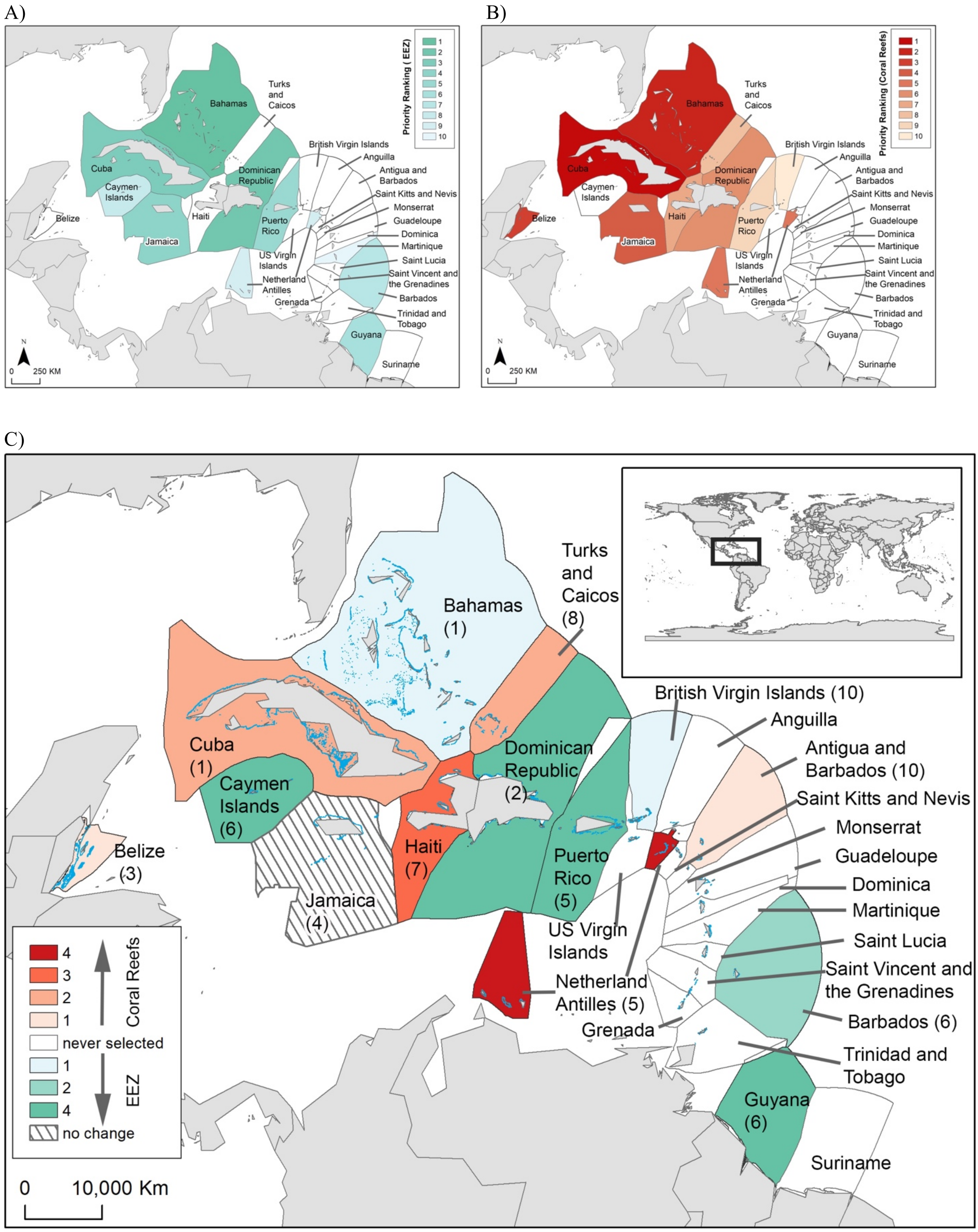


Figure 5.5 Outcomes of the prioritisation for A) Priority ranking for Coral Reefs; B) EEZ, and the variability in priority ranking for the top 10 countries under each benefit.

5.5 DISCUSSION

We develop a decision-support tool to help an organisation determine where it could get the highest return on investment for marine conservation by doing a debt-for-nature swap agreement. The approach has two broad components, first the facility to rule countries in, or out, of the analysis based on a number of political, economic and financial criteria, and a cost-effectiveness analysis that ranks the remaining countries in terms of the net threat averted to one or other benefits per unit dollar. A major component of our approach is the novel integration of threats into the prioritization process which quantifies the relationship between the proposed conservation action(s) and the expected benefits the action(s) can deliver. Our intention was not to be overly prescriptive about threat treatments or benefits, but to illustrate the utility of the tool for decision-making. Therefore, we intentionally conducted a simplified analysis (e.g. choosing to set cost and the probabilities of success equal to 1, and allowing all countries to remain in the feasible set) in order to understand how the tool works in its simplest form. We found that even with this simplified version, our results were surprisingly complex.

We note that across our scenarios, the benefit chosen influenced priorities more than the treatment of threats or benefit weightings. However, we do not suggest this is the case for other regions. The low variability of threats across the Caribbean (Figure 5.3B) suggests the resolution of threats modelled at a global scale masks the influence of threats with highly localised spatial distributions, such as fishing effort. While this was the best publicly available data for our proof-of-concept, we encourage intra-regional prioritisations to incorporate region specific data on threats, where possible. We also note that if we define biotic benefits, such as coral reef habitats, by the area of habitat available, we assume the benefit scales by area. In reality, reefs and other biota will vary in their potential to deliver benefits based on their condition, composition and local pressures (Evans et al. 2015). Often this information is not available at large spatial scales, particularly for marine systems, and so a prudent best practice for future applications would be to test the sensitivity of results that consider the relationship between variability in habitat condition and expected benefits through simulated forecasting (e.g. monte-carlo approaches; (Harris et al. 2018)).

Integrating threats directly into the benefit function creates a more nuanced analysis than what has been developed to date. Traditional strategies to address threats typically identify places under high threat and avoid them completely, or proactively target them (Game *et al.* 2008a; Boon & Beger 2016). Following this logic, the Netherland Antilles, which has one of the highest additive threat values of any country in the Caribbean, would either never or always be a priority for conservation.

However, we found that Netherland Antilles always appeared in the top 10 priorities, albeit with different rankings, illustrating the value of quantifying the relationship between threats that can be abated and the potential benefits provided. In some instances, the impacts of unabatable threats which cannot be addressed through local management (e.g. ocean acidification, surface warming and sea level rise) might outweigh the potential benefits and result in a lower priority ranking. Interpreting results will also be compounded by the probability of success and costs, which may vary substantially by users and contexts.

The straightforward combination of a set of decision rules with a cost-effectiveness analysis means that the tool can be easily expanded. In its current form, single benefits are considered independently but there is scope to build in a weighted sum as follows:

$$B = \beta B_1 + (1\text{-}\beta B_2) \qquad \text{(Eq. 5.2)}$$

where the total benefit, B, is derived from two benefits (B_1 and B_2) which are weighted by a beta parameter (β). This approach solves planning challenges where multiple-objectives are to be achieved (Beger *et al.* 2015).

We consider all threats to be additive in their nature, but much research has been put into understanding how different threatening processes interact with each other in non-additive ways (Darling & Côté 2008; Brown *et al.* 2013). Developing a mechanism to account for threats that have known antagonistic (cancelling out the impacts), or synergistic (enhancing the impacts beyond the additive values) relationships could build in more ecological complexity. Further, in our case-study we consider all additive threats to have the same impact to both of the benefits considered. In reality, the spatial distribution of threats and their impacts on the expected benefits will vary. A priority for future research would be to develop more complex functions that relate actions to the expected benefits, as many conservation actions will not fully abate all the impacts of threats. For example, an effectively managed MPA may remove fishing from strict no-take MPAs, but fishing will continue to impact the ecosystem in the surrounding areas (Edgar *et al.* 2018). Likewise, shipping may be redirected via an MPA to avoid high-risk areas where ship-strikes are likely to occur, but the noise pollution created from the shipping will still broadly impact the system (Gissi *et al.* 2017).

Several key limitations to using a cost-effectiveness approach exist that should be mentioned. For example, it ignores the fundamental spatial planning concept of representation, which aims to ensure a portion of every biodiversity feature (e.g. species, habitats, and processes) receives protection (Moilanen *et al.* 2009c), not just the elements of biodiversity selected as benefits (Brown *et al.* 2015b). Representation is achieved through selecting complementary suites of sites that avoid duplication of effort (Kukkala & Moilanen 2013). Cost-effectiveness analysis inherently ignores site-based dependencies by treating each site as its own individual unit. In addition to this, costs, benefits, and probabilities of success must also be considered as independent elements when in fact there may be dependencies between them.

These limitations should be viewed as development priorities for future iterations of our cost-effectiveness approach, which should have broad appeal as a strategy for prioritising conservation investments. People use cost-effectiveness in most aspects of their daily lives (e.g. shopping), so applying it to solve conservation challenges is both pragmatic and sensible. Conservation must move towards embracing more translational ecology, where stakeholders, ecologists, and decision-makers work collaboratively to develop research that results in improved decision-making (Wall *et al.* 2017). We see the development of this tool as a first step towards making decisions about prioritising conservation finance approaches, such as debt-for-nature swaps, more robust, repeatable and transparent.

6 Discussion

Sustaining global biodiversity and human livelihoods requires coordinating global conservation efforts. Goals for all aspects of biodiversity, ecosystem function, human health, and economic development have been adopted through the Convention on Biological Diversity (2011) Aichi targets and the United Nations' Sustainable Development Goals (https://www.un.org/sustainabledevelopment/sustainable-development-goals). While we have made much progress, we are not achieving our conservation targets (Watson *et al.* 2014; Butchart *et al.* 2015; Klein *et al.* 2015; Kuempel *et al.* 2016) and biodiversity continues to decline (Ceballos *et al.* 2017; Maron *et al.* 2018). Systematic conservation planning can assist governments, NGOs and practitioners to manage natural systems and deliver on these goals (Margules & Pressey 2000). This thesis focuses on one aspect of the much broader conservation planning process – spatial prioritisation - the identification of how, when and where to act to efficiently achieve conservation goals (Pressey and Bottril 2009; Kukkala & Moilanen 2013). There are many ways to prioritise conservation and management actions (e.g. restoration, protection, policy changes) in space and time (***chapter 1***). However, approaches based on the construction of a well-defined quantifiable problem, which links actions to objectives, best equip decision makers with a structure for identifying conservation challenges and establishing clear paths to finding efficient solutions (Groves & Game 2015; Schwartz *et al.* 2017). In this thesis, I examine a broad spectrum of applications around problem-based conservation prioritisations to illustrate their utility for decision-making. I deliberately chose a diversity of problems linked by this common theme to demonstrate its broad application to solving conservation problems. In this final chapter, I first summarise the major findings of each research chapter, then the contributions this thesis offers to the field of decision science for spatial prioritisation, and I conclude with limitations, reflections and future directions.

6.1 Main findings

6.1.1 Integrating research using animal-borne telemetry with the needs of conservation management (chapter 2)

Uncertainty is common in conservation decision-making. We often do not have the time, resources, or capacity to reduce all uncertainties when managing for natural resources (McDonald-Madden *et al.* 2008). The most important uncertainties to resolve are those that are likely to alter our decisions. However, there is a tendency for many scientists to justify their research by what they perceive to be its value for conservation, particularly when it comes to animal demography studies (Campbell *et al.* 2015). For example, understanding individual movement patterns (e.g. where sharks go;

(www.ocearch.com)) drives enormous amounts of research effort in the name of outreach and conservation. However, reducing the greatest uncertainties is not equivalent to reducing the most important uncertainties –the ones most likely to change management decisions. Chapter 2 focused on framing an important and timely message directed at the emerging field of animal telemetry. I first developed a framework for the different ways scientific pursuits can impact the broad field of conservation: from pure scientific research, through to community engagement and, ultimately, to informing management decisions (McGowan *et al.* 2016). While not comprehensive, this framing forces researchers to recognise the difference between indirect and long-term impact, versus direct, short-term and quantifiable impact on conservation decision-making. I urge that more caution be taken before costly data collection is undertaken with the justification of informing conservation by reducing uncertainty. I suggest value of information analysis (VoI) which can be used to examine the trade-off between the ability of new information to reduce decision uncertainty and the costs of collecting more data; which uncertainties may be most important to reduce in order to improve gains in management outcomes (Runge *et al.* 2011); or what the financial value of gaining new information is worth to management (Maxwell *et al.* 2014), be more widely embraced and embedded in the decision-making process. To support this claim, I provide a decision tree to illustrate when new information should be pursued as opposed to managing with uncertainty.

6.1.2 An evaluation of marine Important Bird and Biodiversity Areas (IBAs) in the context of spatial prioritisation (chapter 3)

Identifying locations on the land and sea based purely on their ecological attributes is a common approach driving global protected area policy (Eken *et al.* 2004). Using a combination of expert-derived knowledge and ecological data, these approaches elevate the status of the delineated sites because they meet important ecological criteria, and are then considered to be conservation priorities (BirdLife International 2010c). Di Marco *et al.* (2015) examined sites derived from a prominent criteria-based site prioritisation approach, BirdLife International's Important Bird and Biodiversity Areas (IBAs) and compared them to priorities derived from the problem-based decision support tool Marxan, and found much congruence within terrestrial systems. However, no one had tested the performance of IBAs to capture priorities for important aspects of biodiversity (e.g. species and habitats) in the sea. Chapter 3 filled this gap with an analysis to explore how to best incorporate these sites in problem-based spatial prioritisations which aim to represent biodiversity and minimise impacts to users in marine systems (McGowan *et al.* 2017b). Using the case study of Australia's EEZ, I found that while marine IBAs do not adequately represent other important aspects of marine biodiversity on their own, they do possess unique ecological characteristics that are unlikely to be captured by typical surrogates (e.g. benthic habitats) and should be included as biodiversity features to be represented (e.g. important migratory bottlenecks). Further, when testing different treatments and representation targets for these sites, I found the best strategy was to treat each location individually with its own target. To further develop this best practice guideline, I highlight the rules to determine appropriate targets should be contingent on the underpinning criteria, which are not currently released with the spatially explicit IBA locations. Thus, I call for more transparency regarding the process of delineation, the underpinning criteria and the attributes of the sites in order to progress strategic planning for marine protected areas. Given the increasing prominence of Important Bird and Biodiversity Areas (IBAs) and Key Biodiversity Areas (KBAs) in global conservation governance, this paper provides a path forward for countries and regions to integrate systematic conservation planning with these criteria-based site identification approaches.

6.1.3 Optimal ocean zoning within a sparing versus sharing framework (chapter 4)

The sparing versus sharing dichotomy is the focus of ongoing scientific inquiry, particularly in production landscapes which aim to balance the demand for food production and biodiversity conservation (Fischer *et al.* 2013; Phalan 2018). Some argue that intensely cultivating areas in a smaller footprint allows more space for nature (sparing), while others argue that less intensive techniques distributed over a larger portion of the landscape (sharing) can achieve equivalent yields with better outcomes for biodiversity (Phalan *et al.* 2011). The same dichotomy occurs in the sea but ocean management has never been framed in such a way. The primary goal of this work was to first translate the sparing versus sharing framework to be relevant for marine policy. By identifying three primary management zones: open-access areas, managed fishing zones, and no-take marine protected areas, I provide a departure from the "spare vs. share" allocations that dominate the terrestrial debate. I conceptualised seven possible zoning allocations comprised of spared, shared, and spared/shared seascapes, addressing previous calls to reframe the dichotomous debate (Kremen 2015a). The second goal of this research was to advance the debate in the sea by providing process model simulations. My simple process model, coupled with a clear problem formulation serves as a decision support tool that enables managers to derive optimal seascape allocations for coastal environments. The process model is comprised of several parameters that could be easily estimated for different regions around the world where a single type of fish dominates the fishery. This includes, for example, the relative costs of enforcing protected areas versus fisheries management zones, the growth rates of the fished species, and the management budgets. The problem definition maximises biodiversity whilst meeting a minimum fisheries yield, and allows the user to understand how optimal zoning allocations change as a function of the budget. Ultimately, I found a robust rule of thumb that managers should invest in establishing marine protected areas when budgets are small, but that as the management budget grows, investing in fisheries management alongside protected areas (sparing and sharing) becomes the optimal strategy (McGowan *et al.* 2018). I concluded by making several recommendations for future development of both the theory and modelling approach, including testing this rule for a variety of case-studies around the world.

6.1.4 A flexible decision-support tool to initiate debt-for-nature swaps using conservation finance (chapter 5)

Small Island Developing States and coastal nations steward some of the world's richest marine biodiversity (Roberts *et al.* 2002; Beger *et al.* 2013; Weeks *et al.* 2014). Coral reefs, mangroves, seagrass, and their associated ecological communities provide food security for millions of people (Cabral & Geronimo 2018) and many other critical ecosystem services (e.g. buffering vulnerable communities from the impacts of natural disasters (Bayraktarov *et al.* 2015), and sequestering carbon (Atwood *et al.* 2017). These countries also often carry large financial debt, making investments in conservation challenging. Debt for Nature swaps are a promising conservation finance mechanism to assist with restructuring a nation's sovereign debt in exchange for protecting its biodiversity. In this chapter, I developed a user-inspired tool for a large conservation NGO, The Nature Conservancy, to assist them with future debt for nature swapping agreements in order to promote the establishment of marine protected areas. The tool provides a robust, problem-based prioritisation strategy via a four-part protocol that links the expected benefits to the types of threats that can and cannot be abated by spatial protection. I used the Caribbean as a proof of concept to understand how different classifications of marine threats (e.g. abatable versus unabatable) and different kinds of benefits (e.g. area of EEZ and coral reefs) influence investment priorities. I found that the choice of benefit influenced the outcomes more than the classification of threats, but this may not be the case for other regions in the world which have more variability in the spatial distribution of threats, benefits, costs, and likelihoods of success. I conclude with several recommendations to make the tool more useful to implementing organisations, such as including a weighted sum of two or more benefits, integrating more nuanced interactions between threats (Brown *et al.* 2013), and generating plausible measures of the likelihood of success based on economic, political, and ecological factors (Davidson & Dulvy 2017).

6.2 Major contributions

My thesis draws from the discipline of decision science, a field of research that integrates concepts from economics, mathematics and operations research, to address conservation challenges and encourage better decision-making for natural resource management. I focused on two streams of research related to data and tools. Below I outline the major contributions of this work to progress the field of decision science for conservation.

i) General value of information theory is starting to have relevance for conservation science and other areas of applied ecology (Maxwell *et al.* 2014; Canessa *et al.* 2015). However, unless we have examples from specific sub-fields the idea is unlikely to gain traction. **Chapter 2** is one example of placing the idea of value of information theory into the language and context of scientists interested in tracking animals. This is an especially important step for the field of animal telemetry because much of this work involves capturing and tagging individuals, which can be contentious (McCarthy & Parris 2004) and costly. Importantly, the decision tree I provide leads users (managers or scientists) interested in collecting data for conservation and management decisions through the decision context by linking the data back to the objectives and actions, and therefore has broader relevance to any sub-field of ecology.

ii) There is a long-standing tension between the communities of practice that focus on individual sites to prioritise places for protected areas (and/or other conservation actions) and those that focus on whole systems (e.g. protected area networks) (Knight *et al.* 2007). The latter approach explicitly recognises the truism that an entire network is not the same as the sum of its individual parts because of the interactions between sites in the system. These interactions include the ideas of representation, irreplaceability, connectivity and cost, to name just a few (Ferrier *et al.* 2000; Moilanen *et al.* 2009c). Yet identifying sites based on their attributes (**chapter 1)** continues to have huge appeal and drives global conservation priorities such as the IUCN's Key Biodiversity Area (KBA) program, the extended descendant of IBAs (**chapter 3**). The updated KBA standards explicitly state the need to merge these two approaches (IUCN 2016) and find synergies between their respective communities of practice. To date, guidance on how best to do this has been limited. I provide a first attempt to provide one path towards a resolution of that tension in **chapter 3.**

iii) There is a recognised mismatch between the complexity of conservation challenges and the simplicity of the institutions, assumptions and tools practitioners rely on to solve them (Game *et al.* 2013b). Yet, the path to good decision making does not require equally complex tools. To the contrary, good decision making requires pragmatism and procedure and therefore, intelligent but often drastically simplified "models" of these complex systems linked back to the conservation objectives (Starfield 1997a). Simple tools to guide decision-making are becoming more commonplace and I developed two additional but unique examples in this thesis. In **chapter 4**, I demonstrate how strategically simplifying a process model for a single fished species can help managers decide on the optimal zoning allocations for coastal and marine resources. This work sets a tool-based precedent for the decades old sparing versus sharing debate, which had never been underpinned by such an approach to date (McGowan *et al.* 2018). In **chapter 5,** I developed and demonstrate how a customised cost-effectiveness calculator can enable high level prioritisation strategies for the promising conservation finance mechanism, debt for nature swaps.

6.3 Limitations, reflections and future research

The research presented in this thesis addresses the role of decision-support tools in spatial prioritisation and conservation decision-making. In this section, I focus on the limitations of my research chapters, reflect on the broader limitations of problem-based approaches for spatial conservation prioritisation, and discuss future directions to build on this work and advance the uptake of decision-support tools throughout the field of conservation planning.

6.3.1 The problems with problem-based prioritisation

Problem-based prioritisation requires a series of structured steps (Possingham *et al.* 2001). First the stakeholders need to set clear objectives in order to identify and then select actions that will best achieve the intended outcomes (Tear *et al.* 2005; Groves & Game 2015). To do so requires models that link actions to outcomes and this includes specifying the constraints that bound the decision variables (e.g. budgets, harvests, targets, etc.). Problem-based prioritisations can take many forms and I explored three different problem formulations in this thesis: **chapter 3** used an existing decision-support tool, Marxan, which relies on the well-defined mathematical 'minimum-set problem'. I used Marxan to minimize impacts to users whilst ensuring a minimum target constraint was met for the representation of habitats and species in marine protected areas; **chapter 4** optimized the proportional allocation of a seascape by maximizing a conservation benefit - the amount of fish remaining in the sea, whilst ensuring a minimum harvest constraint was met; and **chapter 5** used economic principles to prioritise countries based on their potential return on investment from engaging in debt for nature swapping arrangements. All of these problems allow for the explicit consideration of costs, benefits, constraints, weightings, and identification of the key factor(s) to be maximised (e.g. benefits) or minimised (e.g. impacts). Below, I discuss some of the limitations and opportunities for future research for problem-based prioritisation.

Whose problems and whose benefits: values still drive problem-based prioritisations

Defining objectives and constraints is invariably the hardest and most subjective component of every problem-based approach. Given how many different possible combinations of elements there are, problem formulation will vary significantly by who constructs it, their interpretation of the challenge, and the values they impose. This begs an important question*: Are these the correct problem formulations for the challenges we seek to address?* For example, if we take the sharing versus sparing problem as mentioned above and defined in **chapter 4,** my objective was to maximise biodiversity (represented by standing stock biomass (e.g. the fish remaining in the sea)). In order to meet fisheries objectives, I considered a minimum harvest threshold as a constraint on the system. We can see how fisheries advocates and resource dependent communities may not approve of their needs being reflected in the problem formulation as a minimum constraint to be met. To the contrary, they would likely hope to see their needs being maximised, or at least, become part of a multi-objective problem aimed at maximising both fisheries and biodiversity benefits simultaneously (Gaines *et al.* 2010; Brown *et al.* 2015a). If we consider the debt for nature swapping tool developed in **chapter 5**, we defined the benefits to be the area of coral reefs and the size of the EEZ belonging to each country, but could have just as likely, data permitting, used the proportion of a country's GDP produced from fishing exports as a benefit, or the number of threatened endemic species found in the country's waters (Davidson & Dulvy 2017). The same conclusions can be drawn for the costs we considered (Naidoo *et al.* 2006; Ban *et al.* 2009): a proxy for opportunity cost to fishers in **chapter 3**, the staffing costs for monitoring MPAs versus regulating fisheries in **chapter 4**, and an equal cost of investment in **chapter 5**. Problem-based prioritisations, as with attribute-based approaches, can be value-laden, suffer from data limitations and uncertainty, and peppered with subjectivity (e.g. the targets we set for biodiversity as in **chapter 3** (Carwardine *et al.* 2009))**.**

Further, *who gets to decide what the suite of possible actions are?* For example, in **chapter 3** I only considered the single action of making marine reserves to protect Australia's marine biodiversity, meaning any part of the sea was either in the reserve network, or outside of it. However, some argue that IBAs are not intended to be protected, but rather monitored with diligence to prevent human activities from degrading the unique features found within (Smith *et al.* 2018). In this light, a three-zone action plan might be more desirable to know where to protect, where to manage, and where to do nothing. In **chapter 4**, the sparing versus sharing optimization, I did account for three zones (protection, management, and open-access), but I did not specify what types of management action should be taken in the managed zone (e.g. reducing fishing effort or changing policies around gear restrictions). The difference in actions will impact stakeholders in different ways. Finally, in **chapter 5**, the action was to promote conservation through debt for nature swaps. Some people might think debt-for-nature swaps are heavy-handed western capitalist approaches, aimed more at promoting international collaboration than at shifting perceptions about smart resource use in the debtor countries (Hansen 1989). Such a top-down approach may adversely affect the resource-dependent and often vulnerable local communities in the countries whose governments decide to commit to a debt for nature swap (Smith *et al.* 2009).

Being explicit but hiding assumptions

While problem-based prioritisations emphasise the need to make objectives, actions, and uncertainty more explicit (Runge *et al.* 2011; Brown *et al.* 2015b), linking actions to outcomes is rife with implicit assumptions of how processes play out across a range of factors. One well-noted implicit assumption is the "scorched earth" reference in protect area planning, where the non-reserved areas are treated as though they will be drastically degraded without protection and thus, contribute nothing to conservation objectives (Edwards *et al.* 2010). The response to this criticism was to begin accounting for the contribution of non-reserved areas towards benefits either through system-wide process models or multiple types of zones (e.g. **chapter 4** (Watts *et al.* 2009; Edwards *et al.* 2010)). More recently there have been calls for establishing specific targets for biodiversity retention in areas outside of the protected area estate (Maron *et al.* 2018).

The flip side of the scorched earth assumption, is the expectation that once an area becomes protected, there will be a complete halt to the destruction of habitat or species in that location (**e.g. chapter 4**) or that the protection will remain in perpetuity. There is overwhelming evidence that PAs globally are falling short of effective management (Gill *et al.* 2017; Jones *et al.* 2018), and many PAs continue to be downgraded and degazetted as a result of changes in national policies (Mascia & Pailler 2010). In reality, even if management was 100% effective and all destructive ocean-based human activities ceased in the PA, the potential for other impacts (e.g. run-off from land clearing (Kroon *et al.* 2016; Tulloch *et al.* 2016) or climate change (Cheung 2018; Hughes *et al.* 2018), to diminish the expected benefit remains high but often unaccounted for. In **chapter 5,** I considered threats that can and cannot be abated by the intended action in the problem formulation, but implicit assumptions were made about how the impacts of unabatable threats affect the benefits (Kuempel in review).

Additionally, there is the implicit assumption that the action will undoubtedly succeed and go on to deliver the expected benefits. Explicitly integrating a probability of success into a problem-based prioritisation tool is becoming more commonplace, for example, the Project Prioritisation Protocol of Joseph *et al.* (2009), or the Cost-effective Resource Calculator of Di Fonzo *et al.* (2017), both of which rely on expert knowledge to estimate these values for a single acting agency. Yet, the probability of success of a conservation action may vary substantially across borders, communities, actions, institutions, the scale of planning (Mills *et al.* 2010) and through time (Cheok *et al.* 2017). The interactions among and between these factors make estimating the probability of success a critical research gap (Halpern *et al.* 2013; Giakoumi *et al.* 2018), particularly for prioritising between countries at a global scale (Davidson & Dulvy 2017) although some expert-based protocols exist (Klein *et al.* 2017). More coordinated synthesis of what works, when, and in what context (Sutherland *et al.* 2004) will help illuminate important factors influencing when conservation actions are likely to succeed (Mascia & Mills 2018). In the near-term, bringing our implicit assumptions into the light and focusing research on developing and parameterising more complex process models to explore the impact these assumptions have on our results should be a research priority.

6.3.2. Looking to the future: Democratising spatial prioritisation

Equipping conservation practitioners with the skills they desire to make better decisions is something I have focused on over the last five years. I have trained over 600 people from 25 countries on six continents in the spatial prioritisation tool, Marxan. The demand for trainings and technical support continues to grow and this demand is not being adequately serviced. These experiences have equipped me with intimate knowledge of the challenges users face when developing spatial prioritisation projects for conservation on the land and sea, and subsequently, where opportunities exist to improve their uptake and execution globally. The following section is dedicated to the essential bottlenecks we must overcome to democratise decision-support for spatial prioritisation. While many of my conclusions are drawn from my experience with Marxan, I believe them to be relevant to any problem-based prioritisation approach and/or tool (such as the debt-for-nature calculator developed in **chapter 5,** among many others).

Building a network of practitioners who are familiar with the theory and principles of decision-making for spatial prioritisation requires bridging the gap between research and practice (Knight *et al.* 2008). A recent review by Sinclair *et. al* (2018) found that 74% of spatial prioritisations conducted with implementation in mind were reported to deliver on the ground actions and the majority of these used computer-based decision-support tools (34% of which used Marxan). This review shows encouraging evidence that the 'science-implementation' gap (Knight *et al.* 2006) may be narrowing when it comes to the uptake of decision-support tools in practice, but a systematic evaluation of the impact of these tools in planning is still required (McIntosh *et al.* 2016).

The value of information

Discussions about prioritising conservation actions overwhelmingly begin with issues of data paucity. This is not surprising if we consider that ecologists and/or data scientists are typically the people tasked with driving conservation efforts (Johnson *et al.* 2015). In **chapter 2** I focus on challenging the common perception that more data will lead to better decisions. While important to know the limitations of data, data-deficiency and uncertainty is not a reason to postpone planning (Giakoumi *et al.* 2011). We must work to make this argument more salient across the field of conservation. For example, there are dozens of papers showing that broad habitat surrogates can effectively represent biodiversity in the absence of species-specific data (e.g. Ward *et al.* 1999; Grantham *et al.* 2009a; Dalleau *et al.* 2010; Grantham *et al.* 2010; Sutcliffe *et al.* 2015), particularly when complementarity-based approaches are used to build spatial plans (Lewandowski *et al.* 2010). But far too few examples of value of information analyses exist for other conservation applications (but see Mazor *et al.* (2016b) for a partial VoI; Maxwell *et al.* (2014) and Canessa *et al.* (2015)). Pursuing more VoI analysis in the context of spatial prioritisation remains an important research priority, but transitioning those from data-collectors and data synthesisers into decision-makers (Gregory *et al.* 2012; Johnson *et al.* 2015) is less about data and more about training in problem formulation.

Formulating the problem

Clearly defining and articulating what we want to achieve in conservation is a difficult challenge. Without putting sufficient rigour into problem definition, we run the risk of solving the wrong problem, wasting resources, and pursuing projects that are not aligned with our strategies and objectives (Game *et al.* 2013a). From my experience, many people try to adjust their problems to fit a particular tool (e.g. Marxan) rather than fitting or building a tool to solve a well-defined problem (e.g. **chapter 5**). To address this, I believe an essential research need is to provide more "matchmaking" between conservation problems and available decision-support tools. While not comprehensive, there are common "classes" of problems (e.g. building representative networks of protected areas (**chapter 3**), optimising between actions (**chapter 4**), evaluating return on investment across a suite of projects or sites (**chapter 5**), maximising a conservation benefit (**Appendix A**), which could be presented as a typology of conservation problems that can be addressed with structured decision-making tools. Such a classification could be broadly based on problems that have mathematical (e.g. **chapter 4**) or algorithmic expressions (e.g. **Appendix A**). Decision-trees, such as the one I developed for animal movement data in **chapter 2,** are useful exercises to reframe the way people think about problem solving and provide structured guidance. Developing decision-trees to lead a user towards a well-crafted problem formulation and subsequently through to the types of support-tools that exist to solve the problem, is an important next step for conservation planning and prioritisation.

Storytelling and Communication

Involving stakeholders and accommodating different values in the process of defining the challenge and objectives is critical to the success of any conservation effort (Biggs *et al.* 2011; Ban *et al.* 2013; Bennett & Dearden 2014; Giakoumi *et al.* 2018). In regards to spatial prioritization, there are many decision-support approaches and tools which enable the integration of multiple values and stakeholder groups (Game *et al.* 2011; Grantham *et al.* 2013; Jumin *et al.* 2017; Gissi *et al.* 2018), including participatory planning approaches (Merrifield *et al.* 2013). Many of these have delivered implemented spatial plans (Sinclair *et al.* 2018) and yet a pervasive mistrust of decision-support tools persists around the world. Synthesising and communicating real-world stories and the role decision-support tools played in the process is essential to build confidence among stakeholders.

Concluding remarks

Too much of conservation thinking has been driven by scientists looking to answer questions that no one has specifically asked. It can also be said that too much of conservation thinking has been driven by scientists providing information, often spatial information (e.g. maps of assets and threats), that appear to be prioritisations but they imply no action. Conservation implementing agencies, such as community groups, governments, environmental non-government organisations, always have actions they can do, or not. These actions involve resources (money and time) and they occur in a place (even if the action is non–spatial by nature - e.g. lobbying or policy reform), it has a place in a state/country). It seems logical to bear these actions in mind at the beginning of any prioritisation. The agencies invariably have broad goals in mind, often these need to be turned into more specific quantifiable objectives. They will also have ideas about how their actions alter outcomes and which of those outcomes relate to their objectives. Translating the hopes, dreams and fears of end-user organisations is a time-consuming and challenging task. But in the end, it is only through a deep and detailed understanding of end-user needs that we can deliver the best decision-support tools for conservation.

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Appendix A. **Flagship species can deliver efficient conservation**

This section is reproduced from the following submitted paper, with some alterations to the structure and formatting:

McGowan J, Beaumont L, Chauvenet A, Atkinson S, Esperon-Rodriguez M, Mittermieir J, Baumgartner J, Beattie A, Dudaniec R, Grenyer R, Harcourt R, Nipperess D, Smith RJ, Stow A & HP Possingham. Flagship species can deliver efficient conservation. *Nature Ecology and Evolution-submitted.*

A1. ABSTRACT

Conservation strategies based on flagship species such as the tiger, panda and gorilla, successfully attract publicity and financing from community and corporate donors. Criticism of this approach comes from advocates of place-based conservation claiming greater efficacy by focusing on areas with multiple conservation assets such as wilderness or endemic species richness. Therefore, the question we ask is simple – to what extent is a flagship species approach a major constraint on delivering efficient and effective place-based conservation? To answer this, we constructed a novel place-based strategy that incorporates flagship species. This achieved 80-89% of the objectives, such as total species conserved, of the purely place-based approach (as measured by 19,616 species ranges). For the first time, our integrated approach provides strong evidence that prudently selected flagships can realize broad conservation objectives. It also equips organizations using flagships to optimise public awareness and funding while prioritizing aspects of both place- and species-based conservation.

A2. INTRODUCTION

There is on-going scientific debate as to whether conservation and management actions to combat the current biodiversity crisis are optimized by approaches that prioritize places versus those that prioritize species, such as via 'flagship' species. Identifying priority locations for global conservation actions helps to direct limited funding towards places that best deliver benefits to biodiversity (Brooks *et al.* 2006; Wilson *et al.* 2006). Therefore, much of conservation has focused on mapping the distribution of biodiversity assets such as centers of endemism or biodiversity hotspots (Myers *et al.* 2000; Roberts *et al.* 2002; Orme *et al.* 2005), taxa-specific diversity(Grenyer *et al.* 2006) (Brum *et al.* 2017; Roll *et al.* 2017), ecosystem services (Turner *et al.* 2007), and areas distinguished as unique for biodiversity (Eken *et al.* 2004; BirdLife International 2010; IUCN 2016a). Many other efforts focus on synthesizing and mapping the distribution of threats to these places(Halpern *et al.* 2015; Tulloch *et al.* 2015; Venter *et al.* 2016a). These maps enable organizations to identify places that align with their core principles: hence, this approach can be used to support high-level strategies for coordinating funding from international or multi-lateral organizations to influence where investments are made (Brooks *et al.* 2006; Smith *et al.* 2009).

Organizations adopting a species-based approach to conservation often promote "flagships" to generate financial support and raise awareness for conservation programs**.** Flagship species can be selected using a variety of physical or ecological attributes, their perceived charisma or cultural value , threat status, or because they are considered surrogates for important ecological functions (e.g. under the umbrella, keystone or indicator species concepts) (Caro 2010; Verissimo *et al.* 2011; Ripple *et al.* 2016; Macdonald *et al.* 2017) . The species-based approach capitalizes on people's affinity for wildlife and is a popular subject for ongoing research in conservation marketing theory (Macdonald *et al.* 2015; Veríssimo *et al.* 2017; Veríssimo *et al.* 2018), which includes efforts to quantify the physical attributes (e.g. body size or eye position) that make a flagship species successful (Smith *et al.* 2012; Macdonald *et al.* 2017). Species-based efforts empower organizations to consolidate their messaging, reach broader public audiences, and see measurable improvements in fundraising for conservation projects

While both approaches have their individual merits, their limitations are not fundamentally dissimilar from each other. For example, identifying a place or species based solely on its ecological or physical attributes is not equivalent to its being identified as a conservation priority. This is because the selection is not bound to a constructed mathematical problem (e.g. maximizing benefits or minimizing threats) or systematic process, nor is it necessarily linked to the costs and feasibility of actions that can be undertaken at the local context to abate threatening processes (Game *et al.* 2013; Brown *et al.* 2015). Further, both approaches tend to bias funding efforts towards a narrow subset of places (e.g. biodiversity hotspots)(Myers *et al.* 2000) or species (e.g. mamamalian megafauna)(Joseph *et al.* 2011; Ripple *et al.* 2016), often ignoring the fundamental concept of complementarity central to systematic conservation planning methods (Segan et al. 20011). Complementarity ensures that priorities represent all biodiversity not just those which co-occur with the desirable attributes of places or species (Di Minin & Moilanen 2013).

Despite these similar limitations, species based approaches using flagships have arguably received more rigorous evaluation and criticism of their ability to deliver effective and efficient conservation (Simberloff 1998; Andelman & Fagan 2000; Williams *et al.* 2000). Thus, the ability of flagshsip species to increase the flow of capital for environmental organizations, married with the ecological criticisms of the approach can lead to a conservation conundrum (Joseph *et al.* 2011). Here we ask: to what degree do flagship species constrain the delivery of conservation objectives? Surprisingly, there has been no rigorous quantitative assessment of the ability of flagships to deliver on conservation objectives relative to place-based strategies. An integrated approach that capitalizes on their synergies is likely to be more desirable, efficient and effective than either approach alone (Likens & Lindenmayer 2012; Di Minin & Moilanen 2013; Bennett *et al.* 2015), yet no such strategy has been developed to date.

Here we present the first approach to conservation prioritization that integrates place-based and species-based strategies. This enables effective marketing and fundraising by conservation organizations worldwide, while simultaneously maximizing a biodiversity benefit, defined as the number of non-flagship species protected, hereafter called "background" species that are protected. Our approach is novel and user-inspired (Wall *et al.* 2017), and demands that site selection also be guided by the presence of one or more unique candidate flagships. We ensure that traditional planning objectives underpinning current global policy efforts can be achieved, for example, complementarity (e.g. representing diverse ecosystems and species; see Figure S1) are achievable. To test our approach, we developed eight global planning scenarios based on different combinations of attributes (see below for more details) for both flagship species and places. We then compare our results to a) that of a "greedy" approach where the sole purpose is to efficiently deliver the maximum conservation benefit in the fewest number of sites, and b) a random site selection (run 100 times for each scenario).

Potential flagship candidates were initially selected from two global lists: one of existing flagship mammals supplemented by those species previously identified as possessing desirable flagship attributes (Smith *et al.* 2012), and one containing reptiles and birds derived from methods using species Wikipedia page views as a proxy for public interest (Mittermeier *et al. (in prep)*; Roll *et al.* 2016). These candidates were further refined to include only species whose ranges have been mapped and made available by the International Union for the Conservation of Nature (IUCN). The resulting 540 species were then classified according to their IUCN Red-List status (www.iucnredlist.org). We considered taxa classified as "near-threatened" or higher to be in need of conservation action and treated this as a second species attribute class when developing scenarios.

For place-based attributes, organizations invariably have different perspectives on what constraints, or enabling factors, are most relevant to their planning objectives. These attributes would likely reflect different aspects of the biodiversity present (e.g. richness of endemism or phylogenetic distinctiveness), but could also consist of socio-economic or political factors related to the feasibility of conservation interventions (Brown *et al.* 2015), reputational risk for the organization, or any other preferences that may rule a site in or out of consideration. We assigned three plausible place-based attributes based on a site's: presence in a global priority terrestrial or freshwater ecoregion (Olson David & Dinerstein 2008); the fraction of its area already protected (e.g. >=10% and <=90% covered by any category of Protected Area (WDPA 2017); and its mean Human Footprint Index value, where a value < 4 was threshold (Figure S2) (Venter *et al.* 2016b; Allan *et al.* 2017). From the combination of these three attributes and the two species-based attributes (all candidates and just threatened candidates), we derived eight different planning scenarios to test our integrated method (Figure 1, Table S1).

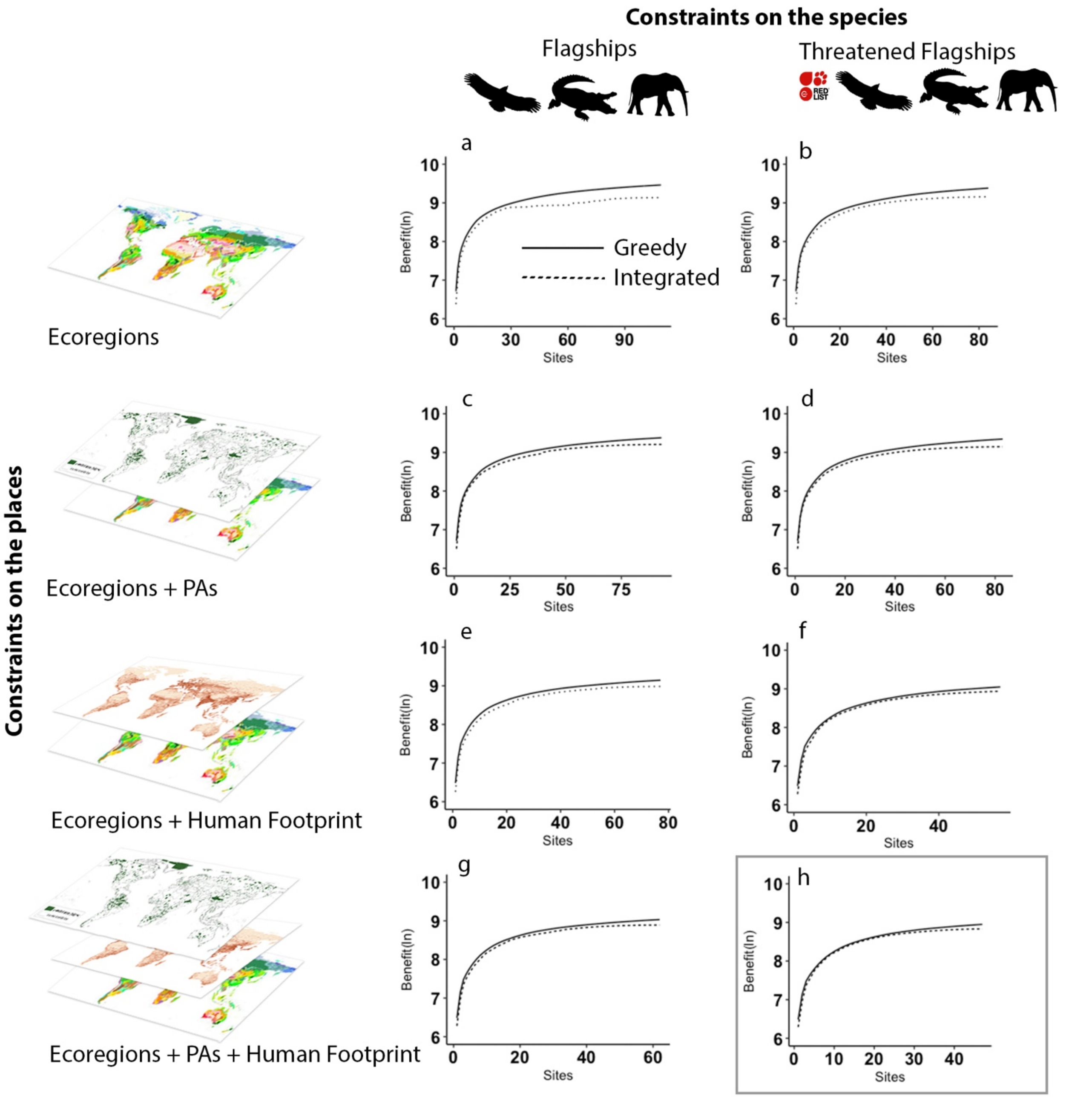

Figure A1: Scenario performance of the greedy versus integrated approach in terms of the efficient accumulation of the conservation benefit (number of background species).

To measure performance efficiency, we compared the total benefit delivered (based on the number of background species available) from the sites prioritized in our integrated approach with that of the equivalent number of sites identified from the greedy and random approaches. We found that our integrated approach retained 80-89% of the conservation benefit delivered by the purely greedy approach, whilst identifying both places and flagship species (Figure 1; Table A1). For example, when considering threatened flagships and all place-based attributes (Figure A1h), the integrated approach delivered 6,849 background species in 47 sites (Table A1). The greedy approach delivered 7,702 species for the same number of sites. Therefore, we consider the efficiency retained with our integrated approach to be 89%. Since resources are limited, we might expect organizations to first prioritize places that contribute most to the overall benefit, thus the potential realized efficiency is likely to be higher. For example, we see efficiency of 87-97% when only considering the 10 most beneficial sites (Table A1, S3). In all cases, the integrated approach performed far better than random site selection (where mean benefit ranged from 30-61%; Table 1; Figure S4), and this efficiency improved when the planning region was more constrained.

Table A1. Results from the greedy, integrated and random approaches describing the maximum benefit available for each scenario (a-h) as described in Figure A1, S1; and the proportion of the benefit retained.

	Number of sites in solution for:			Number of species delivered with:		Efficiency retained:		
	greedy approach	integrated approach	Max benefit (number of background species)	greedy approach for equal number of sites	integrated approach	flagships	top 10 sites	Null Model (mean 100 runs)
a	1,473	107	19,616	12,878	10,545	**82%**	87%	30%
b	1,473	84	19,616	11,961	9,487	**80%**	90%	30%
c	855	93	16,542	11,835	9,965	**84%**	92%	36%
d	855	83	16,542	11,443	9,387	**82%**	92%	36%
e	554	77	12,053	9,362	7,972	**85%**	89%	43%
f	554	58	12,053	8,557	7,621	**89%**	96%	43%
g	287	62	9,833	8,363	7,269	**87%**	93%	61%
h	287	47	9,833	7,702	6,849	**89%**	97%	61%

The scenario described above (Figure 1 Scenario h) was the most constrained and efficient, therefore we discuss these results in more detail. The 47 sites identified using this approach collectively contain 176 candidate flagship species consisting of 111 mammals, 53 birds, and 12 reptiles (Table S2). The number of flagships in a site ranged from 1 to 20. For example, our method shows that a site in China (Figure A2 Site I), primarily covered by Hengduan Shan Conifer Forests (but also flanked by the Tibetan Plateau Steppe and Southwest Temperate Forests ecoregions), is home to the Giant Panda (*Ailuropoda melanoleuca*). In addition to this prominent flagship, other candidate flagship species such as Takin (*Budorcas taxicolor*), Golden snub-nosed monkey (*Rhinopithecus roxellana*), Snow Leopard (*Panthera uncia*), and the Chinese softshell turtle (*Pelodiscus sinensis*) also occur in this location (Figure S5).

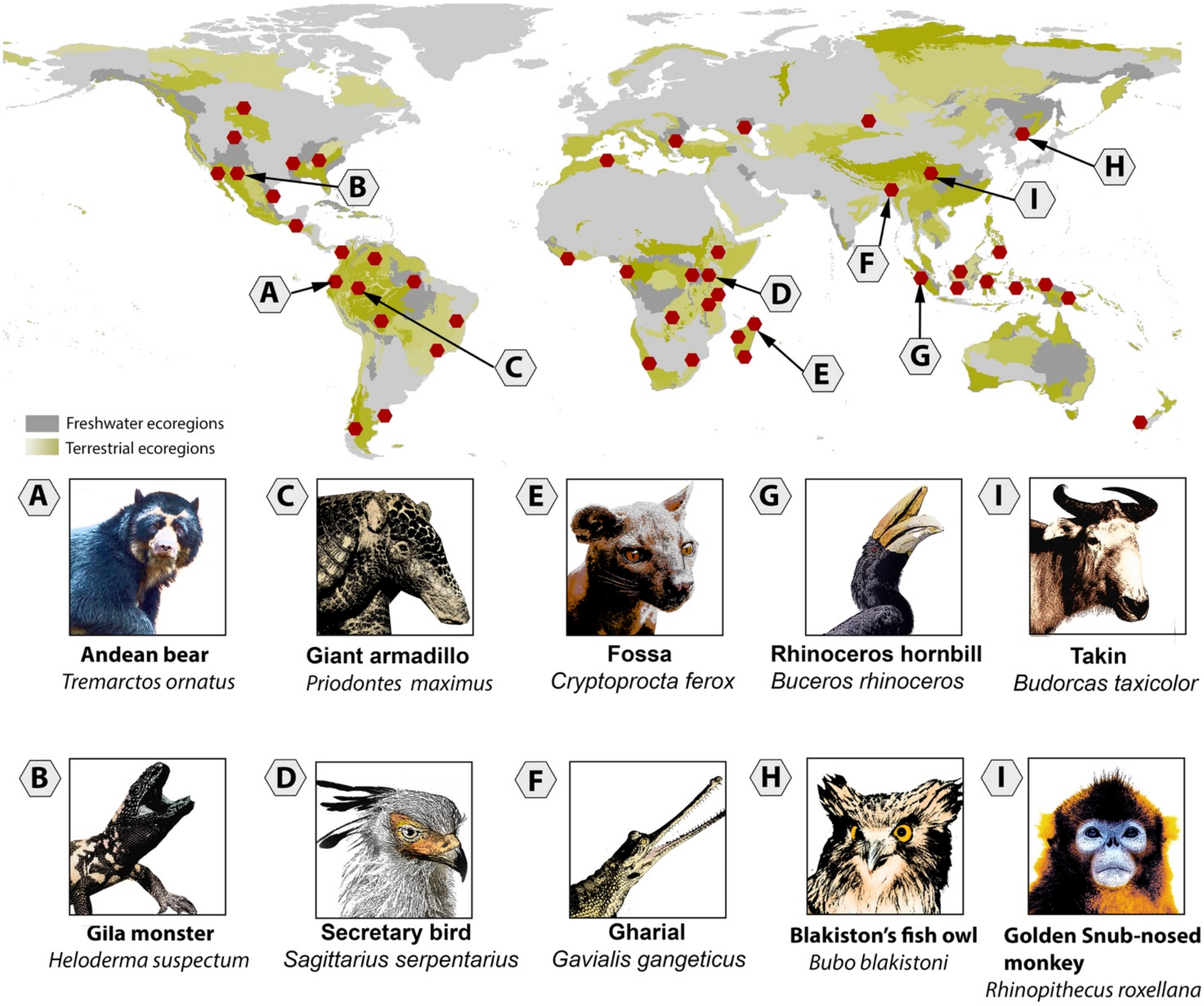

Figure A2. An example of the results produced from the integrated approach delivering both sites and potential flagship species. The map shows the 47 sites and a small sample of the species delivered from the most constrained scenario (Fig A1h). See Table S2 for full list of sites and species. Species illustrations rendered by John Armanini.

Importantly, our intention is not to advocate for a suite of places or flagship species for global conservation. The candidate places and flagships we used were identified through quantitative analyses and provide a useful and plausible collection with which to test the efficiency of species-based conservation. The selection of flagship(s) by an organization should be carefully placed in accordance with their conservation objectives, their target audience and marketing strategies, but also within the local contexts where these species coexist with communities (Leo & Diogo 2013). Similar precautions apply to the attributes guiding candidate places, as what may drive an agenda for an international NGO may not align with the local governments and NGOs responsible for implementing conservation on the ground (Smith *et al.* 2009).

Our research demonstrates that conservation organisations can optimize conservation benefits while retaining the advantages of flagship species for fundraising. Further, the external benefits provided by many flagship campaigns, such as awareness raising for conservation issues, likely outweigh their inefficiencies (Bennett *et al.* 2015). Importantly, our integrated approach provides a flexible and rigorous mechanism to prioritise both places and species to guide future conservation investments. This approach should be considered complementary to the more specific systematic conservation planning approaches that can further guide the most appropriate placement and timing of management actions at finer scales.

A2. METHODS

Selecting candidate species

We used two approaches to identify charismatic candidate species. For mammals, we used the previous conservation flagships (N = 80) as well as the "Cinderella" species (N = 183) identified by (Smith *et al.* 2012). Cinderella species have similar physical characteristics to flagships, namely large body size and forward-facing eyes, but presently do not serve as flagships (Smith *et al.* 2012). For reptiles and birds, we identified candidate species using an approach developed by Roll et al. (Roll *et al.* 2016) that quantifies interest in species based on their online popularity, measured via the number of Wikipedia page views. Popular reptile species were taken from Roll et al (Roll *et al.* 2016), and bird species were identified by matching the global species taxonomy of the International Ornithological Committee (IOC World Bird List version 7.1) to English language Wikipedia pages and extracting views to each species page for the period between 1 January 2016—1 February 2017 (Mittermeier et al., *in prep*.). The top 100 reptiles and 500 birds, measured by total page views, were identified as the flagship representatives of these groups. However, as the range maps were not available for all of the candidate species, our final list of candidates was reduced to 540 species.

Spatial constraints

We created a global terrestrial equal area planning grid covering the WWF Global 200 (Olson David & Dinerstein 2008), a set of 238 terrestrial and freshwater ecoregions of exceptional biodiversity (10,200 sites of 100x100 km). Each site was then assigned to a unique ecoregion. In instances where more than one ecoregion fell within a site, assignment went to the ecoregion with larger proportional coverage. In areas where terrestrial ecoregions overlapped with larger freshwater ecoregions, we assigned cells to the finer resolution terrestrial ecoregions. For each site, we assigned attributes based on the criteria we outlined for the scenarios (Figure 1). We identified the proportion of each cell allocated to any category of IUCN Protected Areas based on the World Database of Protected Areas (WDPA 2017). We removed PAs whose status was identified as 'Proposed', but retain those listed as 'Not Reported'. We used a threshold of ≥ 10% and ≤90%, as cells with few areas protected (≤ 10%) might be difficult to pioneer protection for, while cells with considerable coverage (≥ 90%) might be too spatially constrained for the objectives of the problem we aimed to solve. We recognize that this is an arbitrary threshold, reflective of perceived "feasibility", thus we included additional scenarios where this criterion does not influence the prioritization. Considering PAs constrained the searchable area by 70% (to 3,097 sites). For each site we took the mean value of the Human Footprint Index (Venter *et al.* 2016b) and set a threshold of < 4 for which the site can be considered as not human dominated (Allan *et al.* 2017). Considering the human footprint reduced the searchable area by 61% (to 3,961 sites). Consideration of both constraints together reduced the area by 90% (to 1,068 sites).

Species constraints

Candidate species: We used the IUCN species range maps for our candidate mammals and reptile species (IUCN 2016b) and the Handbook of the Birds of the World for candidate birds (World 2016). We considered species with an IUCN status of Near-threatened or higher to be in need of conservation action, reducing the list of candidate flagships by 37% (from 540 to 338 species).

Background species: In addition to mammals, birds and reptiles, background species comprised all freshwater crustaceans and carnivorous insects, and amphibians for which IUCN polygons exist (i.e. N = 19,616). For all species ranges, we followed the treatment of Butchart et. al (Butchart *et al.* 2015) and retained those parts of their distributions marked as native or re-introduced, or with presence coded as extant, possibly extant, or possibly extinct. We created a presence-absence matrix for both the candidate and background species based on the intersections of these ranges with the planning areas utilized in the different scenarios. Given that many species occupy ranges much smaller than our planning unit size, we erred on the side of caution and did not assign a minimum size threshold to reflect species' presence (Visconti *et al.* 2013) All geoprocessing of spatial data was completed using PostGIS2.3 and ArcGIS v10.3 (ESRI, Redlands).

Scenarios

Based on different combinations of the species- and place-based attributes, we developed eight integrated global planning scenarios (referred to as ‘a’-‘h’ (Table 1). Each scenario was subjected to the customized integrated approach (Figure S1, notated code is also provided in Appendix B of the Supplementary materials), the greedy approach, and 100 iterations of the random approach. All programming was developed in R programming language (R Core Team 2013).

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Supplementary material for Appendix A. Flagship species can deliver efficient conservation

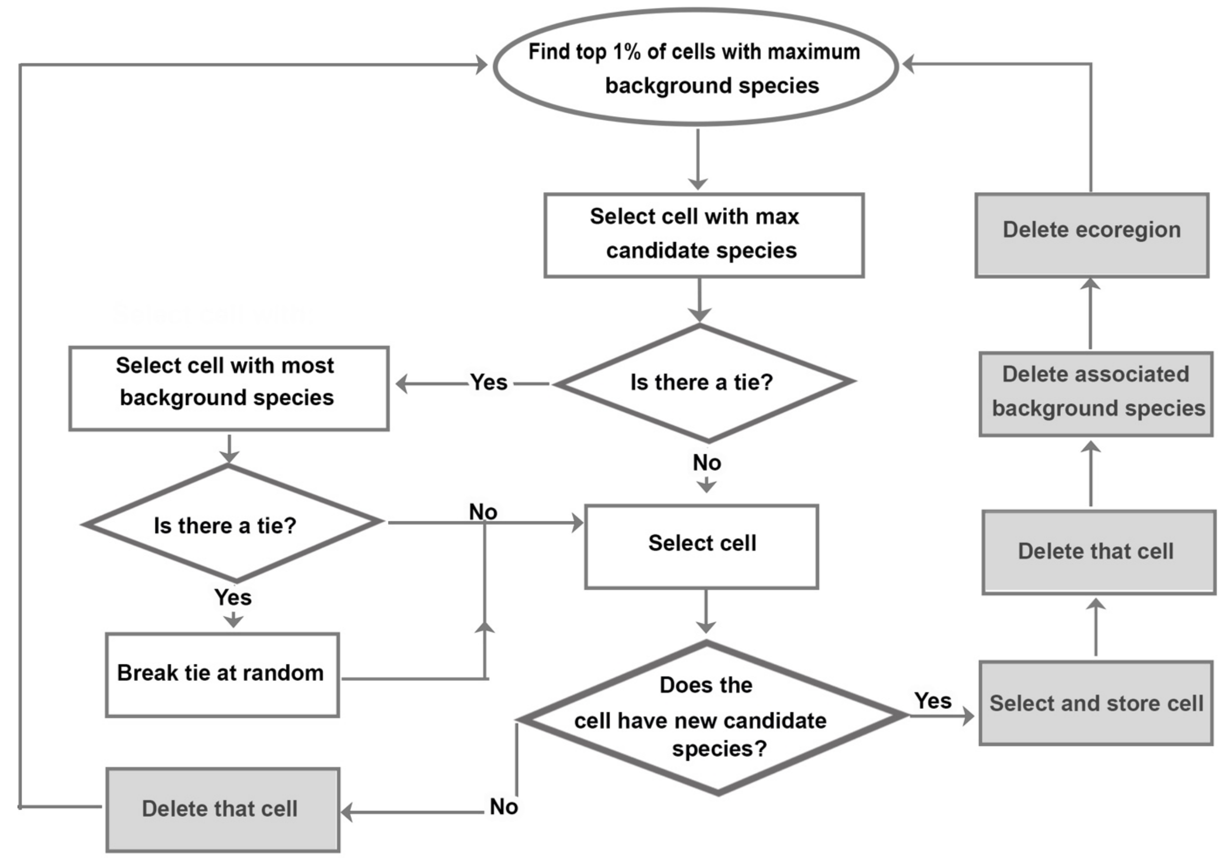
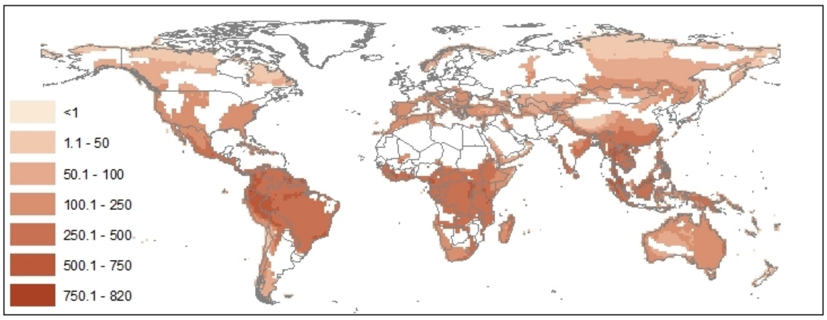


Figure S1. Algorithmic decision tree for the integrated approach which begins by finding the sites with the maximum number of background species which also have the most candidate species in them. Given a tie, we preference the sites with the most background species. Once a site is selected, both the ecoregion and background species are deleted to ensure complementarity is achieved (cells in grey).

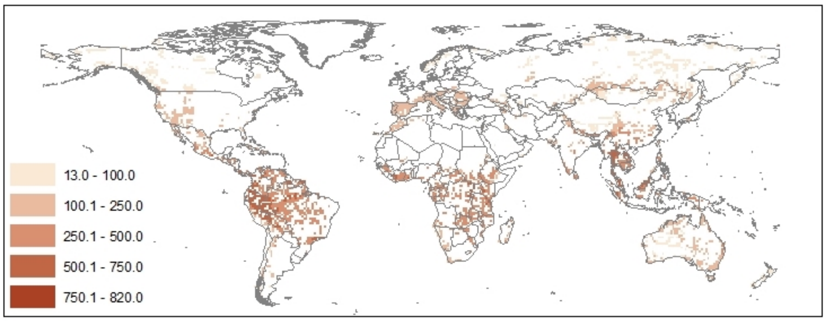
Table S1. Scenario matrix showing how different combinations of species- and place-based attributes create the eight planning scenarios.

		Candidate flagship attributes	
place-based attributes		All flagship species	Threatened flagship species only (Near-threatened or higher)
	All sites found in WWF ecoregions	a	b
	Ecoregions +Protected Areas	c	d
	Ecoregions + Human Footprint	e	f
	Ecoregions +Protected Areas + Human Footprint	g	**h**

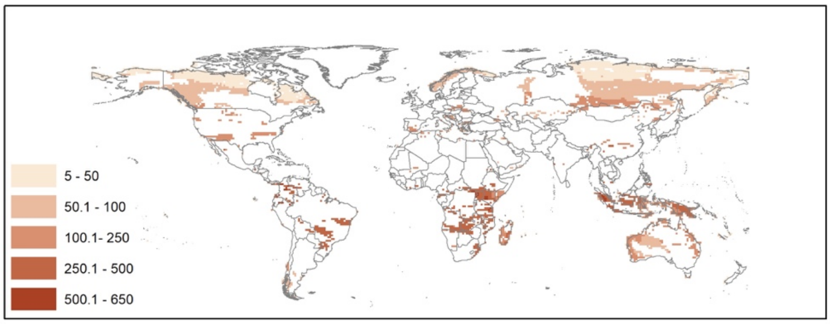
Ecoregions



Ecoregions + Protected Areas (PAs)



Ecoregions + Human Footprint (HF)



Ecoregions + PAs + HF)

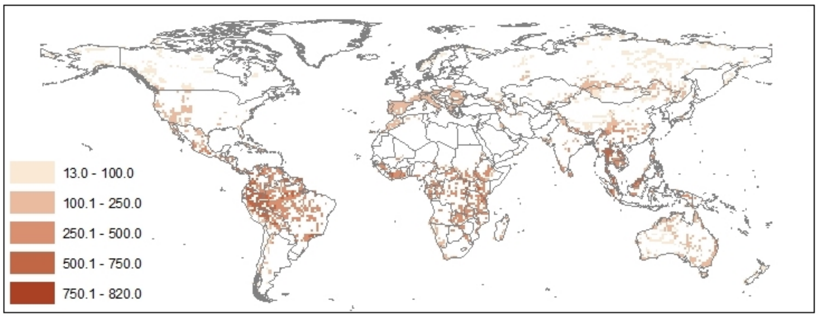


Figure S2. Distributions of richness for the conservation benefit (the number of background species) that are available for the different place-based constraints.

Source code for the integrate model

```
## Code for Integrated Analysis of McGowan et.al (2018)
## Code written by A. Chauvenet and supplemented by J. McGowan
```

The following code assumes you have two matrices organised by the binary presence of species (rows) in each unique site (columns, or" PUID" in code). For our analysis, these matrices are "Back_Tot" and "Cand_Tot" which relates to the Background species (max 19, 616 species) and Candidate flagship matix (max 540 species), respectively.

Bring in PU_layer with attributes that are desirable for breaking ties, this can be anything (e.g. the number of endmecis species found there). You can use Figure A1 for guidance. Read in the matrices.

```
##Start here
PU<-read.csv('PU_Layer.csv',header=T)
Candtot_m<-read.csv("candidatematrix.csv",row=1)
Backtot_m<-read.csv("backgroundmatrix.csv",row=1)

master_back<-Backtot_m #makes duplicate backup
master_cand<-Candtot_m #makes duplicate backup

#### Below is the integrated algorithm from Figure A1.

SAVED <- c() ## name of the Cand species selected
SAVED_count <- c()
SAVED2 <-
  c() ##Number of unique new candidates species selected in the Max step (including 0 if no new
species)
PUIDs <- c() ## the puid of the Max
BACK <- c() ## the total number of Background save in each puid
ECO_rem_rec <-
  c() ## the Number of puids removed with each ecoregions

Backtot_m <- master_back # reloads to start fresh each time
Candtot_m <- master_cand

count <- 0 # couting the steps and selection of PUIDS
while (dim(Backtot_m)[1] > 0 &&
       dim(Backtot_m)[2] > 0)
  # condition to keep loop running until nothing left
{
  count <- count + 1

  #1/ find PUID which maximises the number of background species (and candidate species if there
is a tie)
  max_sum <-
    apply(Backtot_m, 2, sum) #sums over each column in the matrix
  len1 <-ifelse(length(max_sum)<100,1,round(1 * length(max_sum) / 100,0))
     #grabs the top 1 percent with max species
  max_N <- sort(max_sum, decreasing = TRUE)[1:len1]
  max_pos <-
```

```
    order(max_sum, decreasing = T)
# position in Backtot_m that has the largest # of background species
  temp <- c()
  for (j in 1:len1) {
    pos2 <-
      which(colnames(Candtot_m) == names(max_sum[max_pos])[j])
##finds the correspsonding Number of candidates in the top 1% that is selected
    temp <- c(temp, sum(Candtot_m[, pos2]))
  }
  #breaking ties between candidate sites
  temp_max_pos <- which(temp == max(temp))
  temp_max<-temp[temp_max_pos]
  if (length(temp_max) > 1)
    temp_max_pos <-
    temp_max_pos[which(max_N[temp_max_pos] == max(max_N[temp_max_pos]))]
# if tie, pick the site with the max background species
 # above line is where we might change code to break ties with different attribute, or #species
combos in this section
 (max_N[temp_max]==max(max_N[temp_max]
  if (length(temp_max_pos) > 1)
    temp_max_pos <-
    sample(temp_max_pos, 1)

# if the max background tie is another tie, break tie at random

  back_puid_max <- max_N[temp_max_pos] #collect the winning site
  max_pos <-
    which(colnames(Backtot_m) == names(max_N)[temp_max_pos]) # find the position
  max_puid <-
    colnames(Backtot_m)[max_pos] #find the unique id of winning site

 ##The following section deals with complementarity by removing the selected site, #removing its
ecoregion, and removing all associated species from the background species #matrix and storing the
candidates that have been selected

# storing candidates
  cand_pos <- which(colnames(Candtot_m) == max_puid)
  saved_pos <- which(Candtot_m[, cand_pos] == 1)
  SAVED <-
    c(SAVED, rownames(Candtot_m)[saved_pos])# storing the candidates from the winning site
  diff <-
    length(SAVED) - length(unique(SAVED))
# if duplicates, find them and only store unique candidates in this list
  SAVED <- unique(SAVED)
  SAVED2 <- c(SAVED2, length(saved_pos) - diff)
  SAVED_count <-
    c(SAVED_count, rep(count, (length(saved_pos) - diff)))
  print(c("count", count))
  print(c("unique number of candidate saved", sum(SAVED2)))
  PUIDs <- c(PUIDs, max_puid)

  if (SAVED2[length(SAVED2)] > 0) {
```

```
    # removing the site puid after selection from Background matrix
    print(c("dim backtot 0", dim(Backtot_m)))
    BACK <- c(BACK, sum(Backtot_m[, max_pos]))
    saved_back <- which(Backtot_m[, max_pos] == 1)
    Backtot_m <- Backtot_m[, -max_pos]
    print(c("dim backtot 1 (rem col)", dim(Backtot_m)))
    if (length(saved_back) > 0)
      Backtot_m <- Backtot_m[-saved_back, ]
    print(c("dim backtot 2 (rem rows)", dim(Backtot_m)))
    ECO_rem_rec <- c(ECO_rem_rec, length(max_pos))

    # removing the associated puids with the ecoregion that has been selected
    ECO_rem <- PU$G200_ID[which(PU$puid == PUIDs[length(PUIDs)])]
    puid_rem <-
      PU$puid[which(PU$G200_ID == ECO_rem)] # all the piuds belonging to that ecoregion
    temp_puid <- c()
    for (k in 1:length(puid_rem)) {
      temp_puid <- c(temp_puid, which(colnames(Backtot_m) == puid_rem[k]))
    }
    ECO_rem_rec <- c(ECO_rem_rec, length(temp_puid))
    if (length(temp_puid) > 0)
      Backtot_m <- Backtot_m[, -temp_puid]
    print(c("dim backtot 3 (rem col)", dim(Backtot_m)))
  } else{
    Backtot_m <- Backtot_m[, -max_pos]
    BACK <- c(BACK, 0)
    ECO_rem_rec <- c(ECO_rem_rec, 0)
  }

}

#Below helps link up stored lists
cbind(SAVED,SAVED_count)

Puids<-as.data.frame(PUIDs)
Back<-as.data.frame(BACK)
Pus<-cbind(Puids,Back)
Pus$search_no<-seq.int(nrow(Pus))
Cands<-cbind(SAVED,SAVED_count)
colnames(Cands)[2]<-"search_no"

Base_result<-merge(Cands,Pus, by="search_no") #final output
write.csv(Base_result,"Base_results_TH.csv",row.names=T)
```

Null model methods and results

We generated null models based on the four scenarios which do not consider flagship species or ecoregional complementarity. We apply the random selection of sites to scenarios a, c, e, and g (Table S1), and repeated this 100 times each to evaluate the mean benefit achieved under random selection (reported in main Table A1).

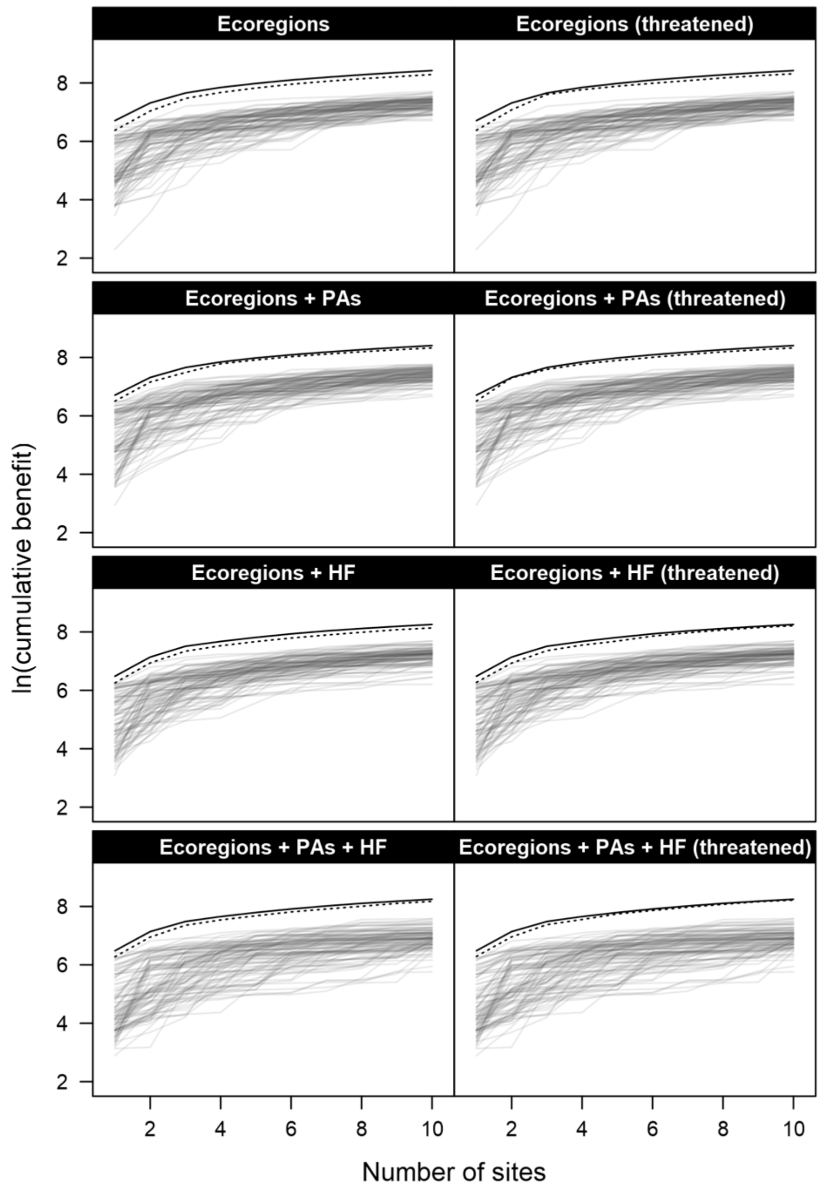


Figure S3. The performance of the greedy (solid), integrated (dashed) and random (grey) approaches for the top 10 sites delivering the greatest number of background species.

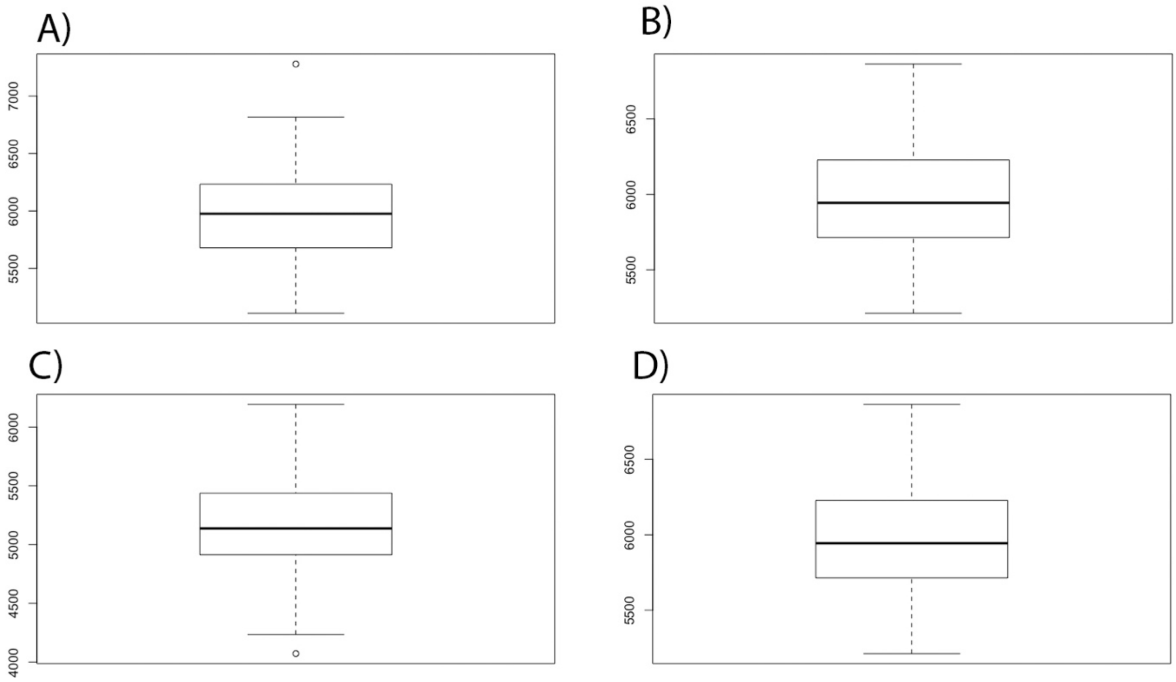


Figure S4. The benefit delivered (out of 19,616 background species) from 100 random selections of sites for panel A) Scenario a (mean = 5,973 species); panel B) Scenario c (mean =5,968 species); panel C) Scenario e (mean = 5,136 species); and Panel D) Scenario g (mean = 5,968 species).

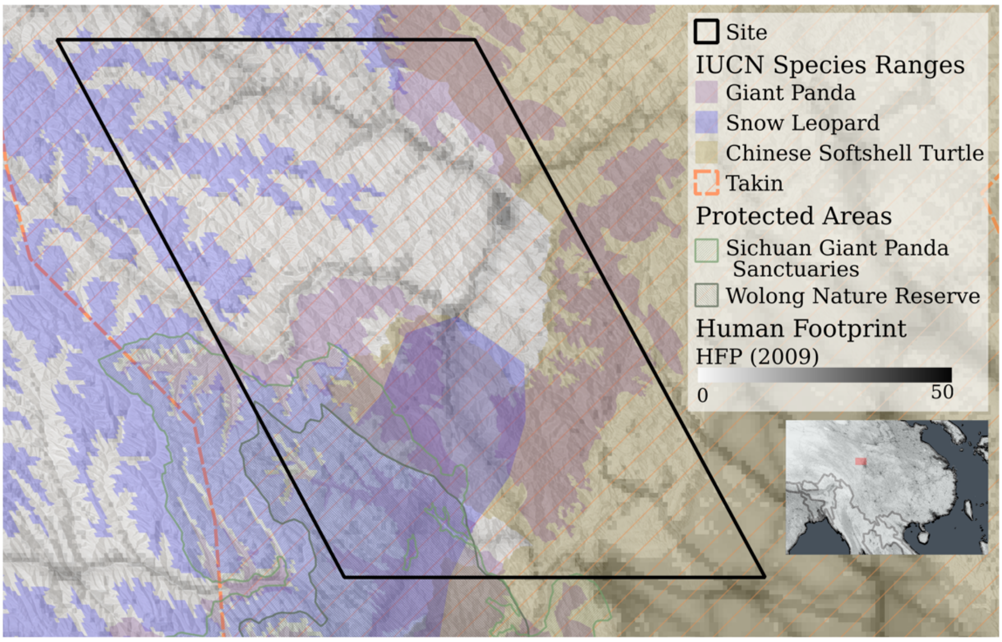


Figure S5. Example of the distribution of candidate flagship species, protected areas, human footprint within a site I (Figure A1 main text).

Supplemental Table S2. Data of 176 candidate flagship species consisting of 111 mammals, 53 birds, and 12 reptiles from 47 sites identified using our approach integrating prioritization which delivers both sites and potential flagship species (data from IUCN 2017). For details on how IUCN status and current population trend are assigned see IUCN 2012.

Site	Class	Order	Family	Species	Common name	IUCN status	Current Population Trend	Countries of occurrence
1	Aves	Bucerotiformes	Bucerotidae	*Buceros bicornis*	Great Hornbill	Near Threatened	Decreasing	Bhutan; Cambodia; China; India; Indonesia; Lao People's Democratic Republic; Malaysia; Myanmar; Nepal; Thailand; Viet Nam; Bangladesh
1	Aves	Bucerotiformes	Bucerotidae	*Buceros rhinoceros*	Rhonocerous hornbill	Near Threatened	Decreasing	Brunei Darussalam; Indonesia; Malaysia; Thailand
1	Mammalia	Carnivora	Canidae	*Cuon alpinus*	Dhole	Endangered	Decreasing	Bangladesh; Bhutan; Cambodia; China; India; Indonesia; Lao People's Democratic Republic; Malaysia; Myanmar; Nepal; Thailand
1	Mammalia	Carnivora	Felidae	*Neofelis diardi*	Sunda coulded leopard	Vulnerable	Decreasing	Brunei Darussalam; Indonesia (Kalimantan, Sumatera); Malaysia (Sabah, Sarawak)
1	Mammalia	Carnivora	Felidae	*Panthera tigris*	Tiger	Endangered	Decreasing	Bangladesh; Bhutan; China (Anhui - Regionally Extinct, Beijing - Regionally Extinct, Chongqing - Regionally Extinct, Fujian - Possibly Extinct, Guangdong - Possibly Extinct, Guangxi - Regionally Extinct, Guizhou - Regionally Extinct, Hebei - Regionally Extinct, Heilongjiang, Henan - Regionally Extinct, Hubei - Regionally Extinct, Hunan - Possibly Extinct, Jiangsu - Regionally Extinct, Jiangxi - Possibly Extinct, Jilin, Liaoning - Regionally Extinct, Shaanxi - Possibly Extinct, Shandong - Regionally Extinct, Shanghai - Regionally Extinct, Shanxi - Regionally Extinct, Sichuan - Regionally Extinct, Tianjin - Regionally Extinct, Tibet [or Xizang], Xinjiang - Regionally Extinct, Yunnan, Zhejiang - Possibly Extinct); India; Indonesia (Bali - Regionally Extinct, Jawa - Regionally Extinct, Sumatera); Lao People's Democratic

								Republic; Malaysia (Peninsular Malaysia); Myanmar; Nepal; Russian Federation; Thailand
1	Mammalia	Carnivora	Felidae	*Pardofelis marmorata*	Marbled Cat	Near Threatened	Decreasing	Bangladesh; Bhutan; Brunei Darussalam; Cambodia; China; India; Indonesia (Kalimantan, Sumatera); Lao People's Democratic Republic; Malaysia (Peninsular Malaysia, Sabah, Sarawak); Myanmar; Nepal; Thailand; Viet Nam
1	Mammalia	Carnivora	Felidae	*Prionailurus planiceps*	Flat-headed Cat	Endangered	Decreasing	Brunei Darussalam; Indonesia (Kalimantan, Sumatera); Malaysia (Peninsular Malaysia, Sabah, Sarawak)
1	Mammalia	Carnivora	Ursidae	*Helarctos malayanus*	Malayan Sun Bear, Sun Bear	Vulnerable	Decreasing	Bangladesh; Brunei Darussalam; Cambodia; India; Indonesia; Lao People's Democratic Republic; Malaysia; Myanmar; Thailand; Viet Nam
1	Mammalia	Carnivora	Viverridae	*Arctictis binturong*	Binturong, Bearcat, Palawan Binturong	Vulnerable	Decreasing	Bangladesh; Bhutan; Cambodia; China; India; Indonesia (Jawa, Kalimantan, Sumatera); Lao People's Democratic Republic; Malaysia (Peninsular Malaysia, Sabah, Sarawak); Myanmar; Nepal; Philippines; Thailand; Viet Nam
1	Mammalia	Carnivora	Viverridae	*Cynogale bennettii*	Otter Civet, Otter-civet, Sunda Otter Civet	Endangered	Decreasing	Brunei Darussalam; Indonesia (Kalimantan, Sumatera); Malaysia (Peninsular Malaysia, Sabah, Sarawak)
1	Mammalia	Cetartiodactyla	Suidae	*Sus barbatus*	Bearded Pig, Western Bearded Pig	Vulnerable	Decreasing	Brunei Darussalam; Indonesia (Kalimantan, Sumatera); Malaysia (Peninsular Malaysia, Sabah, Sarawak)
1	Mammalia	Perissodactyla	Tapiridae	*Tapirus indicus*	Asian Tapir, Indian Tapir, Malay Tapir, Malayan Tapir	Endangered	Decreasing	ndonesia (Sumatera); Malaysia; Myanmar; Thailand
1	Mammalia	Primates	Cercopithecidae	*Macaca nemestrina*	Southern pig-tailed macaque	Vulnerable	Decreasing	Brunei Darussalam; Indonesia (Kalimantan); Malaysia (Peninsular Malaysia, Sabah); Thailand; Singapore
1	Mammalia	Primates	Cercopithecidae	*Presbytis melalophos*	Sumatran surili	Endangered	Decreasing	Indonesia (Sumatera)
1	Mammalia	Primates	Hylobatidae	*Hylobates agilis*	Agile gibbon	Endangered	Decreasing	indonesia (Kalimantan, Sumatera); Malaysia (Peninsular Malaysia); Thailand
1	Mammalia	Primates	Hylobatidae	*Symphalangus syndactylus*	Siamang	Endangered	Decreasing	Indonesia (Sumatera); Malaysia (Peninsular Malaysia); Thailand

1	Mammalia	Proboscidea	Elephantidae	*Elephas maximus*	Asian Elephant, Indian Elephant	Endangered	Decreasing	Bangladesh; Bhutan; Cambodia; China; India; Indonesia (Kalimantan, Sumatera); Lao People's Democratic Republic; Malaysia (Peninsular Malaysia, Sabah); Myanmar; Nepal; Sri Lanka; Thailand; Viet Nam
1	Reptilia	Squamata	Elapidae	*Ophiophagus hannah*	King cobra	Vulnerable	Decreasing	Bangladesh; Bhutan; Brunei Darussalam; Cambodia; China; Hong Kong; India (Andaman Is., Andhra Pradesh, Arunachal Pradesh, Assam, Bihar, Goa, Jharkand, Karnataka, Kerala, Manipur, Meghalaya, Mizoram, Nagaland, Orissa, Sikkim, Tamil Nadu, Tripura, Uttaranchal, Uttar Pradesh, West Bengal); Indonesia (Bali, Jawa, Kalimantan, Sulawesi, Sumatera); Lao People's Democratic Republic; Malaysia (Peninsular Malaysia, Sabah, Sarawak); Myanmar; Nepal; Philippines; Singapore; Thailand; Viet Nam
2	Aves	Accipitriformes	Accipitridae	*Aquila heliaca*	Eastern imperial eagle	Vulnerable	Decreasing	Afghanistan; Armenia; Austria; Azerbaijan; Bangladesh; Bhutan; Bosnia and Herzegovina; Bulgaria; Cambodia; China; Croatia; Czech Republic; Djibouti; Egypt; Eritrea; Ethiopia; Georgia; Greece; Hong Kong; Belarus; Cameroon; Cyprus; Denmark; Finland; France; Germany; Italy; Libya; Lithuania; Malaysia; Morocco; Poland; Singapore; Slovenia; Sweden; Togo Hungary; India; Iran, Islamic Republic of; Iraq; Israel; Japan; Jordan; Kazakhstan; Kenya; Korea, Democratic People's Republic of; Korea, Republic of; Kuwait; Kyrgyzstan; Lao People's Democratic Republic; Lebanon; Macao; Macedonia, the former Yugoslav Republic of; Moldova; Mongolia; Montenegro; Myanmar; Nepal; Oman; Pakistan; Palestinian Territory, Occupied; Qatar; Romania; Russian Federation (Central Asian Russia, Eastern Asian Russia, European Russia); Saudi Arabia; Serbia; Slovakia; Sudan; Syrian Arab Republic; Taiwan, Province of China; Tajikistan;

								Tanzania, United Republic of; Thailand; Turkey; Turkmenistan; Ukraine; United Arab Emirates; Uzbekistan; Viet Nam; Yemen
2	Aves	Accipitriformes	Accipitridae	*Gypaetus barbatus*	Bearded vulture	Near Threatened	Decreasing	Afghanistan; Algeria; Andorra; Armenia; Azerbaijan; Bhutan; China; Egypt; Eritrea; Ethiopia; France; Georgia; Greece; India; Iran, Islamic Republic of; Iraq; Israel; Kazakhstan; Kenya; Kyrgyzstan; Lesotho; Mongolia; Morocco; Nepal; Pakistan; Russian Federation (Central Asian Russia, Eastern Asian Russia, European Russia); Saudi Arabia; South Africa; Spain; Sudan; Tajikistan; Tanzania, United Republic of; Turkey; Turkmenistan; Uganda; Uzbekistan; Yemen; Austria; Italy; Switzerland; Bulgaria; Croatia; Cyprus; Czech Republic; Djibouti; Germany; Korea, Democratic People's Republic of; Lebanon; Mauritania; Mozambique; Namibia; Portugal; Romania; Somalia; Zimbabwe
2	Aves	Accipitriformes	Accipitridae	*Gyps africanus*	White backed vulture	Critically Endangered	Decreasing	Angola; Benin; Botswana; Burkina Faso; Burundi; Cameroon; Central African Republic; Chad; Congo, The Democratic Republic of the; Côte d'Ivoire; Eritrea; Ethiopia; Gambia; Ghana; Guinea; Guinea-Bissau; Kenya; Malawi; Mali; Mauritania; Mozambique; Namibia; Niger; Nigeria; Rwanda; Senegal; Sierra Leone; Somalia; South Africa; South Sudan; Sudan; Swaziland; Tanzania, United Republic of; Togo; Uganda; Zambia; Zimbabwe; Lesotho; Liberia
2	Aves	Accipitriformes	Accipitridae	*Neophron percnopterus*	Egyptian Vulture	Endangered	Decreasing	Afghanistan; Albania; Algeria; Andorra; Angola; Armenia; Azerbaijan; Benin; Bulgaria; Burkina Faso; Cameroon; Cape Verde; Central African Republic; Chad; Cyprus; Djibouti; Egypt; Eritrea; Ethiopia; France; Georgia; Ghana; Gibraltar; Greece; Guinea; India; Iran, Islamic Republic of; Iraq; Israel; Italy; Jordan; Kazakhstan; Kenya; Kuwait; Kyrgyzstan; Lebanon; Libya;

								Macedonia, the former Yugoslav Republic of; Mali; Malta; Mauritania; Morocco; Namibia; Nepal; Niger; Nigeria; Oman; Pakistan; Palestinian Territory, Occupied; Portugal; Russian Federation (Central Asian Russia - Vagrant, European Russia); Saudi Arabia; Senegal; Somalia; South Sudan; Spain (Canary Is.); Sudan; Syrian Arab Republic; Tajikistan; Tanzania, United Republic of; Togo; Tunisia; Turkey; Turkmenistan; United Arab Emirates; Uzbekistan; Western Sahara; Yemen; Austria; Bangladesh; Belgium; Botswana; Congo, The Democratic Republic of the; Czech Republic; Denmark; Estonia; Finland; Gambia; Hungary; Mongolia; Mozambique; Myanmar; Norway; Poland; Qatar; Slovakia; Slovenia; Sri Lanka; Svalbard and Jan Mayen; Sweden; Switzerland; United Kingdom; Zimbabwe; China; Côte d'Ivoire; Guinea-Bissau; Uganda
2	Aves	Accipitriformes	Accipitridae	*Polemaetus bellicosus*	Martial eagle	Vulnerable	Decreasing	Angola; Benin; Botswana; Burkina Faso; Burundi; Cameroon; Central African Republic; Chad; Congo, The Democratic Republic of the; Côte d'Ivoire; Eritrea; Ethiopia; Gambia; Ghana; Guinea; Guinea-Bissau; Kenya; Malawi; Mali; Mauritania; Mozambique; Namibia; Niger; Nigeria; Rwanda; Senegal; Sierra Leone; Somalia; South Africa; South Sudan; Sudan; Swaziland; Tanzania, United Republic of; Togo; Uganda; Zambia; Zimbabwe; Liberia
2	Aves	Accipitriformes	Accipitridae	*Terathopius ecaudatus*	Bateleur	Near Threatened	Decreasing	Angola; Benin; Botswana; Burkina Faso; Burundi; Cameroon; Central African Republic; Chad; Congo; Congo, The Democratic Republic of the; Côte d'Ivoire; Djibouti; Egypt; Eritrea; Ethiopia; Gabon; Gambia; Ghana; Guinea; Guinea-Bissau; Kenya; Malawi; Mali; Mauritania; Mozambique; Namibia; Niger; Nigeria; Rwanda; Saudi Arabia; Senegal; Sierra Leone; Somalia; South Africa; South Sudan;

								Sudan; Swaziland; Tanzania, United Republic of; Togo; Uganda; Yemen; Zambia; Zimbabwe; Iraq; Israel; Lesotho; Liberia; Tunisia
2	Aves	Accipitriformes	Sagittariidae	*Sagittarius serpentarius*	Secretarybird	Vulnerable	Decreasing	Angola; Benin; Botswana; Burkina Faso; Burundi; Cameroon; Central African Republic; Chad; Congo, The Democratic Republic of the; Côte d'Ivoire; Djibouti; Eritrea; Ethiopia; Ghana; Kenya; Lesotho; Malawi; Mali; Mauritania; Mozambique; Namibia; Niger; Nigeria; Senegal; Somalia; South Africa; South Sudan; Sudan; Swaziland; Tanzania, United Republic of; Togo; Uganda; Zambia; Zimbabwe; Liberia; Rwanda; Guinea-Bissau
2	Aves	Gruiformes	Gruidae	*Balearica regulorum*	Grey Crowned-crane	Endangered	Decreasing	Angola; Botswana; Burundi; Congo, The Democratic Republic of the; Kenya; Malawi; Mozambique; Namibia; Rwanda; South Africa; Tanzania, United Republic of; Uganda; Zambia; Zimbabwe; Lesotho; Swaziland
2	Aves	Otidiformes	Otididae	*Ardeotis kori*	Kori bustard	Near Threatened	Decreasing	Angola; Botswana; Ethiopia; Kenya; Mozambique; Namibia; Somalia; South Africa; South Sudan; Sudan; Tanzania, United Republic of; Uganda; Zambia; Zimbabwe
2	Aves	Psittaciformes	Psittacidae	*Agapornis fischeri*	Fischer's lovebird	Near Threatened	Decreasing	Tanzania, United Republic of; Burundi; Kenya; Rwanda; Uganda
2	Mammalia	Carnivora	Felidae	*Acinonyx jubatus*	Cheetah, Hunting Leopard	Vulnerable	Decreasing	Algeria; Angola; Benin; Botswana; Burkina Faso; Central African Republic; Chad; Ethiopia; Iran, Islamic Republic of; Kenya; Mali; Mozambique; Namibia; Niger; South Africa; South Sudan; Tanzania, United Republic of; Uganda; Zambia; Zimbabwe; Swaziland
2	Mammalia	Cetartiodactyla	Hippopotamidae	*Hippopotamus amphibius*	Common Hippopotamus, Hippopotamus, Large Hippo	Vulnerable	Stable	Angola; Benin; Botswana; Burkina Faso; Burundi; Cameroon; Central African Republic; Chad; Congo; Congo, The Democratic Republic of the; Côte d'Ivoire; Equatorial Guinea; Ethiopia; Gabon; Gambia; Ghana; Guinea; Guinea-Bissau; Kenya;

								Malawi; Mali; Mozambique; Namibia; Niger; Nigeria; Rwanda; Senegal; Sierra Leone; Somalia; South Africa; South Sudan; Sudan; Swaziland; Tanzania, United Republic of; Togo; Uganda; Zambia; Zimbabwe
2	Mammalia	Perissodactyla	Rhinocerotidae	*Diceros bicornis*	Black Rhinoceros, Hook-lipped Rhinoceros	Critically Endangered	Increasing	Angola; Kenya; Mozambique; Namibia; South Africa; Tanzania, United Republic of; Zimbabwe; Botswana; Malawi; Swaziland; Zambia
2	Mammalia	Proboscidea	Elephantidae	*Loxodonta africana*	African elephant	Vulnerable	Increasing	Angola; Benin; Botswana; Burkina Faso; Cameroon; Central African Republic; Chad; Congo; Congo, The Democratic Republic of the; Côte d'Ivoire; Equatorial Guinea; Eritrea; Ethiopia; Gabon; Ghana; Guinea; Guinea-Bissau; Kenya; Liberia; Malawi; Mali; Mozambique; Namibia; Niger; Nigeria; Rwanda; Senegal; Sierra Leone; Somalia; South Africa; South Sudan; Tanzania, United Republic of; Togo; Uganda; Zambia; Zimbabwe; Swaziland
3	Aves	Accipitriformes	Accipitridae	*Harpia harpyja*	Harpy eagle	Near Threatened	Decreasing	Argentina; Belize; Bolivia, Plurinational States of; Brazil; Colombia; Costa Rica; Ecuador; French Guiana; Guatemala; Guyana; Honduras; Mexico; Nicaragua; Panama; Paraguay; Peru; Suriname; Venezuela, Bolivarian Republic of
3	Aves	Caprimulgiformes	Apodidae	*Chaetura pelagica*	Chimney swift	Near Threatened	Decreasing	Aruba; Bahamas; Belize; Bermuda; Brazil; Canada; Cayman Islands; Chile; Colombia; Costa Rica; Cuba; Dominican Republic; Ecuador; El Salvador; Guatemala; Haiti; Honduras; Mexico; Nicaragua; Panama; Peru; Puerto Rico; Saint Pierre and Miquelon; Turks and Caicos Islands; United States; Venezuela, Bolivarian Republic of; Anguilla; Barbados; Greenland; Jamaica; Portugal; United Kingdom; Virgin Islands, U.S.
3	Mammalia	Carnivora	Felidae	*Leopardus tigrinus*	Oncilla	Vulnerable	Decreasing	Bolivia, Plurinational States of; Brazil; Colombia; Costa Rica; Ecuador; French Guiana; Guyana; Panama; Peru; Suriname; Venezuela, Bolivarian Republic of

3	Mammalia	Carnivora	Mustelidae	*Pteronura brasiliensis*	Giant otter	Endangered	Decreasing	Bolivia, Plurinational States of; Brazil; Colombia; Ecuador; French Guiana; Guyana; Paraguay; Peru; Suriname; Venezuela, Bolivarian Republic of
3	Mammalia	Cingulata	Chlamyphoridae	*Priodontes maximus*	Giant Armadillo	Vulnerable	Decreasing	Argentina (Chaco, Formosa, Salta, Santiago del Estero); Bolivia, Plurinational States of; Brazil (Acre, Amapá, Amazonas, Espírito Santo, Goiás, Maranhão, Mato Grosso, Mato Grosso do Sul, Minas Gerais, Pará, Paraná, Rondônia, Roraima, Tocantins); Colombia (Colombia (mainland)); Ecuador (Ecuador (mainland)); French Guiana; Guyana; Paraguay; Peru; Suriname; Venezuela, Bolivarian Republic of (Venezuela (mainland))
3	Mammalia	Perissodactyla	Tapiridae	*Tapirus terrestris*	Lowland Tapir, Brazilian Tapir, South American Tapir	Vulnerable	Decreasing	Argentina; Bolivia, Plurinational States of; Brazil; Colombia; Ecuador; French Guiana; Guyana; Paraguay; Peru; Suriname; Venezuela, Bolivarian Republic of
3	Mammalia	Primates	Atelidae	*Ateles belzebuth*	White-bellied spider monkey	Endangered	Decreasing	Brazil (Amazonas, Roraima); Colombia (Colombia (mainland)); Ecuador (Ecuador (mainland)); Peru; Venezuela, Bolivarian Republic of
3	Mammalia	Primates	Atelidae	*Ateles chamek*	Peruvian spider-monkey	Endangered	Decreasing	Bolivia, Plurinational States of; Brazil (Acre, Amazonas, Mato Grosso, Rondônia); Peru
3	Mammalia	Primates	Atelidae	*Lagothrix lagotricha*	Brown woolly monkey	Vulnerable	Decreasing	Brazil (Amazonas); Colombia (Colombia (mainland)); Ecuador (Ecuador (mainland)); Peru
3	Mammalia	Primates	Atelidae	*Lagothrix poeppigii*	Silvery woolly monkey	Vulnerable	Decreasing	Brazil (Acre, Amazonas); Ecuador (Ecuador (mainland)); Peru
3	Mammalia	Primates	Pitheciidae	*Cacajao calvus*	Bald uakari	Vulnerable	Decreasing	Brazil; Peru
3	Reptilia	Crocodylia	Alligatoridae	*Melanosuchus niger*	Black caiman	Lower Risk/conservation dependent	Need updating	Bolivia, Plurinational States of; Brazil; Colombia; Ecuador; French Guiana; Guyana; Peru
4	Aves	Accipitriformes	Accipitridae	*Stephanoaetus coronatus*	Crowned eagle	Near Threatened	Decreasing	Angola; Burundi; Cameroon; Central African Republic; Congo; Congo, The Democratic Republic of the; Côte d'Ivoire; Equatorial Guinea; Ethiopia; Gabon; Ghana; Guinea; Guinea-Bissau; Kenya; Liberia; Malawi; Mozambique; Nigeria; Rwanda; Senegal; Sierra Leone; South Africa; South Sudan;

								Sudan; Swaziland; Tanzania, United Republic of; Togo; Uganda; Zambia; Zimbabwe; Benin; Botswana
4	Mammalia	Carnivora	Herpestidae	*Liberiictis kuhni*	Liberian Mongoose	Vulnerable	Decreasing	Côte d'Ivoire; Liberia
4	Mammalia	Carnivora	Viverridae	*Genetta johnstoni*	Johnston's Genet	Near Threatened	Decreasing	Côte d'Ivoire; Ghana; Guinea; Liberia; Senegal; Sierra Leone
4	Mammalia	Cetartiodactyla	Hippopotamidae	*Choeropsis liberiensis*	Pygmy Hippopotamus	Endangered	Decreasing	Côte d'Ivoire; Guinea; Liberia; Sierra Leone
4	Mammalia	Primates	Cercopithecidae	*Cercocebus atys*	Sooty mangabey	Near Threatened	Decreasing	Côte d'Ivoire; Guinea; Guinea-Bissau; Liberia; Senegal; Sierra Leone
4	Mammalia	Primates	Cercopithecidae	*Cercopithecus diana*	Diana monkey	Vulnerable	Decreasing	Côte d'Ivoire; Guinea; Liberia; Sierra Leone
4	Mammalia	Primates	Cercopithecidae	*Colobus polykomos*	King colobus	Vulnerable	Unknown	Côte d'Ivoire; Guinea; Guinea-Bissau; Liberia; Sierra Leone
4	Mammalia	Primates	Cercopithecidae	*Procolobus badius*	Western red colobus	Endangered	Decreasing	Côte d'Ivoire; Guinea; Liberia; Sierra Leone
4	Mammalia	Primates	Hominidae	*Pan troglodytes*	Chimpanzee, Common Chimpanzee, Robust Chimpanzee	Endangered	Decreasing	Angola; Burundi; Cameroon; Central African Republic; Congo; Congo, The Democratic Republic of the; Côte d'Ivoire; Equatorial Guinea (Equatorial Guinea (mainland)); Gabon; Ghana; Guinea; Guinea-Bissau; Liberia; Mali; Nigeria; Rwanda; Senegal; Sierra Leone; South Sudan; Tanzania, United Republic of; Uganda
5	Mammalia	Carnivora	Ursidae	*Tremarctos ornatus*	Andean Bear, Spectacled Bear	Vulnerable	Decreasing	Bolivia, Plurinational States of; Colombia; Ecuador; Peru; Venezuela, Bolivarian Republic of
5	Mammalia	Perissodactyla	Tapiridae	*Tapirus pinchaque*	Mountain Tapir, Andean Tapir, Woolly Tapir	Endangered	Decreasing	Colombia; Ecuador; Peru
5	Mammalia	Primates	Atelidae	*Ateles fusciceps*	Black-headed Spider Monkey	Critically Endangered	Decreasing	Colombia (Colombia (mainland)); Ecuador (Ecuador (mainland)); Panama
5	Reptilia	Crocodylia	Crocodylidae	*Crocodylus acutus*	American crocodile	Vulnerable	Increasing	Belize; Colombia; Costa Rica; Cuba; Dominican Republic; Ecuador; El Salvador; Guatemala; Haiti; Honduras; Jamaica; Mexico; Nicaragua; Panama; Peru; United States (Florida); Venezuela, Bolivarian Republic of
6	Aves	Accipitriformes	Accipitridae	*Aegypius monachus*	Cinereous vulture	Near Threatened	Decreasing	Afghanistan; Armenia; Azerbaijan; Bhutan; Bulgaria; China; Croatia; Georgia; Greece; India; Iran, Islamic Republic of; Iraq; Israel;

								Kazakhstan; Korea, Democratic People's Republic of; Korea, Republic of; Kuwait; Kyrgyzstan; Lebanon; Macedonia, the former Yugoslav Republic of; Mongolia; Montenegro; Myanmar; Nepal; Pakistan; Portugal; Russian Federation (Central Asian Russia, Eastern Asian Russia, European Russia); Saudi Arabia; Serbia; Spain; Sudan; Syrian Arab Republic; Tajikistan; Turkey; Turkmenistan; Ukraine; Uzbekistan; Viet Nam; France; Austria; Bangladesh; Belarus; Bosnia and Herzegovina; Cambodia; Egypt; Germany; Gibraltar; Hungary; Japan; Jordan; Latvia; Malaysia; Netherlands; Oman; Poland; Slovakia; Switzerland; Taiwan, Province of China; Thailand; Tunisia; Yemen; Albania; Morocco
6	Aves	Accipitriformes	Accipitridae	*Gyps bengalensis*	White-rumped vulture	Critically Endangered	Decreasing	Afghanistan; Bangladesh; Bhutan; Cambodia; India; Iran, Islamic Republic of; Myanmar; Nepal; Pakistan; Brunei Darussalam; Russian Federation (European Russia)
6	Aves	Accipitriformes	Accipitridae	*Sarcogyps calvus*	Red-headed Vulture	Critically Endangered	Decreasing	Bangladesh; Cambodia; China; India; Lao People's Democratic Republic; Myanmar; Nepal; Viet Nam; Pakistan
6	Aves	Galliformes	Phasianidae	*Coturnix japonica*	Japanese quail	Near Threatened	Decreasing	Bhutan; China; India; Japan; Korea, Democratic People's Republic of; Korea, Republic of; Lao People's Democratic Republic; Mongolia; Myanmar; Russian Federation (Eastern Asian Russia); Thailand; Viet Nam; taly; Réunion; United States (Hawaiian Is.); Cambodia; Philippines
6	Aves	Psittaciformes	Psittacidae	*Psittacula eupatria*	Alexandrine parakeet	Near Threatened	Decreasing	Afghanistan; Bangladesh; Bhutan; Cambodia; India; Lao People's Democratic Republic; Myanmar; Nepal; Pakistan; Sri Lanka; Thailand; Viet Nam; Iran, Islamic Republic of; Qatar; Turkey; United Arab Emirates
6	Mammalia	Carnivora	Ailuridae	*Ailurus fulgens*	Red Panda, Lesser Panda, Red Cat-bear, Tolai Hare	Endangered	Decreasing	Bhutan; China; India; Myanmar; Nepal
6	Mammalia	Carnivora	Felidae	*Neofelis nebulosa*	Clouded Leopard	Vulnerable	Decreasing	Bangladesh; Bhutan; Cambodia; China; India; Lao People's Democratic Republic;

								Malaysia (Peninsular Malaysia); Myanmar; Nepal; Thailand; Viet Nam
6	Mammalia	Carnivora	Felidae	*Prionailurus viverrinus*	Fishing Cat	Vulnerable	Decreasing	Bangladesh; Cambodia; India; Myanmar; Nepal; Pakistan; Sri Lanka; Thailand
6	Mammalia	Carnivora	Ursidae	*Melursus ursinus*	Sloth Bear	Vulnerable	Decreasing	India; Nepal; Sri Lanka
6	Mammalia	Cetartiodactyla	Bovidae	*Bos gaurus*	Gaur	Vulnerable	Decreasing	Bhutan; Cambodia; China; India; Lao People's Democratic Republic; Malaysia (Peninsular Malaysia); Myanmar; Nepal; Thailand; Viet Nam
6	Mammalia	Cetartiodactyla	Bovidae	*Bubalus arnee*	Wild water buffalo	Endangered	Decreasing	Bhutan; Cambodia; India; Myanmar; Nepal; Thailand
6	Mammalia	Cetartiodactyla	Suidae	*Porcula salvania*	Pygmy Hog	Critically Endangered	Decreasing	India
6	Mammalia	Perissodactyla	Rhinocerotidae	*Rhinoceros unicornis*	Indian Rhinoceros, Great Indian Rhinoceros	Vulnerable	Increasing	India; Nepal
6	Mammalia	Primates	Cercopithecidae	*Macaca arctoides*	Stump-tailed Macaque, Bear Macaque, Stumptail Macaque	Vulnerable	Decreasing	Cambodia; China; India; Lao People's Democratic Republic; Malaysia; Myanmar; Thailand; Viet Nam; Hong Kong
6	Mammalia	Primates	Cercopithecidae	*Macaca leonina*	Macaca leonina	Vulnerable	Decreasing	Bangladesh; Cambodia; China; India; Lao People's Democratic Republic; Myanmar; Thailand; Viet Nam
6	Mammalia	Primates	Cercopithecidae	*Trachypithecus pileatus*	Capped langur	Vulnerable	Decreasing	Bangladesh; Bhutan; India (Assam, Manipur, Meghalaya, Nagaland); Myanmar
6	Mammalia	Primates	Hylobatidae	*Hoolock hoolock*	Western Hoolock Gibbon, Hoolock Gibbon, Western Hoolock	Endangered	Decreasing	Bangladesh; India (Assam); Myanmar
6	Reptilia	Crocodylia	Crocodylidae	*Crocodylus palustris*	Mugger crocodile	Vulnerable	Stable	India; Iran, Islamic Republic of; Nepal; Pakistan; Sri Lanka
6	Reptilia	Crocodylia	Gavialidae	*Gavialis gangeticus*	Gharial	Critically Endangered	Decreasing	ndia (Bihar, Uttar Pradesh); Nepal
6	Reptilia	Squamata	Pythonidae	*Python bivittatus*	Burmese python	Vulnerable	Decreasing	Bangladesh; Cambodia; China (Fujian, Guangdong, Guangxi, Hainan, Sichuan, Yunnan); Hong Kong; India (Arunachal Pradesh); Indonesia (Bali, Jawa, Sulawesi); Lao People's Democratic Republic; Myanmar; Nepal; Thailand; Viet Nam; Singapore; United States (Florida)

7	Mammalia	Carnivora	Eupleridae	*Cryptoprocta ferox*	Fossa	Vulnerable	Decreasing	Madagascar
7	Mammalia	Primates	Indriidae	*Indri indri*	Indri	Critically Endangered	Decreasing	Madagascar
7	Mammalia	Primates	Indriidae	*Propithecus candidus*	Silky sifaka	Critically Endangered	Decreasing	Madagascar
7	Mammalia	Primates	Lemuridae	*Eulemur albifrons*	White-headed lemur	Endangered	Decreasing	Madagascar
7	Mammalia	Primates	Lemuridae	*Eulemur macaco*	Black Lemur	Vulnerable	Decreasing	Madagascar
7	Mammalia	Primates	Lemuridae	*Eulemur rubriventer*	Red-bellied Lemur	Vulnerable	Decreasing	Madagascar
7	Mammalia	Primates	Lemuridae	*Hapalemur occidentalis*	Sambirano Lesser Bamboo Lemur, Western Gentle Lemur, Western Grey Bamboo Lemur, Western Lesser Bamboo Lemur	Vulnerable	Decreasing	Madagascar
7	Mammalia	Primates	Lemuridae	*Varecia rubra*	Red Ruffed Lemur, Red-ruffed Lemur	Critically Endangered	Decreasing	Madagascar
7	Mammalia	Primates	Lemuridae	*Varecia variegata*	Black-and-white Ruffed Lemur, Ruffed Lemur	Critically Endangered	Decreasing	Madagascar
8	Aves	Trogoniformes	Trogonidae	*Pharomachrus mocinno*	Resplendent quetzal	Near Threatened	Decreasing	Costa Rica; El Salvador; Guatemala; Honduras; Mexico; Nicaragua; Panama
8	Mammalia	Perissodactyla	Tapiridae	*Tapirus bairdii*	Baird's Tapir, Central American Tapir	Endangered	Decreasing	Belize; Colombia; Costa Rica; Guatemala; Honduras; Mexico; Nicaragua; Panama
8	Mammalia	Primates	Atelidae	*Alouatta pigra*	Black Howling Monkey	Endangered	Decreasing	Belize; Guatemala; Mexico (Campeche, Chiapas, Quintana Roo, Tabasco, Yucatán)
8	Mammalia	Primates	Atelidae	*Ateles geoffroyi*	Geoffroy's spider monkey	Endangered	Decreasing	Belize; Colombia (Colombia (mainland)); Costa Rica (Costa Rica (mainland)); El Salvador; Guatemala; Honduras (Honduras (mainland)); Mexico; Nicaragua (Nicaragua (mainland)); Panama
9	Aves	Struthioniformes	Casuariidae	*Casuarius bennetti*	Dwarf cassowary	Least Concern	Stable	Indonesia (Papua); Papua New Guinea
9	Aves	Struthioniformes	Casuariidae	*Casuarius casuarius*	Southern Cassowary	Least Concern	Decreasing	Australia; Indonesia; Papua New Guinea

9	Mammalia	Diprotodontia	Macropodidae	*Dendrolagus goodfellowi*	Goodfellow's tree-kangaroo	Endangered	Decreasing	Indonesia (Papua); Papua New Guinea
9	Mammalia	Diprotodontia	Macropodidae	*Dendrolagus notatus*	Ifola tree-kangaroo	Endangered	Decreasing	Papua New Guinea
9	Mammalia	Diprotodontia	Phalangeridae	*Spilocuscus rufoniger*	Black-spotted Cuscus	Critically Endangered	Decreasing	Indonesia; Papua New Guine
10	Aves	Psittaciformes	Psittacidae	*Ara militaris*	Military macaw	Vulnerable	Decreasing	Argentina; Bolivia, Plurinational States of; Colombia; Ecuador; Mexico; Peru; Venezuela, Bolivarian Republic of
11	Mammalia	Carnivora	Felidae	*Panthera leo*	Lion	Vulnerable	Decreasing	Angola; Benin; Botswana; Burkina Faso; Cameroon; Central African Republic; Chad; Congo, The Democratic Republic of the; Ethiopia; India; Kenya; Malawi; Mozambique; Namibia; Niger; Nigeria; Senegal; Somalia; South Africa; South Sudan; Sudan; Swaziland; Tanzania, United Republic of; Uganda; Zambia; Zimbabwe
11	Mammalia	Carnivora	Herpestidae	*Bdeogale omnivora*	Sokoko Mongoose	Vulnerable	Decreasing	Kenya; Tanzania, United Republic of
12	Aves	Struthioniformes	Rheidae	*Rhea americana*	Greater rhea	Near Threatened	Decreasing	Argentina; Bolivia, Plurinational States of; Brazil; Paraguay; Uruguay
12	Mammalia	Cetartiodactyla	Cervidae	*Blastocerus dichotomus*	Marsh deer	Vulnerable	Decreasing	Argentina; Bolivia, Plurinational States of; Brazil; Paraguay; Peru
13	Mammalia	Primates	Cercopithecidae	*Cercocebus torquatus*	Collared mangabey	Vulnerable	Decreasing	Cameroon; Equatorial Guinea; Gabon; Nigeria
13	Mammalia	Primates	Cercopithecidae	*Colobus satanas*	Black colobus	Vulnerable	Decreasing	Cameroon; Congo; Equatorial Guinea (Bioko); Gabon
13	Mammalia	Primates	Cercopithecidae	*Mandrillus sphinx*	Mandrill	Vulnerable	Unknown	Cameroon; Congo; Equatorial Guinea; Gabon
13	Mammalia	Primates	Hominidae	*Gorilla gorilla*	Lowland Gorilla, Western Gorilla	Critically Endangered	Decreasing	Angola (Cabinda); Cameroon; Central African Republic; Congo; Equatorial Guinea (Equatorial Guinea (mainland)); Gabon; Nigeria
14	Aves	Piciformes	Picidae	*Leuconotopicus borealis*	Red-cockaded woodpecker	Near Threatened	Decreasing	United States
14	Aves	Piciformes	Picidae	*Melanerpes erythrocephalus*	Red-headed woodpecker	Near Threatened	Decreasing	Canada; Mexico; United States
15	Aves	Strigiformes	Strigidae	*Strix occidentalis*	Spotted owl	Near Threatened	Decreasing	Canada; Mexico; United States
15	Reptilia	Squamata	Helodermatidae	*Heloderma suspectum*	Gila monster	Near Threatened	Decreasing	Mexico; United States

16	Mammalia	Carnivora	Felidae	*Panthera uncia*	Ounce, Snow Leopard	Vulnerable	Decreasing	Afghanistan; Bhutan; China (Gansu, Nei Mongol, Qinghai, Sichuan, Tibet [or Xizang], Xinjiang, Yunnan); India (Arunachal Pradesh, Himachal Pradesh, Jammu-Kashmir, Sikkim, Uttaranchal); Kazakhstan; Kyrgyzstan; Mongolia; Nepal; Pakistan; Russian Federation; Tajikistan; Uzbekistan
16	Mammalia	Carnivora	Ursidae	*Ailuropoda melanoleuca*	Giant Panda	Vulnerable	Increasing	China (Gansu, Hubei - Regionally Extinct, Hunan - Regionally Extinct, Shaanxi, Sichuan)
16	Mammalia	Cetartiodactyla	Bovidae	*Budorcas taxicolor*	Takin	Vulnerable	Decreasing	Bhutan; China; India; Myanmar
16	Mammalia	Primates	Cercopithecidae	*Rhinopithecus roxellana*	Golden snub-nosed monkey	Endangered	Decreasing	China (Gansu, Hubei, Shanxi, Sichuan)
16	Reptilia	Testudines	Trionychidae	*Pelodiscus sinensis*	Chinese softshell turtle	Vulnerable	Decreasing	China; Japan; Taiwan, Province of China; Viet Nam; Thailand; United States
17	Aves	Accipitriformes	Accipitridae	*Pithecophaga jefferyi*	Philippine Eagle	Critically Endangered	Decreasing	Philippines
17	Aves	Columbiformes	Columbidae	*Caloenas nicobarica*	Nicobar pigeon	Near Threatened	Decreasing	Cambodia; India; Indonesia; Malaysia; Myanmar; Palau; Papua New Guinea; Philippines; Solomon Islands; Thailand; Viet Nam
17	Mammalia	Cetartiodactyla	Suidae	*Sus philippensis*	Philippine Warty Pig	Vulnerable	Decreasing	Philippines
18	Mammalia	Primates	Indriidae	*Propithecus verreauxi*	Verreaux's Sifaka	Endangered	Decreasing	Madagascar
18	Mammalia	Primates	Lemuridae	*Eulemur collaris*	Collared brown lemur	Endangered	Decreasing	Madagascar
18	Mammalia	Primates	Lemuridae	*Hapalemur meridionalis*	Bamboo lemur	Vulnerable	Decreasing	Madagascar
20	Mammalia	Carnivora	Felidae	*Pardofelis badia*	Borneo Bay Cat, Bay Cat, Bornean Bay Cat, Bornean Marbled Cat	Endangered	Decreasing	Indonesia (Kalimantan); Malaysia (Sabah, Sarawak)
20	Mammalia	Primates	Cercopithecidae	*Nasalis larvatus*	Proboscis monkey	Endangered	Decreasing	Brunei Darussalam; Indonesia (Kalimantan); Malaysia (Sabah, Sarawak)
20	Mammalia	Primates	Cercopithecidae	*Presbytis chrysomelas*	Sarawak surili	Critically Endangered	Decreasing	Brunei Darussalam; Indonesia (Kalimantan); Malaysia (Sarawak)
20	Mammalia	Primates	Cercopithecidae	*Presbytis frontata*	White-fronted surili	Vulnerable	Decreasing	Indonesia (Kalimantan); Malaysia (Sarawak)

20	Mammalia	Primates	Hominidae	*Pongo pygmaeus*	Bornean Orangutan	Critically Endangered	Decreasing	Indonesia (Kalimantan); Malaysia (Sabah, Sarawak)
20	Mammalia	Primates	Hylobatidae	*Hylobates muelleri*	Muller's gibbon	Endangered	Decreasing	Indonesia (Kalimantan)
20	Reptilia	Crocodylia	Crocodylidae	*Crocodylus siamensis*	Siamese crocodile	Critically Endangered	Decreasing	Cambodia; Indonesia (Jawa - Possibly Extinct, Kalimantan); Lao People's Democratic Republic; Thailand; Viet Nam
25	Aves	Gruiformes	Gruidae	*Anthropoides paradiseus*	Blue crane	Vulnerable	Stable	Namibia; South Africa; Botswana; Lesotho; Swaziland; Zimbabwe
26	Aves	Psittaciformes	Cacatuidae	*Cacatua sulphurea*	Yellow-crested cockatoo	Critically Endangered	Decreasing	Indonesia; Timor-Leste; Singapore
26	Mammalia	Carnivora	Viverridae	*Macrogalidia musschenbroekii*	Brown Palm Civet, Musang, Sulawesi Civet, Sulawesi Palm Civet	Vulnerable	Decreasing	Indonesia (Sulawesi)
26	Mammalia	Cetartiodactyla	Bovidae	*Bubalus depressicornis*	Lowland anoa	Endangered	Decreasing	nesia (Sulawesi)
26	Mammalia	Cetartiodactyla	Bovidae	*Bubalus quarlesi*	Mountain anoa	Endangered	Decreasing	nesia (Sulawesi)
26	Mammalia	Cetartiodactyla	Suidae	*Babyrousa celebensis*	Sulawesi Babirusa	Vulnerable	Decreasing	Indonesia
26	Mammalia	Diprotodontia	Phalangeridae	*Ailurops ursinus*	Bear Cuscus, Bear Phalanger, Sulawesi Bear Cuscus	Vulnerable	Decreasing	Indonesia
26	Mammalia	Diprotodontia	Phalangeridae	*Strigocuscus celebensis*	Small Sulawesi Cuscus, Little Celebes Cuscus, Small Cuscus	Vulnerable	Decreasing	Indonesia
26	Mammalia	Primates	Cercopithecidae	*Macaca tonkeana*	Tonkean macaque	Vulnerable	Decreasing	Indonesia (Sulawesi)
28	Aves	Accipitriformes	Accipitridae	*Milvus milvus*	Red kite	Near Threatened	Decreasing	Albania; Andorra; Austria; Belarus; Belgium; Bosnia and Herzegovina; Bulgaria; Croatia; Czech Republic; Denmark; France; Germany; Gibraltar; Hungary; Iran, Islamic Republic of; Italy; Latvia; Liechtenstein; Lithuania; Luxembourg; Malta; Moldova; Morocco; Netherlands; Poland; Portugal; Romania; Russian Federation (European Russia); Serbia; Slovakia; Slovenia; Spain (Canary Is. - Possibly Extinct); Sweden; Switzerland; Tunisia; Turkey; Ukraine; United Kingdom; Armenia; Azerbaijan; Bangladesh; Cyprus;

								Estonia; Finland; Georgia; Iceland; India; Iraq; Ireland; Israel; Jordan; Lebanon; Libya; Macedonia, the former Yugoslav Republic of; Mauritania; Montenegro; Nepal; Norway; Sudan; Syrian Arab Republic; San Marino
28	Reptilia	Testudines	Emydidae	*Emys orbicularis*	European pond turtle	Lower Risk/near threatened	Unknown	Albania; Algeria; Armenia; Austria; Azerbaijan; Belarus; Bosnia and Herzegovina; Bulgaria; Croatia; Czech Republic; France; Georgia; Germany; Greece; Hungary; Iran, Islamic Republic of; Italy; Kazakhstan; Latvia; Liechtenstein; Lithuania; Macedonia, the former Yugoslav Republic of; Malta; Moldova; Monaco; Montenegro; Morocco; Netherlands; Poland; Portugal; Romania; Russian Federation; Serbia; Slovakia; Slovenia; Spain; Switzerland; Syrian Arab Republic; Tunisia; Turkey; Turkmenistan; Ukraine; Belgium; Luxembourg; United Kingdom
34	Aves	Psittaciformes	Psittacidae	*Anodorhynchus hyacinthinus*	Hyacinth macaw	Vulnerable	Decreasing	Bolivia, Plurinational States of; Brazil; Paraguay
34	Mammalia	Primates	Atelidae	*Ateles marginatus*	White-cheeked spider monkey	Endangered	Decreasing	Brazil (Mato Grosso, Pará)
34	Mammalia	Primates	Atelidae	*Ateles paniscus*	Guiana Spider Monkey, Black Spider Monkey, Red-faced Black Spider Monkey	Vulnerable	Decreasing	Brazil (Amapá, Pará, Roraima); French Guiana; Guyana; Suriname
35	Mammalia	Primates	Cercopithecidae	*Cercopithecus lhoesti*	L'hoest's monkey	Vulnerable	Decreasing	Burundi; Congo, The Democratic Republic of the; Rwanda; Uganda
43	Mammalia	Carnivora	Canidae	*Lycaon pictus*	African wild dog	Endangered	Decreasing	Angola; Benin; Botswana; Burkina Faso; Central African Republic; Chad; Ethiopia; Kenya; Malawi; Mozambique; Namibia; Niger; Senegal; South Africa; South Sudan; Sudan; Tanzania, United Republic of; Zambia; Zimbabwe
52	Mammalia	Carnivora	Canidae	*Canis simensis*	Ethiopian wolf	Endangered	Decreasing	Ethiopia
52	Mammalia	Cetartiodactyla	Bovidae	*Tragelaphus buxtoni*	Mountain Nyala	Endangered	Decreasing	Ethiopia
52	Mammalia	Primates	Cercopithecidae	*Chlorocebus djamdjamensis*	Bale Mountains vervet	Vulnerable	Decreasing	Ethiopia

63	Mammalia	Cetartiodactyla	Bovidae	*Ammotragus lervia*	Aoudad, Barbary Sheep, Uaddan	Vulnerable	Decreasing	Algeria; Chad; Egypt; Libya; Mali; Mauritania; Morocco; Niger; Sudan; Tunisia; Mexico; Spain (Canary Is.); United States
82	Mammalia	Carnivora	Felidae	*Leopardus guigna*	Kodkod	Vulnerable	Decreasing	Argentina; Chile
95	Mammalia	Primates	Cebidae	*Cebus xanthosternos*	Golden-bellied capuchin	Critically Endangered	Decreasing	Brazil (Bahia)
101	Reptilia	Crocodylia	Crocodylidae	*Crocodylus intermedius*	Orinoco crocodile	Critically Endangered	Decreasing	Colombia; Venezuela, Bolivarian Republic of
102	Aves	Otidiformes	Otididae	*Otis tarda*	Great Bustard	Vulnerable	Decreasing	Afghanistan; Armenia; Austria; Bulgaria; China; Croatia; Czech Republic; Georgia; Germany; Greece; Hungary; Iran, Islamic Republic of; Iraq; Italy; Kazakhstan; Kyrgyzstan; Macedonia, the former Yugoslav Republic of; Moldova; Mongolia; Montenegro; Morocco; Portugal; Romania; Russian Federation (Central Asian Russia, Eastern Asian Russia, European Russia); Serbia; Slovakia; Spain; Syrian Arab Republic; Tajikistan; Turkey; Turkmenistan; Ukraine; Uzbekistan; Albania; Belgium; Cyprus; Denmark; Egypt; Finland; France; Gibraltar; Ireland; Israel; Japan; Korea, Democratic People's Republic of; Korea, Republic of; Latvia; Lebanon; Luxembourg; Malta; Netherlands; Pakistan; Saudi Arabia; Tunisia
106	Mammalia	Primates	Indriidae	*Propithecus deckenii*	Van Der Decken's Sifaka, Decken's Sifaka	Endangered	Decreasing	Madagascar
112	Aves	Galliformes	Phasianidae	*Tympanuchus cupido*	Greater prairie chicken	Vulnerable	Decreasing	United States
112	Aves	Piciformes	Picidae	*Campephilus principalis*	Ivory-billed woodpecker	Critically Endangered	Decreasing	Cuba; United State
112	Reptilia	Testudines	Chelydridae	*Macrochelys temminckii*	Alligator snapping turtle	Vulnerable	Need updating	United States
124	Mammalia	Primates	Cercopithecidae	*Cercocebus sanjei*	Sanje mangabey	Endangered	Decreasing	Tanzania, United Republic of
124	Mammalia	Primates	Cercopithecidae	*Procolobus gordonorum*	Udzungwa red colobus	Endangered	Decreasing	Tanzania, United Republic of
129	Aves	Accipitriformes	Accipitridae	*Haliaeetus pelagicus*	Steller's Sea-eagle	Vulnerable	Decreasing	China; Japan; Korea, Democratic People's Republic of; Korea, Republic of; Russian

								Federation (Eastern Asian Russia); Taiwan, Province of China; United States
129	Aves	Anseriformes	Anatidae	*Clangula hyemalis*	Long-tailed Duck	Vulnerable	Decreasing	Austria; Azerbaijan; Belarus; Belgium; Bulgaria; Canada; China; Czech Republic; Denmark; Estonia; Faroe Islands; Finland; France; Germany; Greece; Greenland; Hungary; Iceland; India; Iran, Islamic Republic of; Ireland; Italy; Japan; Kazakhstan; Korea, Democratic People's Republic of; Korea, Republic of; Kyrgyzstan; Latvia; Lithuania; Macedonia, the former Yugoslav Republic of; Mexico; Montenegro; Nepal; Netherlands; Norway; Pakistan; Poland; Romania; Russian Federation (Central Asian Russia, Eastern Asian Russia, European Russia); Saint Pierre and Miquelon; Serbia; Slovakia; Slovenia; Spain; Svalbard and Jan Mayen; Sweden; Switzerland; Turkmenistan; Ukraine; United Kingdom; United States; Armenia; Bermuda; Bosnia and Herzegovina; Croatia; Israel; Jordan; Luxembourg; Portugal; Turkey
129	Aves	Gruiformes	Gruidae	*Grus japonensis*	Red-crowned Crane	Endangered	Decreasing	China; Japan; Korea, Democratic People's Republic of; Korea, Republic of; Mongolia; Russian Federation (Eastern Asian Russia); Taiwan, Province of China
129	Aves	Strigiformes	Strigidae	*Bubo blakistoni*	Blakiston'sfish owl	Endangered	Decreasing	China; Japan; Russian Federation (Eastern Asian Russia)
130	Aves	Charadriiformes	Charadriidae	*Charadrius melodus*	Piping plover	Near Threatened	Increasing	Bahamas; Barbados; Bermuda; Canada; Cuba; Dominican Republic; Guadeloupe; Haiti; Jamaica; Martinique; Mexico; Nicaragua; Puerto Rico; Saint Kitts and Nevis; Saint Pierre and Miquelon; Turks and Caicos Islands; United States; Virgin Islands, British; Virgin Islands, U.S.; Anguilla; Antigua and Barbuda; Ecuador; Saint Vincent and the Grenadines
131	Aves	Psittaciformes	Psittacidae	*Ara glaucogularis*	Blue-throated macaw	Critically Endangered	Stable	Bolivia, Plurinational States of
137	Aves	Galliformes	Phasianidae	*Centrocercus urophasianus*	Greater sage-grouse	Near Threatened	Decreasing	Canada; United States

145	Aves	Psittaciformes	Cacatuidae	*Cacatua moluccensis*	Salmon-crested Cockatoo	Vulnerable	Decreasing	Indonesia
146	Mammalia	Perissodactyla	Equidae	*Equus zebra*	Mountain Zebra, Hartmann's Mountain Zebra	Vulnerable	Unknown	Namibia; South Africa (Eastern Cape Province, Northern Cape Province, Western Cape)
152	Aves	Gruiformes	Rallidae	*Gallirallus australis*	Weka	Vulnerable	Decreasing	New Zealand
152	Aves	Gruiformes	Rallidae	*Porphyrio hochstetteri*	Takahe	Endangered	Stable	New Zealand
152	Aves	Psittaciformes	Strigopidae	*Nestor notabilis*	Kea	Endangered	Decreasing	New Zealand
152	Aves	Psittaciformes	Strigopidae	*Strigops habroptila*	Kakapo	Critically Endangered	Increasing	New Zealand
157	Aves	Psittaciformes	Psittacidae	*Amazona oratrix*	Yellow-headed amazon	Endangered	Decreasing	Belize; Guatemala; Mexico
159	Aves	Charadriiformes	Scolopacidae	*Numenius borealis*	Eskimo Curlew	Critically Endangered	Unknown	Argentina; Barbados; Brazil; Canada; Chile; Mexico; Paraguay; United States; Uruguay
161	Aves	Struthioniformes	Casuariidae	*Casuarius unappendiculatus*	Northern Cassowary	Least Concern	Decreasing	Indonesia; Papua New Guinea
161	Mammalia	Diprotodontia	Macropodidae	*Dendrolagus inustus*	Grizzled Tree Kangaroo	Vulnerable	Decreasing	Indonesia; Papua New Guinea
403	Aves	Gruiformes	Gruidae	*Grus americana*	Whooping Crane	Endangered	Increasing	Canada; United States
527	Mammalia	Cetartiodactyla	Bovidae	*Saiga tatarica*	Saiga/mongolian Saiga, Saiga, Saiga Antelope	Critically Endangered	Decreasing	Kazakhstan; Mongolia; Russian Federation; Turkmenistan; Uzbekistan
589	Mammalia	Primates	Hylobatidae	*Hylobates albibarbis*	Bornean white-bearded gibbon	Endangered	Decreasing	Indonesia (Kalimantan)

Appendix B. **Supplementary Information for Chapter 3**

Percentage Gap Analysis

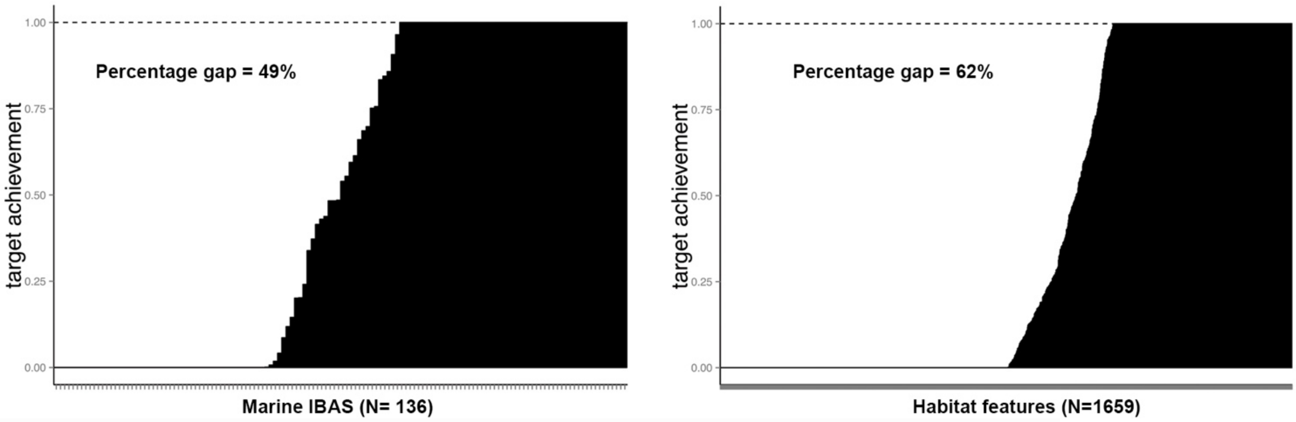
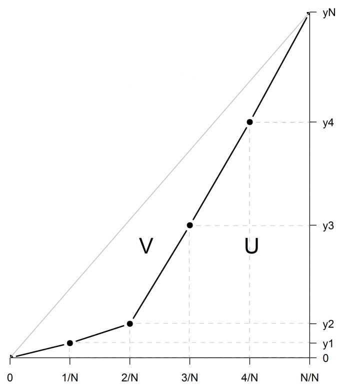


Figure B1: Results of the surrogacy effectiveness analysis derived from the Percentage gap metric.

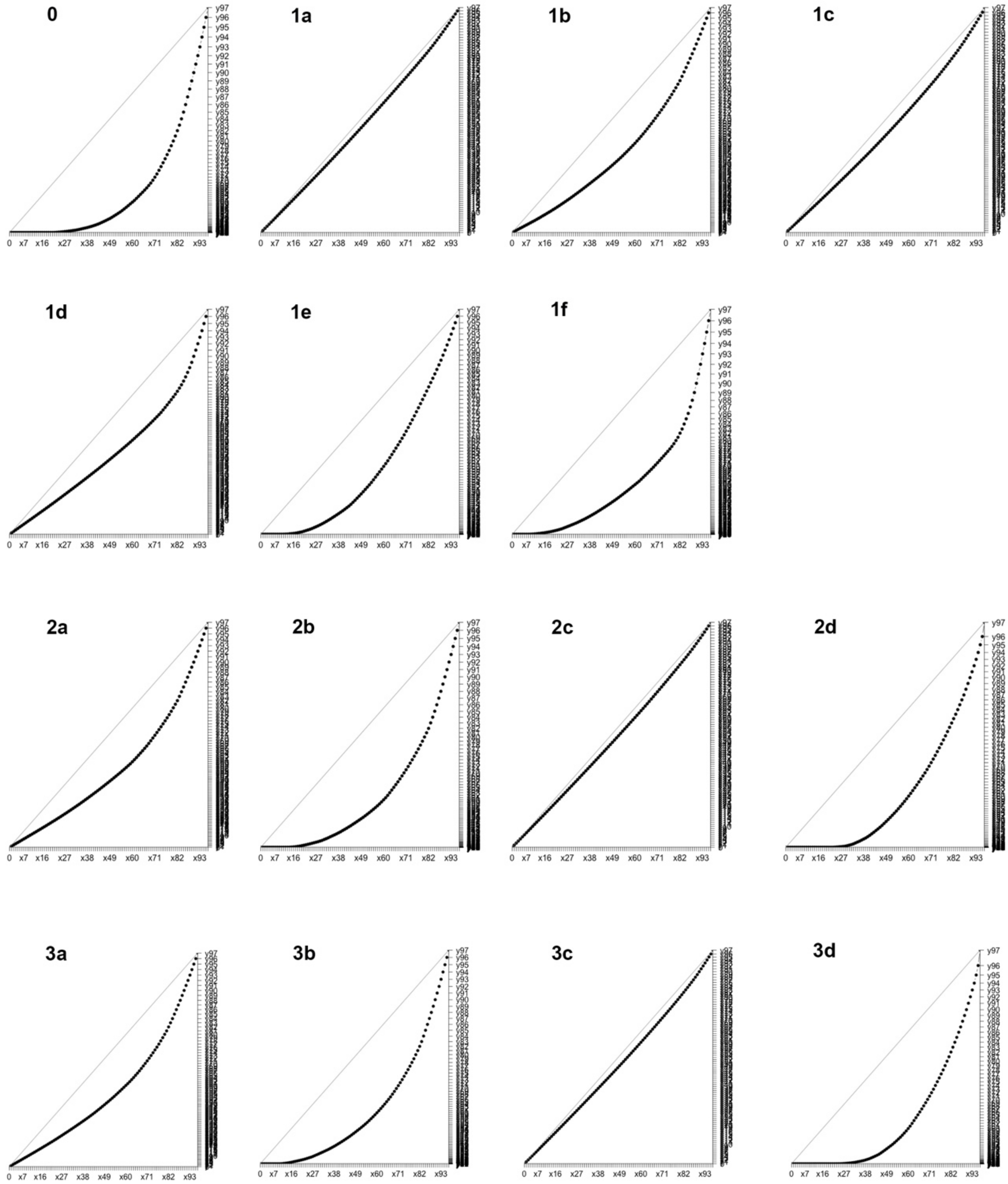
Protection Equality Analysis

Explanation of how Lorenz curves derive proportional protection equality (PE) values.
PE is based on a modified version of the Gini coefficient which ranges between 0 and 1 (Barr *et al.* 2011; Chauvenet *et al.* 2017). For example, if a network protects the same proportion of every feature's distribution, *PE* would be equal to 1. The more disparity there is in protection across features, the more unequal the network is and the lower the PE value.



Text Box 1 This illustration shows how Protection Equality is calculated (here N=5). The line of perfect equality is shown in grey. The black curve is equivalent to the Lorenz curve and formed by the cumulative level of protection of each ecoregion i against i/N, with $1 \leq i \leq N$.

a)



b)

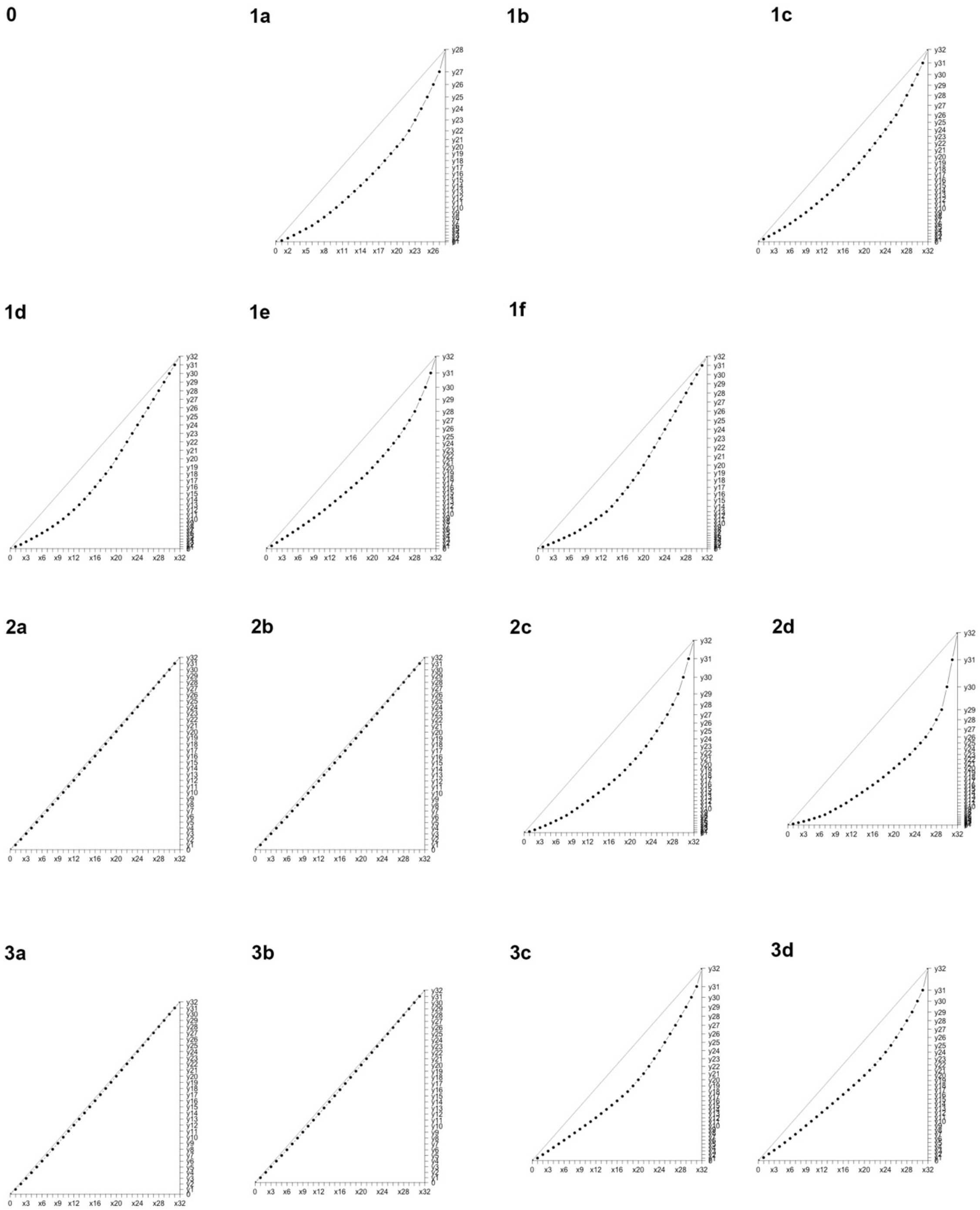


Figure B2. Lorenz curves to derive Protection Equality (PE_P) for a) bioregions b) species abundances.

Appendix C. **Supplementary Information for Chapter 4**

Optimal seascape allocation and process model Code for Matlab

McGowan et al. 2018 Optimal Ocean Zoning under a Sharing versus Sparing Framework

```
% Management impacts on the ecosystem
   dF =  0.35 % What proportion of habitat is damaged by open access fishing
   dM =  0.5*dF; % What proportion of habitat is damaged by managed fishing

% Management costs
   cM = 2; % Cost of managing fishery per unit area
   cR = 1; % Cost of managing reserve per unit area
   cRF = 0.00; % Fixed cost of having a reserve
   cMF = 0.00 % Fixed cost of having a managed area

% Biological parameters
   K = 30; % Carrying capacity of the whole environment
   L = 5; % Fecundity of adults
   s = 1; % Strength of DD mortality (Intrinsic survival)

% Compute MSY in managed area
   MSY= ( sqrt( s*L*(1 - dM) ) - 1 )^2 / (s*L/K);
   BMSY = ( sqrt( s*L*(1 - dM) ) - 1 ) / (s*L/K);

% Adult Biomasss under no fishing
   VirginBiomass = (1 - 1/(s*L)) * K

% Fishing Survival (1 minus these quantities is the fishing mortality)
   sM = 1/sqrt((1-dM)*L*s) % sM is the survival that yields MSY if whole area was managed
   gamma = .1; %equilibrium, post harvest, under open access, as a proportion (gamma of virgin biomass)
   sF = min( 1, (1 + s*L*gamma*VirginBiomass/K)/(s*L*(1-dF)) ) % Survival of adults from open
access fishing

%set the catch constraint to be some proportion of MSY (first option)
%or to open access harvest
  %CT = .05 * MSY;
   CT = (1-sF)*(1 - dF)*s*L*gamma*VirginBiomass/(1 + s*L*gamma*VirginBiomass/K)

%number of steps for the simulation (increasing these values makes the
%graphs smoother, decreasing these values makes the simulation take less
%time
   vG=700;
   vB=100;
   Bmax=1.5; %maximum budget

%vectors for the simulation
   RVec = linspace(0.0,1,vG); % Size of the reserve (optimizing over this)
   MVec = linspace(0.0,1,vG); % Size of the managed area (optimizing over this)
   BVec = linspace(0,Bmax,vB); % Varying Budgets

%Rule of thumb: when this variable is positive Reserves are prefered,
%when negative managed areas are prefered
```

```
   checkS = sM - sF
   ChooseReserves = (1-sF*(1-dF))/cR - (sM*(1-dM)-(1-dF)*sF)/cM

%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%
%%%%%%%%%%%%
% Simulate Optimal Strategy under varying budgets
%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%%
%%%%%%%%%%%%

   %default outputs are set as if all areas are open access (F)
     BestMv=zeros(1,vB); BestFv=zeros(1,vB); BestAv=zeros(1,vB); BestCv=zeros(1,vB);

  for repsB = 1:length(BVec)

    %reset optimization variables for each budget
        BestBiomass = max(0,  (sF*(1 - dF) - 1/(s*L))*K);
        BestCatch = (1 - sF)*(1 - dF)* (s*L*BestBiomass)/(1+s*L*BestBiomass/K);
        BestF=1; BestM = 0; BestR = 0; BestPropB = 0;

    %A variable that forces optimization of catch when harvest constraint
    %is not satisfied (to activate this set catchFlag = -1, to turn off set catchFlag = 1)
        catchFlag = 1;

   for repsR = 1:length(RVec)
     for repsM = 1:length(MVec)

        R = RVec(repsR);
        M = MVec(repsM);
        B = BVec(repsB);

        %Only check pairs of R & M that make sense and can be afforded
        if ( R + M < 1   &   R*cR + M*cM + cRF*(R>0) + cMF*(M>0) <= B )
          % Compute proportion of open access fishing
            F = 1 - R - M;
          % Compute equilibrium adult abundance A, post harvest, then catch C
            A = (sM*(1 - dM)*M + sF*(1 - dF)*F + R - 1/(s*L))*K;
            A = max(A,0);
            C = ( (1 - sM)*(1 - dM)*M + (1 - sF)*(1 - dF)*F ) * (s*L*A)/(1+s*L*A/K);

          % If the result is better than any yet encountered, save them as
          % the optimal outcome. Then keep running the method in case there
          % are better options out there.

            if A > BestBiomass & C > CT
              BestBiomass = A;
              BestCatch = C;
              BestM = M;
              BestF = F;
              BestPropB = ( R*cR + M*cM + cRF*(R>0) + cMF*(M>0) )/B;
              catchFlag = 1;
            elseif catchFlag == -1 & C > BestCatch
              BestBiomass = A;
```

```
                BestCatch = C;
                BestM = M;
                BestF = F;
                BestPropB = ( R*cR + M*cM + cRF*(R>0) + cMF*(M>0) )/B;
            end
        end
    end

 end

 %store the optimal strategy for each Budget value
    BestMv(repsB) = BestM;
    BestFv(repsB) = BestF;
    BestAv(repsB) = BestBiomass/VirginBiomass;
    BestCv(repsB) = BestCatch/MSY;
    BestPBv(repsB) = BestPropB;
end

BestRv = 1 -BestMv-BestFv;

% This is the strategy plot (I don't fill in with color, but you do that
% in photoshop anyway, area under solid curve is proportion in reserve
% area between the solid and dashed curve is area under managed area

   close all %close open figure windows

   figure()
   subplot(4,1,1);
   plot(BVec, BestRv + BestMv, '.'); hold on;
   plot(BVec, BestRv, '-'); hold off;
   xlabel('Budget'); ylabel('Proportion');title('strategy')

   subplot(4,1,2);
   plot(BVec, BestCv, '-');
   xlabel('Budget'); ylabel('Catch');

   subplot(4,1,3);
   plot(BVec, BestAv, '-');
   xlabel('Budget'); ylabel('Biomass');

   subplot(4,1,4);
   plot(BVec, BestPBv, '-');
   xlabel('Budget'); ylabel('Money Spent
```

Additional sensitivity tests

Habitat damage caused by fishing

The following analysis tests the sensitivity of our results when no habitat damage occurs in either zone ($D_M = 0$ and $D_F = 0$). Also refer to Figure 4.4 in the main text where we test the sensitivity of different levels of damage in the open-access zone (D_F).

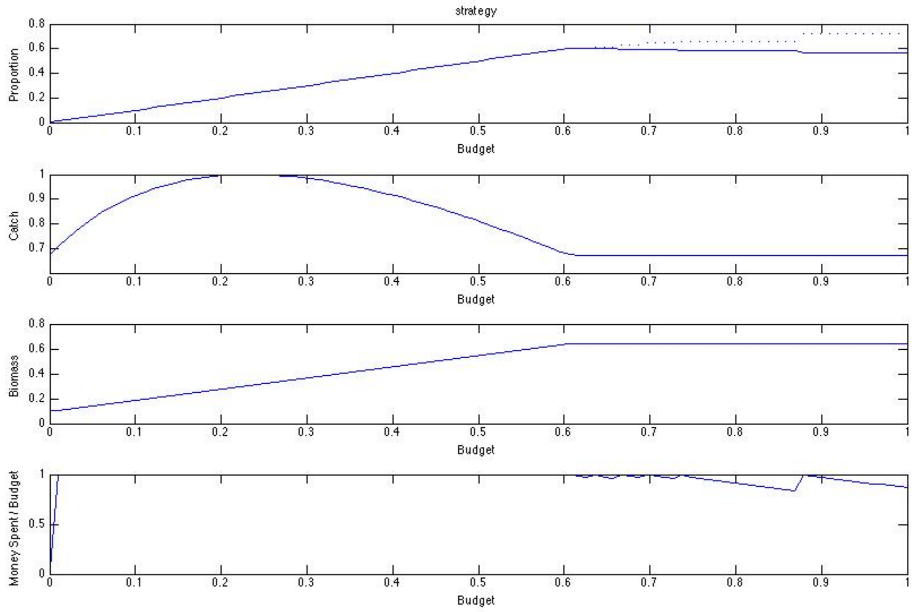


Figure C1. Optimal allocation of zones for our case study parameterization when no damage occurs from fishing efforts. Under the strategy plot (top), the area under the solid line is the proportion in reserves *(R),* the area under the dotted line is the proportion in the managed *(M),* and the remaining proportion is open-access *(F)*.

Sensitivity testing of Cost ratios and incorporating fixed costs

The following analyses test the sensitivity of our results under different cost ratios. We find the strategy of sparing first, where the optimal allocation is comprised of reserves and open-access zones is robust when budgets are small.

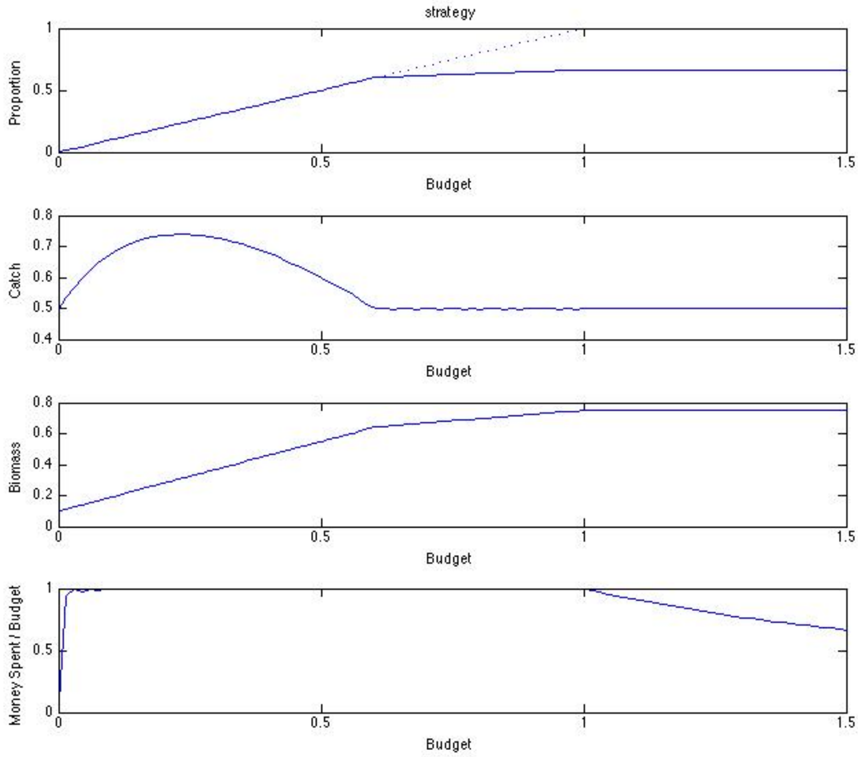


Figure C2. Optimal allocation of zones under different budgets for our case study parameterization when costs to manage and reserve are equal ($C_R = C_M$). Under the strategy plot (top), the area under the solid line is the proportion in reserves *(R),* the area under the dotted line is the proportion in the managed *(M),* and the remaining proportion is open-access *(F)*.

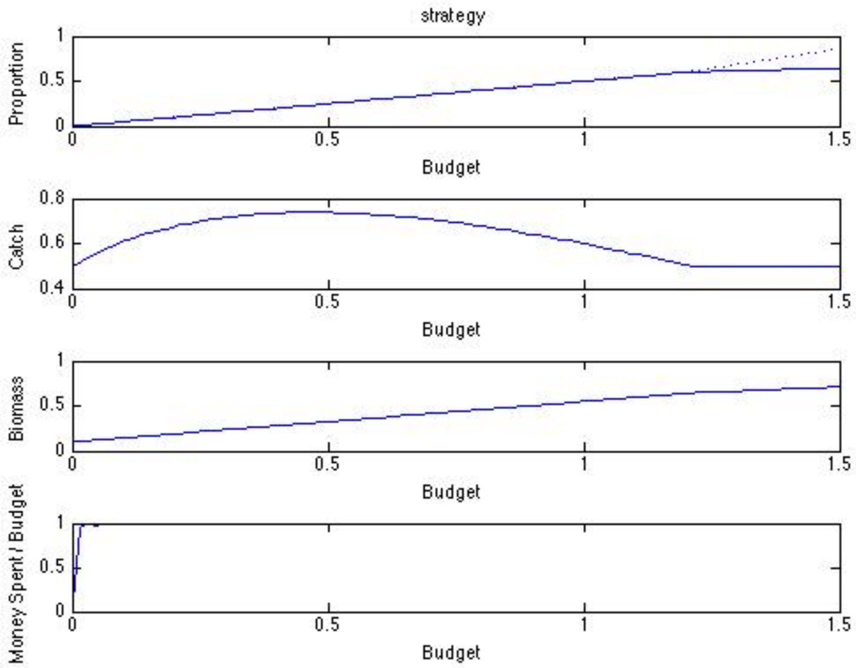


Figure C3. Optimal allocation of zones under different budgets for our case study parameterization when cost to reserve is twice the cost to manage ($C_R = 2C_M$). Under the strategy plot (top), the area under the solid line is the proportion in reserves *(R),* the area under the dotted line is the proportion in the managed *(M),* and the remaining proportion is open-access *(F)*.

Our model also allows fixed costs to be assigned to each management zone in additional to the individual zone costs (see lines 14-15 in Appendix A). Here, the fixed cost of establishing reserves (*cRF*) (e.g. pre-establishment transaction costs) is greater than the fixed cost to establish management (*cMF*).

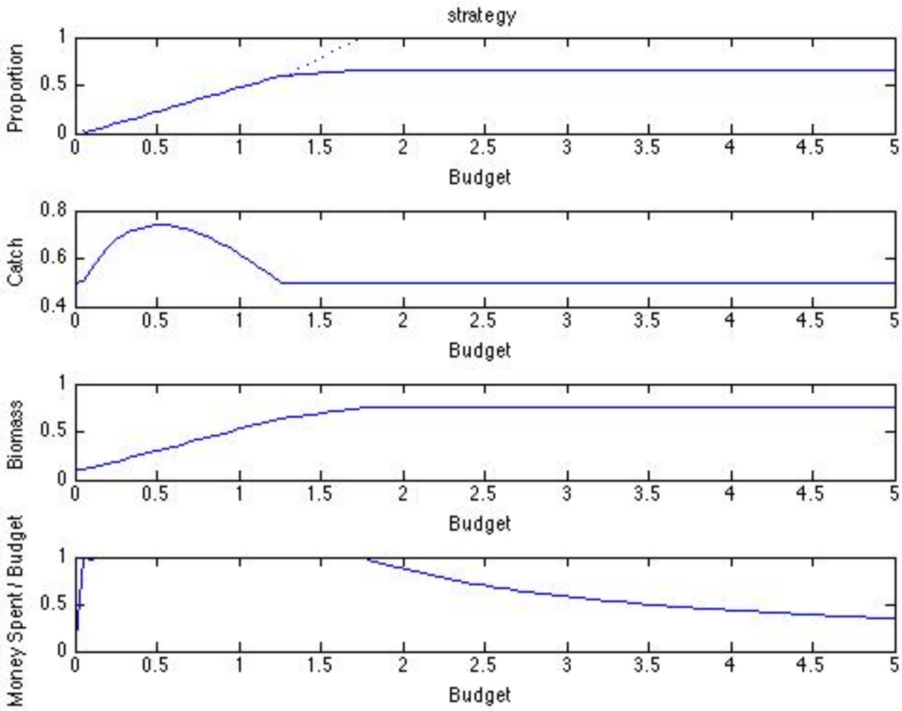


Figure C4. Optimal allocation of zones under different budgets for our case study parameterization when fixed costs are included in addition to proportional costs. Under the strategy plot (top), the area under the solid line is the proportion in reserves *(R),* the area under the dotted line is the proportion in the managed *(M),* and the remaining proportion is open-access *(F)*. For this example, we use the case-study parameterization with the additional paramaters of *cRF*=0.3 and *cMF* = 0.1.

References:

Barr, L.M., Pressey, R.L., Fuller, R.A., Segan, D.B., McDonald-Madden, E. & Possingham, H.P. (2011). A new way to measure the world's protected area coverage. *PLoS ONE*, 6, e24707.

Chauvenet, A.L.M., Kuempel, C.D., McGowan, J., Beger, M. & Possingham, H.P. (2017). Methods for calculating Protection Equality for conservation planning. *PLOS ONE*, 12, e0171591.

Appendix D. **Supplementary information for Chapter 5**

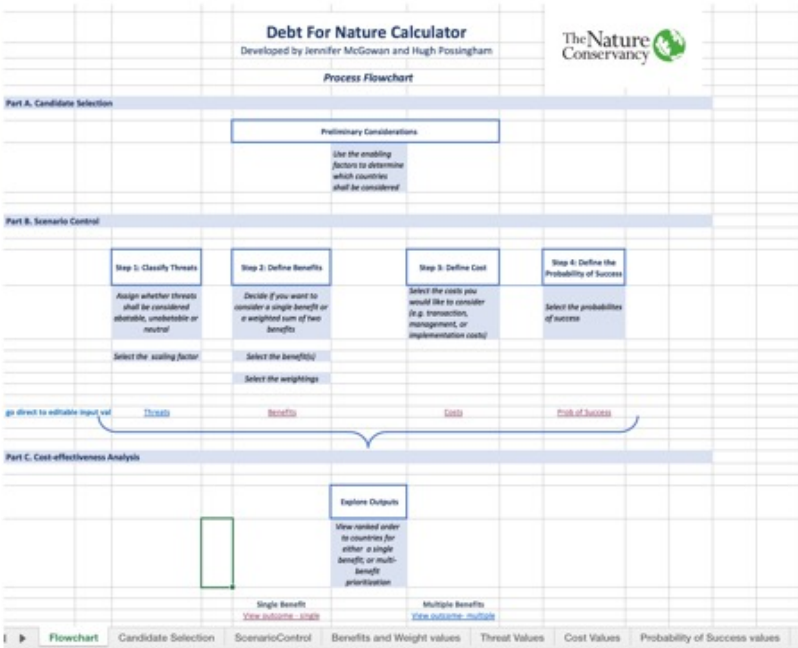
The Nature Conservancy

Debt for Nature Prioritization Tool

Developed by Jennifer McGowan and Hugh P. Possingham

The Process Flowchart outlines the three part protocol to use the tool and provides quick links to:

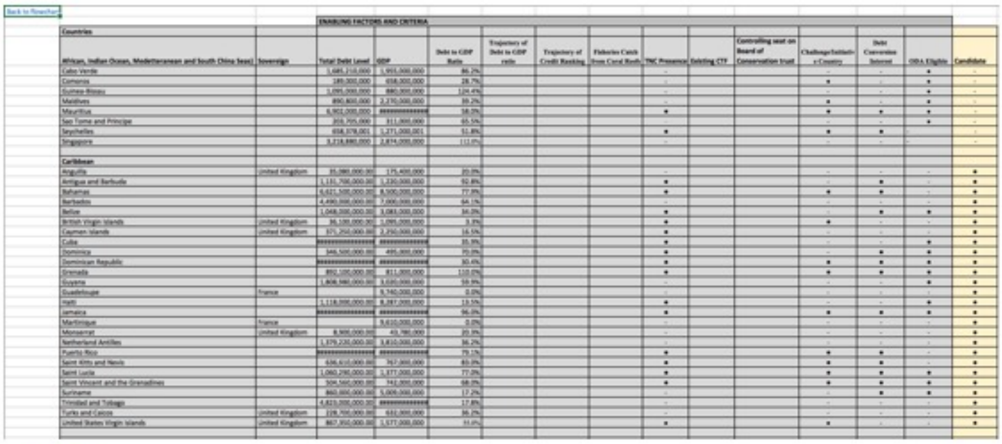
- Candidate Selection
- Scenario control
- the editable input value data pages
- the prioritization outcomes



The Nature Conservancy

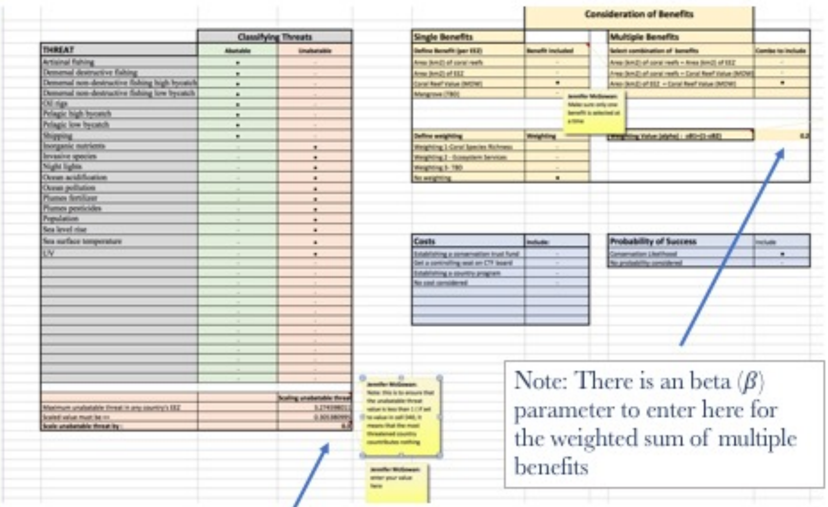
Debt for Nature Prioritization Tool: supporting documentation

Part A: Candidate Selection allows users to track and define the enabling factors that help determine the candidacy of a country. Note: how best to determine candidacy will be context species and can be guided by expert opinion or more formal structured decision making.
The "Candidate" column allows users to toggle individual countries to be in (•) or out (-) of consideration in the prioritization.



Part B: The Scenario Control is where the inputs are defined for an individual scenario's construction.
The four components to consider relate to:

- **Threats** which can be characterized as abatable or unabatable depending on the intended conservation action (e.g. protected areas) Note: scaling factor
- **Benefits** which can be prioritized as a single benefit with optional weighting, or a weighted sum of two benefits (see next page for more details0
- **Costs** which can reflect any start-up or operational costs to take the action
- **Probability of Success** which can reflect any single relative value ranging between 0 and 1.



Note: There is an beta (β) parameter to enter here for the weighted sum of multiple benefits

Note: There is a scaling factor (α) to enter here for the unabatable threats

Debt for Nature Prioritization Tool: supporting documentation

Part B continued: Benefits

There are two options in the "Scenario Control" related to benefits. Each

1. If using "Single benefits with optional weighting" the outcome can be found in **"Do Not Edit Outcome Single"**
2. A weighted sum of two benefits the outcome can be found in **"Do Not Edit Outcome Multi"**

1. Single benefits with optional weighting

Please ensure that only one benefit is selected (•) at a time

There is an optional weighting for the single benefit based on a single ecological or socio-economic value. Only one weighting can be applied at a time.

If no weighting is desired, then please ensure "No weighting" is selected (•).

2. A weighted sum of two benefits

Please only select one combination of benefits (•)

There is an beta parameter (β) which corresponds to

$$B = \beta B_1 + (1-\beta B_2)$$

and where B_1 and B_2 are in order of appearance in the Combined benefit selection.

For example: If "Combo to include" is selected for (•) Area (km2) of coral reefs + Area (km2) of EEZ and $\beta = 0.2$
Then the benefit (B) will be equal to:
Area (km2) of coral reefs *(0.2) + Area (km2) of EEZ*(0.8)

Note: the "benefit weighting" column defaults to 1

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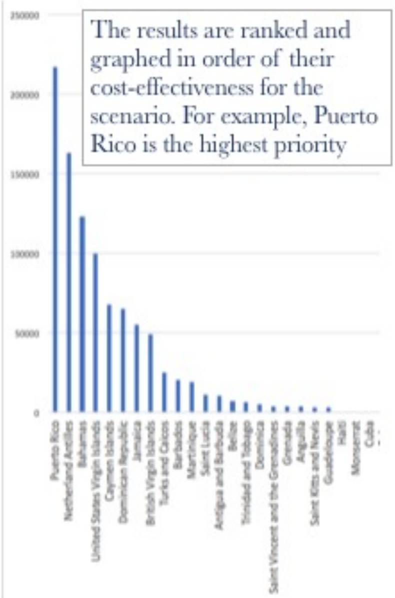
Debt for Nature Prioritization Tool: supporting documentation

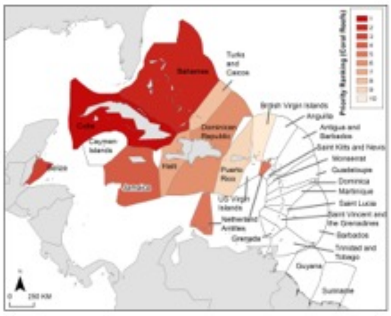
Part C: Exploring Outputs
is where the results of the scenario can be viewed. The "Outcome of Candidates" column calculates the cost-effectiveness for each candidate country (determined in "Part A: Candidate Selection").

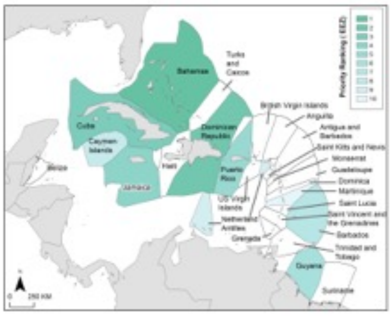
The values for each component of the equation are populated based on what is selected in the "Scenario Control" tab and based on the values provided in the pages from the "Quick Links" in the Process Flowchart.

Maps displaying results were generated outside of the tool but included for illustrative purposes.



The results are ranked and graphed in order of their cost-effectiveness for the scenario. For example, Puerto Rico is the highest priority





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