Systematic conservation planning in marine environments – sensitivities of the planning framework to aspects of scale

Thesis submitted by

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The only things that can be universal, in a sense, are scaling things

Mitchell Feigenbaum

Statement of contribution of others

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Papers in review

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Conference presentations

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Cheok, J., R. L. Pressey, R. Weeks, S. Andréfouët, and J. Moloney. 2016. Detailing the effects of planning-unit size, thematic resolution of habitats, and socioeconomic costs on spatial priorities for marine conservation. 4th Oceania Congress for Conservation Biology, Brisbane, Australia.

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[1] Chapter 2
[2] Chapter 3
[3] Chapter 4
[4] Chapter 5

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Technical reports

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Abstract

Problems of scale abound in the science, governance, and conservation planning of complex social-ecological systems. In systematic conservation planning processes, which aim to effectively and efficiently allocate conservation interventions in space and time, nearly half of the stages in the planning framework involve decisions directly related to scale. The implications of scale-related problems are still poorly understood by conservation planners and researchers, as well as approaches to deal with these problems and integrate explicit multiscale thinking into the planning process. Thus, the overall goal of this thesis is to improve understanding of the different influences of scale on conservation planning outcomes, with the ultimate goal of making specific recommendations to improve the conservation planning framework to deal with scale more explicitly. As such, the structure of this thesis mirrors the relevant stages in the planning framework that involve scale-explicit decisions, organized by the two groups of scale considerations: technical versus practical.

The first research objective of my thesis seeks to understand the extent to which technical aspects of setting spatial priorities for marine conservation ('spatial prioritisations') influence where priorities are determined, and how this relates to conservation strategies that rely on broad, coarse-resolution prioritisations to guide the locations of finer-resolution priorities are actions. I address this objective in Chapter 2 by quantifying the individual and interacting effects of three prioritisation factors on spatial priorities for marine conservation: (1) planning-unit size, (2) thematic resolution of coral reef classes, and (3) spatial variability of socioeconomic costs. I used Fiji and Micronesia as case studies and found that all three factors influenced spatial priorities to different extents, with the spatial variability of socioeconomic costs having the largest influence, followed by planning-unit size and thematic resolution of reef classes. Furthermore, I identified an interaction effect between the thematic resolution of reef classes and the socioeconomic cost data used. These findings have important implications for the strategy of relying on coarse-resolution prioritisations to guide finer-resolution assessments and invalidate a number of implicit assumptions that are made when adopting such strategy.

Progressing to practical considerations of scale, my second research objective seeks to investigate the implications of another strategy commonly assumed or proposed to overcome scale mismatches between regional and local perspectives: dynamically iterating between regional-extent planning and locally applied actions ('iterative planning'), as conservation plans are incrementally implemented across a region. To address this objective in Chapter 3, I specifically

explore how frequently regional priorities should be updated as local actions are gradually implemented. Using Fiji as a case study region, I found that changes in the frequency of updating regional priorities did not influence the total time taken to achieve conservation objectives, or the total extent of final reserve systems. However, I did identify two potential benefits to updating priorities more frequently: faster achievement of objectives for high-priority features, and greater potential to capitalise on areas that have previously had conservation efforts applied. This work provides insights into trade-offs to consider regarding the frequency of updating regional conservation assessments, which vary depending on specific planning contexts.

My third research objective seeks to determine if there is an optimal scale at which to conduct conservation planning, as a precursor to understanding how best to integrate planning across multiple scales ('multiscale conservation planning'). I address this in Chapter 4 by elucidating the respective strengths and weaknesses of conservation plans developed at different jurisdictional levels in the Coral Triangle region (e.g., local, national) to adequately consider multiple social and ecological scales. I found that no plans I assessed were able to adequately address all social and ecological scales, and that plans generally best addressed social and ecological components representative of the same level at which the plan was developed. This research adds nuanced appreciation of the limitations of lower- versus higher-level conservation planning. While these respective limitations are understood as the general inability to consider components at other scales, I demonstrate that these limitations can be attributed to differences in technical versus conceptual abilities. My findings demonstrate the necessity for vertical integration between planning levels as a means to overcome their respective limitations.

The fourth and final research objective of my thesis seeks to investigate the concept of multiscale conservation planning. It is overwhelmingly evident that the consideration and understanding of any social and ecological system must consider multiple scales explicitly. Thus, my thesis culminates in Chapter 5 with a theoretical and empirical examination of what it might mean to conduct multiscale conservation planning, a critical frontier in this field. Using Papua New Guinea and the Solomon Islands as case studies, I provide empirical evidence that refutes the conventional notion that conservation planning across multiple scales occurs unidirectionally ('scaling up' versus 'scaling down') and present a novel archetype that more realistically reflects multiscale planning in practice: 'multidirectional scaling'. I also evaluate factors that impeded or facilitated successful outcomes across multiple scales and reveal six scale-explicit characteristics for effective multiscale planning, the first two of which are novel concepts to the literature: (1) multiscale understanding, (2) scale jumping, (3) leadership characteristics, (4) stakeholder engagement, (5) policy frameworks, and (6) institutional settings. I propose these six

characteristics constitute a new form of conservation capital, 'scalar capital', as a necessary resource or investment for successful outcomes across multiple scales.

My thesis contributes nuanced understanding of the sensitivities of the conservation planning framework to aspects of scale, in both theory and practice. I offer specific recommendations for each of the relevant stages in the conservation planning framework that involve scale-explicit concerns and illuminate some implications of existing problems and influences of scale. Essentially, it is the aim of my thesis to conduct research that can enable conservation practitioners to consider aspects of scale more explicitly and improve the efficacy of conservation planning outcomes. Conservation planning in practice must progress to view any system to manage and govern as inherently complex and multiscale; similarly, planning processes across multiple scales should adopt a 'planning system identity' (such as in complex systems) to correspond in design with the systems that they seek to manage.

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Chapter 1

General Introduction

1 General Introduction

In Chapter 1, I introduce the research context and rationales that lay the foundation for my thesis. I identify and discuss key knowledge gaps within this context, describing the overall goal of this thesis, and how I address each gap with four broad research objectives. Finally, I present an overview of my thesis structure and summarise each of my data-based chapters (Chapters 2-5). I wrote the chapter. Weeks assisted with structuring and editing the chapter. Pressey assisted with editing the chapter.

1 General Introduction

Environmental problems and ecosystem degradation are now globally widespread (Levin et al. 2013, Palomo et al. 2014, Cumming et al. 2017). With the increasing globalisation of socioeconomic and cultural systems (Hill et al. 2015), the scale of problems that require solutions are also becoming increasingly complex (Virapongse et al. 2016). At the same time, there is little evidence of the successes of conservation efforts around the world, in terms of avoided loss of biodiversity or real gains in human well-being (McShane et al. 2011, Gill et al. 2017, Barnes et al. 2018). It is now more pertinent than ever that endeavours in conservation and natural resource governance be conducted effectively and to consider complex social-ecological systems as a whole (Levin et al. 2013).

1.1 Research context

1.1.1 Concept of scale in ecology and environmental science

The importance of scale has been recognised for decades in sciences concerned with the spatial organisation of human activities and physical processes (Marceau 1999). Issues of scale pervade many fields of enquiry, from geography, the very foundation of which concerns scaling, to atmospheric and earth sciences, which define linkages between local and global patterns, to physical and biological oceanography, where scale guides research and defines sub-disciplines, to physics and certain mathematics, where scale is inextricable in investigations (Wiens 1989). More recently, the developing field of environmental assessment and management has recognised the importance of scale and cross-scale dynamics in environmental processes and change (Cash and Moser 2000). These dynamics of scale are inherent in our understanding of patterns and processes occurring in the natural world.

For more than half a century, studies have shown that social and ecological conclusions derived at one scale are specific to that scale and may not remain valid at different scales (Wiens 1989, Marceau 1999). For example, there is substantial evidence in the scientific literature demonstrating an effect of resolution and extent on understanding spatial patterns (Stoms 1994, Qi and Wu 1996, Wu 2004, Rahbek 2005). However, the term 'scale' can refer to several related but distinct concepts (e.g., extent, level, grain, resolution, range, footprint, or cartographic ratio) and contexts (e.g., observation scales, scales of ecological phenomena, or scales used in spatial statistical analyses and decision-making for conservation) (Dungan et al. 2002, Pressey et al. 2007). For clarity of understanding, explicit definitions of the scale-related terms used throughout

my thesis are described in Table 1.1. It is within the context of decision-making in systematic conservation planning (Margules and Pressey 2000), particularly for marine environments, that I examine a number of pervasive scale-related problems.

Table 1.1 Definitions of key scale-related terms and concepts.

Key term	Definition
Scale	The spatial, quantitative, or analytical spectra that are used to measure and understand social or ecological phenomena, and the relational comparisons between different points along these spectra
Level	The units of analysis that are located at different positions on a scale (for example, local and national levels occur along a scale of jurisdictions; Cash et al. 2006)
Extent	The size of the spatial, temporal, quantitative, or analytical dimensions of scale
Resolution	The precision used in measurement (Scholes et al. 2013)
Spatial resolution	The minimum spatial unit of measured information
Thematic resolution	The amount of geospatial information influencing landscape classification or categorisation (Dalleau et al. 2010)
Fine-resolution	Typically associated with conservation prioritisations at local scales (i.e., small spatial extents) and generally involving data at fine (high) resolutions
Coarse-resolution	Typically associated with conservation prioritisations at regional or global scales (i.e., large spatial extents) and generally involving data at coarse (low) resolutions

1.1.2 Systematic conservation planning

The process of systematic conservation planning (hereafter, 'conservation planning') concerns deciding when, where, and how to allocate constrained resources to conserve biodiversity, ecosystem services, and other valuable attributes of the natural environment (Pressey and Bottrill 2009). Conservation planning involves stages from stakeholder engagement through to the application and maintenance of conservation actions (Figure 1.1), and is ideally characterised as a process that should be systematic, flexible, transparent, and accountable (Margules and Pressey 2000). Despite some successful outcomes occurring from conservation planning efforts (e.g., Kapos et al. 2008), factors related to scale that affect nearly half of the stages of the conservation planning framework (Figure 1.1) have frequently resulted in less effective or unsuccessful outcomes (Mills et al. 2010). Of the scale-affected stages identified by Mills et al. (2010), I

distinguish those that concern more technical aspects of conservation planning (e.g., data selection and resolution of assessing spatial priorities; stages 5 & 6 in Figure 1.1) and those more practical in nature (e.g., scoping the optimal scale at which to plan and transitioning from planning at regional scales to implementation at local scales; stages 1, 9 & 10 in Figure 1.1).

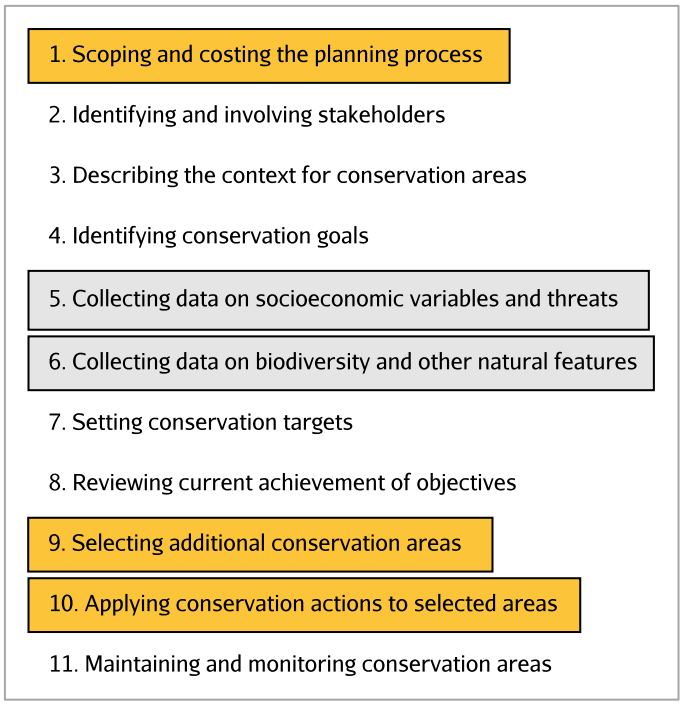


Figure 1.1 Systematic conservation planning framework adapted from Pressey and Bottrill (2009). The framework is represented here as consisting of 11 main stages. While presented in linear order, application of this framework in practice should involve feedbacks between later and earlier stages. For example, information learned from maintaining and monitoring applied conservation areas (stage 11) should be used to inform the selection of any additional conservation areas (stage 9). Stages highlighted with boxes indicate those that involve decisions directly related to spatial scale (Mills et al. 2010). Grey-coloured boxes signify technical concerns in conservation planning (e.g., data selection or resolution of assessments); orange-coloured boxes, practical concerns (e.g., transitioning from regional designs to local conservation actions).

1.1.3 Influence of scale in spatial prioritisations

Spatial prioritisations (hereafter, 'prioritisations') are a key component in conservation planning (stages 5 & 6; Figure 1.1) to ensure efficient and systematic allocation of priority areas for creating protected reserves, which, while not solely adequate for nature conservation, play a critical role in preserving natural biodiversity values (Margules et al. 2002). Prioritisations are typically undertaken by dividing the planning region (the geographic area that is the focus for conservation) into planning units. These are often arbitrary spatial units of assessment and comparison used to determined high-priority areas within which conservation actions might be applied. To allow assessment of the relative conservation value of each unit, planning units are

intersected with different spatial layers commonly containing data on biodiversity features (e.g., Pressey & Logan 1995) or socioeconomic costs (e.g., Richardson et al. 2006).

Originally, biodiversity data (or associated surrogates) alone were used for prioritisation assessments in systematic conservation planning (Margules et al. 2002, Ban and Klein 2009). In the last decade however, conservation planners have increasingly realised the importance of integrating socioeconomic data in prioritisations (Naidoo et al. 2006, Carwardine et al. 2008, Ban et al. 2009a, 2013). The significance of incorporating socioeconomic data stems from the recognition that the success of implementing conservation actions is highly dependent on predominant stakeholder and community support (Stewart and Possingham 2005). Furthermore, there is evidence indicating that conservation priorities are more sensitive to variation in cost data and degree of threats, compared to variability in how biodiversity is measured (Bode et al. 2008).

Dealing with these spatial data layers alone involves decisions about various aspects of resolution (i.e., spatial, thematic; Table 1.1); additionally, the size or extent of planning regions and planning units can vary. Unsurprisingly, a number of terrestrial studies on conservation prioritisations have shown that the resolution (of spatial data or planning units) can greatly influence their outcomes (Pressey and Logan 1995, 1998, Rouget 2003, Araújo 2004, Pascual-Hortal and Saura 2007, Arponen et al. 2012). Fewer studies demonstrate analogous findings in marine environments (Richardson et al. 2006, Dalleau et al. 2010, Hamel et al. 2013). Thus, there is strong evidence to suggest that the different modes of resolution acting, and potentially interacting, in the prioritisation process will impact how and where areas are identified as priorities. The extent of these effects is not fully understood, however. Given that prioritisations are increasingly used to determine or justify conservation actions, this is a vital knowledge gap to fill.

Conservation planning often favours large planning regions because these can capture regional-scale ecological processes and patterns, which underpin regional biodiversity (Poiani et al. 2000, Rouget et al. 2006, Pressey et al. 2007). However, there typically exists a trade-off between extent of the planning region, the size (spatial resolution) of the planning units, and the attainable underlying data used to assess the region. This trade-off primarily exists due to resource limitations: it is generally not feasible to collect, map, or analyse biodiversity or socioeconomic data at very fine spatial and thematic resolutions across large extents (Rouget 2003). As a result, prioritisations conducted across large regions are limited by both coarse-resolution data and large planning units (hereafter, 'coarse prioritisations') (Mills et al. 2010), and are associated with a number of shortcomings (Pressey and Logan 1998, Rouget 2003, Richardson et al. 2006, Payet et al. 2010). In particular, Hamel, Andréfouët and Pressey (2013) found that prioritising with coarse

(large) planning units exacerbated problems encountered when trying to achieve conservation objectives for small islands that included fishery objectives. This is a salient consideration with almost all the existing tropical coral reef habitats occurring in the Indo-Pacific (Spalding et al. 2007), where most nations are archipelagic.

Coarse (e.g., national level) prioritisations of conservation priorities have become increasingly common around the world, both in terrestrial and marine environments (Olson and Dinerstein 2002, Alpine and Hobday 2007, Klein et al. 2010, Beger et al. 2013, Mazor et al. 2014). One reason for this is the influence of international commitments to achieve global conservation targets (e.g., the Convention on Biological Diversity; Jones et al. 2011, Gray et al. 2014, Watson et al. 2016). Additionally, environmental problems are rarely restricted to local extents and this motivates conservation planning at regional or global extents, which attempts to formulate appropriately-scaled responses to the large scale of problems being managed (Agardy 2005). As a strategy to overcome the disparity between necessary broad-scale ecological views and the much finer-resolutions at which conservation actions need to be applied (Mills et al. 2010), coarse prioritisations have been suggested as a starting point to guide subsequent finer-resolution conservation assessments (Larsen and Rahbek 2003, Fjeldså 2007). However, there are at least two major problems that can arise with this strategy. These are related to two implicit assumptions that are likely to be invalid. One is the assumption of homogeneity, i.e., that coarse-resolution planning units are internally homogeneous with respect to conservation priority, or that biodiversity features with broad thematic resolution (e.g., ecoregions) are homogeneous with respect to physical and biological characteristics. The other assumption is that of nestedness, i.e., that fine-resolution priorities naturally nest spatially within coarse-resolution priorities. These potentially significant problems have received very little attention in the prioritisation literature, despite being critical aspects that may be undermining recent efforts directed at broad-extent (i.e., regional, international) marine conservation prioritisations (e.g., Beger et al. 2013).

The problem with the first assumption of homogeneity is that natural environments inherently contain spatial and temporal heterogeneity, which occurs at different spatial and temporal scales (Barry and Dayton 1991, Pinel-Alloul 1995, García-Charton and Pérez-Ruzafa 1999, Naidoo et al. 2006). Prioritisation of a large region at coarse-resolution (i.e., with large planning units) will identify areas that are homogenous with respect to conservation priority, while underlying environmental and/or socioeconomic data at fine-resolutions would almost certainly be heterogeneous within the prioritised planning unit (Figure 1.2). This means that the underlying variation is essentially lost in the translation to coarse-resolution priorities, which can potentially result in the exclusion of fine-resolution priorities (see Rouget 2003, Possingham et al. 2005, Richardson et al. 2006). If fine-scale variation is not considered during the prioritisation stage,

effective conservation actions for these fine-resolution patterns and processes are incidental at best. While one study has demonstrated that reasonable incidental representation is possible with coarse prioritisations identifying marine priorities (Bridge et al. 2016), this haphazard approach should not be relied upon. A considerable challenge here is to determine what levels of resolution are most appropriate for conservation planning, noting the trade-offs between the costs of obtaining very fine-resolution data and expedited decision-making possible with more easily available coarse-resolution data.

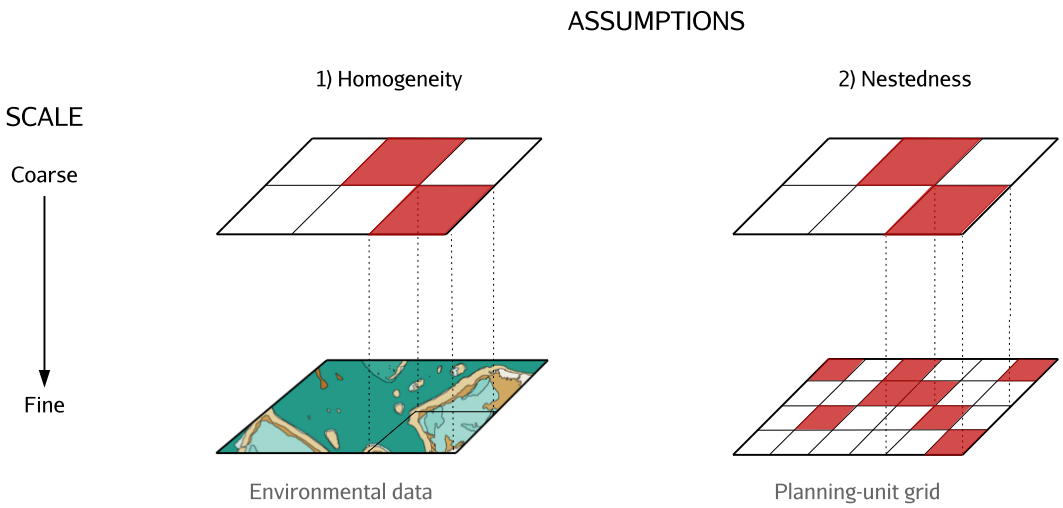


Figure 1.2 Schematic representation of the assumptions of homogeneity (1) and nestedness (2), implicit in the approach of relying on coarse prioritisations to guide finer-resolution assessments. Red grid cells indicate high-priority planning units, demonstrating the likely disparities with fine-resolution environmental variation (1) or priority areas determined using smaller planning units (2).

The second assumption of nestedness is similarly unreliable because it presumes that conservation priorities determined with coarse-resolution planning units will encompass the same areas as those determined with fine-resolution planning units. As with the first assumption, using coarse-resolution planning units will likely result in a loss of compared environmental (or socioeconomic) heterogeneity between units, with priorities over larger areas averaged out and finer variations lost. This means that areas identified as high priority using coarse-resolution planning units may not necessarily translate to an area of the same priority level when fine-resolution planning units are used (Figure 1.2). Such may be the case when the high-priority, fine-resolution planning unit is a small fraction of an otherwise low-priority large (coarse) planning unit, resulting in a low priority level overall. Given the increasing prevalence of regional planning exercises and the common suggestion that planning will progress from coarse to finer scales in many of them (e.g., Klein et al. 2010, Beger et al. 2013), the strategy of relying on coarse prioritisation to guide finer ones and its implicit assumptions must be examined empirically.

1.1.4 Transitioning between regional conservation assessments and local actions

A further problem related to scale occurs when navigating the transition from broad-scale, coarse-resolution regional conservation assessments to the implementation of conservation actions at local levels. By assessment, I refer to the design phase of conservation planning (stages 1-9; Figure 1.1), particularly the spatial prioritisation process. By implementation, I reference the translation of assessments into applied actions on the ground (stage 10; Figure 1.1), such as the implementation of protected areas. With regional priority assessments, the spatial extent of the planning region and the planning units becomes mismatched to that of implementing conservation actions, which occur at much finer-resolution local levels (Pressey et al. 2013). For example, mean sizes of marine protected areas have been reported around ranges of 1-100 km^2 (Edgar et al. 2014, White et al. 2014), while planning-unit sizes in coarse prioritisation have been as large as 900 km^2 (Venter et al. 2014) or even using whole marine ecoregions (Klein et al. 2010). This mismatch of scales is understood to be an important factor in the failure of regional or global planning to inform local actions (Mills et al. 2010) and is well known as an 'implementation crisis' (Biggs et al. 2011) in conservation. This can be seen as a quintessential 'problem of fit' (Cash et al. 2006).

As a result of the mismatch between regional conservation assessments and local actions, modifications are required to reconcile the differences between these spheres of operation. This includes incorporating newly obtained information on ecological or social features or constraints, updating the original design to adjust for over- or under-achievement of objectives, and evaluations of implementation procedures (Mills et al. 2010, Pressey et al. 2013). The most effective strategy to incorporate these modifications into the planning process is still not well understood. Nevertheless, it is argued in the literature that a necessary strategy to incorporate these modifications is in a dynamic, iterative manner (hereafter, 'iterative planning') (Holness and Biggs 2011, Pressey et al. 2013, Beger et al. 2015). As new information emerges during the on-ground implementation of conservation actions and adjustments to regional assessments are inevitably made, this will potentially change new areas of priority for subsequent actions (Pressey et al. 2013). Thus, feedbacks between the regional and local perspectives should ideally occur as regional plans transition to implemented actions.

There is considerable acknowledgement of cross-level (between institutions of the same operating scale; e.g., between regional institutions) and cross-scale (between institutions of different operating scales; e.g., between regional and local institutions) dynamics in conservation (Berkes 2002, 2006, Agardy 2005, Cash et al. 2006, Knight et al. 2006, Mills et al. 2010, Pressey et al. 2013). However, no investigations have been conducted to quantify the potential differences in important planning outcomes, such as the extent to which designs need to be adjusted when

making the dynamic transition to applied local actions, or the implications of this on the achievement of conservation objectives through time. Some studies have quantified differences in the efficiency of transitioning to local implementation by incorporating governance units in the assessment stages (Aswani and Lauer 2006, Weeks et al. 2010b, Horigue et al. 2015). This approach operates on the assumption that doing so facilitates local actions through minimising the amount of adjustment necessary to planning units. Mills et al. (2014) have proposed using social network analyses to facilitate implementation by strengthening linkages between local and regional conservation actors.

Despite these studies, explicit methods for the dynamic transition between different scales of planning and action remain obscure and have yet to be formally quantified or operationalised. The successful transition between regional assessments and local implementation is a vital knowledge gap to fill in improving conservation outcomes (Knight et al. 2008). Without this knowledge, planning is rendered essentially inapplicable and ineffective, wasting already scarce conservation resources (Bottrill et al. 2008).

1.1.5 Integrating planning across multiple scales with social-ecological systems theory

Over the last two decades, social-ecological systems (SES) theory has increasingly been adopted as the lens through which environmental management and governance are understood and applied (Berkes and Folke 1998, Ostrom 2009, Ban et al. 2013, Cumming et al. 2015). In contrast to earlier conservation concepts, where the focus was on separating human and ecological systems through exclusionary protected areas, SES theory recognises that every ecological system is inextricably linked to a social system (Palomo et al. 2014). Ostrom's (2009) framework for SES theory extended considerations, albeit implicitly, to the multiple scales that operate within these complex systems (Figure 1.3). The consideration of multiple social and ecological scales has been a crucial move towards dealing with problems of scale that have long challenged our ability to effectively manage the environment (Cash and Moser 2000).

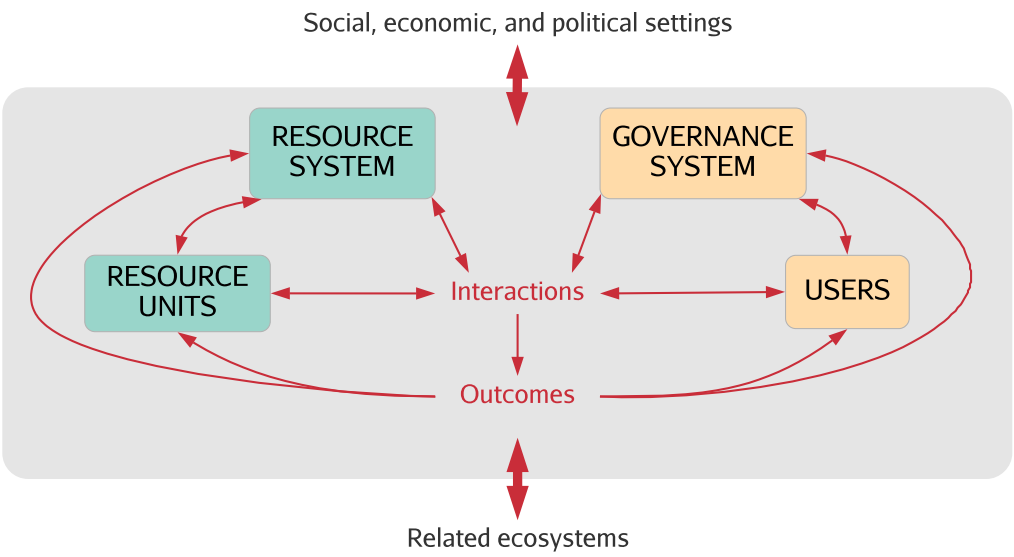


Figure 1.3 Social-ecological systems framework reproduced from Ostrom (2009). The four main subsystems of the broader SES system are shown as resource units, resource system, governance system, and users.

Problems of scale in conservation planning commonly involve scale mismatches or problems of fit (Bodin et al. 2014, Epstein et al. 2015), which arise when social or ecological patterns or interactions are scale-dependent (Levin 1992, Ament and Cumming 2016) and operate on different scales (Cumming et al. 2006, 2017, Guerrero et al. 2013). A widespread manifestation of this problem of fit occurs when the spatial extent of an environmental resource misaligns with the jurisdictional extent of institutions in place to manage the resource. An example can be found in the upper tributary watershed in montane mainland Southeast Asia, where multiple counties and local governments have jurisdiction over different parts of the same resource (Lebel et al. 2008). Another common problem relates to temporal mismatches, whereby the temporal scale of the environmental phenomenon operates on timescales different to the institutional response. For example, fishing quotas that are calculated on the basis of maximum sustainable yield are incompatible with the intrinsic dynamics of fish populations (Epstein et al. 2015).

With the prevalence of scale-related problems in conservation planning, the need to consider multiple scales explicitly is now extensively recognised in the literature (Lengyel et al. 2014, Weeks et al. 2014, Guerrero and Wilson 2017). Suggested methods have included: using basic decision frameworks to incorporate scale considerations in conservation triage (du Toit 2010); evaluating stakeholders at different scales through social network analysis to inform actions (Guerrero et al. 2013, Mills et al. 2014); and integrating ecological data representing different scales (e.g., ecosystems, processes, species) into spatial prioritisations (Squeo et al. 2012, Bombi et al. 2013). Other approaches propose sequential planning processes undertaken at successively

higher or lower levels. In 'scaling up', separate local planning processes are coordinated and placed within a broader context (e.g., Lowry et al. 2009). In 'scaling down', planning incorporates patterns or processes at progressively finer scales, within areas of interest identified at broader scales (Groves et al. 2002). Scaling up and scaling down are not without their limitations however, with both approaches facing governance and implementation challenges (Lovell et al. 2002, Lowry et al. 2009, Mills et al. 2010, Gaymer et al. 2014).

Despite frequent calls for integration across scales, conservation scientists, policymakers and practitioners have yet to define explicitly what this means or demonstrate how they should approach it (Guerrero et al. 2015b). Given that hundreds of conservation plans are developed every year (Álvarez-Romero et al. in press), more effective and deliberate planning across multiple scales could improve conservation outcomes and achieve greater impact with the limited resources available for conservation. Currently, our understanding of systematic conservation planning techniques far exceeds our ability to apply them effectively to real-world conservation problems (Knight et al. 2006). The core of these problems revolves around: uncertainties about the size of planning units and data resolution, the scales at which we assess conservation priorities and how these translate to actual actions on the ground, and the explicit integration of multiple social and ecological scales throughout the conservation planning framework.

1.2 Improving explicit considerations of scale in conservation planning

It is apparent that conservation planning must move towards more explicit multiscale considerations, in both theory and practice. To provide the contextual backbone of my thesis, I identify specific knowledge gaps that pertain to the scale-affected stages in the conservation planning framework (Figure 1.1) and outline these below.

1.2.1 Technical concerns (stages 5 & 6)

While scale-explicit considerations have likely occurred most in these stages in the conservation planning literature, there is still a lack of understanding of scale influences on spatial prioritisations in marine environments. Importantly, very few studies have focused on any interaction effects between prioritisation factors. Related to these scale-related concerns is the often-assumed strategy of relying on coarse-resolution prioritisations to guide the locations of finer-resolution prioritisations. This is seen as a way of overcoming the discrepancy between necessarily broad, regional perspectives on planning and fine-resolution data, usually only available across smaller extents. Here, I identify two important knowledge gaps to fill:

1. Understanding the individual and interacting effects between different levels of resolution of prioritisation factors on marine spatial priorities.
2. Examining the ability of coarse prioritisations to guide finer-resolution assessments.

1.2.2 Practical concerns (stages 1, 9 & 10)

To begin with, it is still not well understood what the optimal scale to plan at is. Does an optimal scale of planning even exist, or is integrating conservation planning across multiple scales (hereafter, 'multiscale planning') necessary? If multiscale planning is required, how does the initial scale of planning influence the essential later stages of implementing conservation action? The implementation crisis in conservation planning is a critical bottleneck in the effectiveness and success of conservation planning outcomes. The process of dynamically transitioning between regional conservation assessments and locally applied actions and the implications of doing so, particularly in prevailing contexts where actions are applied incrementally over time, must be examined further. Additionally, if multiscale conservation planning is being undertaken, it is crucial to evaluate these processes to understand specific conditions that influence successful outcomes so that future applications can be more effective. I identify five key knowledge gaps related to these concerns:

1. Identifying the optimal frequency with which regional assessments should be updated in the transition to local actions.
2. Understanding the extent to which plans need to change when transitioning from regional assessments to local actions.
3. Elucidating the respective strengths and weaknesses of plans developed at different levels in an SES context.
4. Understanding whether multiscale conservation planning occurs in practice and if so, through what mechanisms.
5. Discerning the factors that impede or facilitate successful outcomes in multiscale planning.

1.3 Thesis goals and objectives

Given the diverse problems of scale that can manifest throughout the conservation planning framework, the overall goal of this thesis is to understand the different influences of scale on outcomes of this framework, with the ultimate goal of making specific recommendations to improve the framework to deal with scale more explicitly. To achieve these goals, I address the knowledge gaps discussed above with the following research objectives:

Objective 1. Understand the extent to which technical aspects of setting spatial priorities for marine conservation influence where priorities are determined, and how this relates to assumptions of homogeneity and nestedness.

Objective 2. Quantitatively investigate and operationalise the transition from regional conservation assessments to implementing local actions.

Objective 3. Determine if there is an optimal scale at which to plan to address multiple social and ecological scales.

Objective 4. Investigate the theory and practice behind multiscale conservation planning.

1.4 Study regions

I address the research objectives outlined above using countries with developing economies and significant marine-resource dependency in Southeast Asia and the Western Pacific as case study regions. The Coral Triangle region includes six countries: Indonesia, Malaysia, Papua New Guinea (PNG), Philippines, the Solomon Islands (SI), and Timor Leste. This region is of particular interest and concern for conservation scientists and practitioners because of its global biodiversity importance coupled with highly varied socioeconomic, cultural, and political contexts (Mills et al. 2010, Fidelman et al. 2012). All but two of the Coral Triangle countries (Malaysia and Timor Leste) have some form of decentralised natural resource governance (where decision-making power is devolved to local governments or customary clans; Fidelman et al. 2012), for which problems of scale mismatches are known to be especially acute (Mills et al. 2010). Similarly, the nearby archipelagic and non-industrialised countries of Fiji and Micronesia (the latter considered here to include the Mariana Islands, Marshall Islands, Palau, Guam, and the Federated States of Micronesia) also urgently require effective conservation action but the mismatch between regional-level planning and local-level implementation is large, with primarily devolved resource governance occurring in conjunction with complex social, economic and political factors shaping conservation decisions (Govan et al. 2009, Weeks et al. 2010b, Hamel et al. 2013, Horigue et al. 2015). For these reasons, the developing economies of Southeast Asia and the Western Pacific are significant and relevant study regions to use in my investigations on the problems of scale and scale mismatches in the conservation planning framework. Furthermore, with much of the conservation planning literature published in the context of industrialised countries with centralised management (Fisher et al. 2011) and evidence to suggest these do not apply in non-industrialised contexts (Keppel et al. 2012b), there is a strong need for a more specific focus within these study regions.

1.5 Thesis structure

This thesis presents a total of six chapters (Figure 1.4), which consist of a general introduction (this chapter, **Chapter 1**), four data-based chapters (**Chapters 2-5**), and a general discussion (**Chapter 6**). All data-based chapters are presented in this thesis as manuscripts formatted for publication in peer-reviewed journals. The structure of this thesis reflects the varying natures of concern that describe the scale-related decisions and problems that can manifest in the conservation planning framework (Figures 1.1 & 1.4): technical concerns (**Chapter 2**), practical concerns (**Chapters 4 & 5**), and the transition between these two (**Chapter 3**).

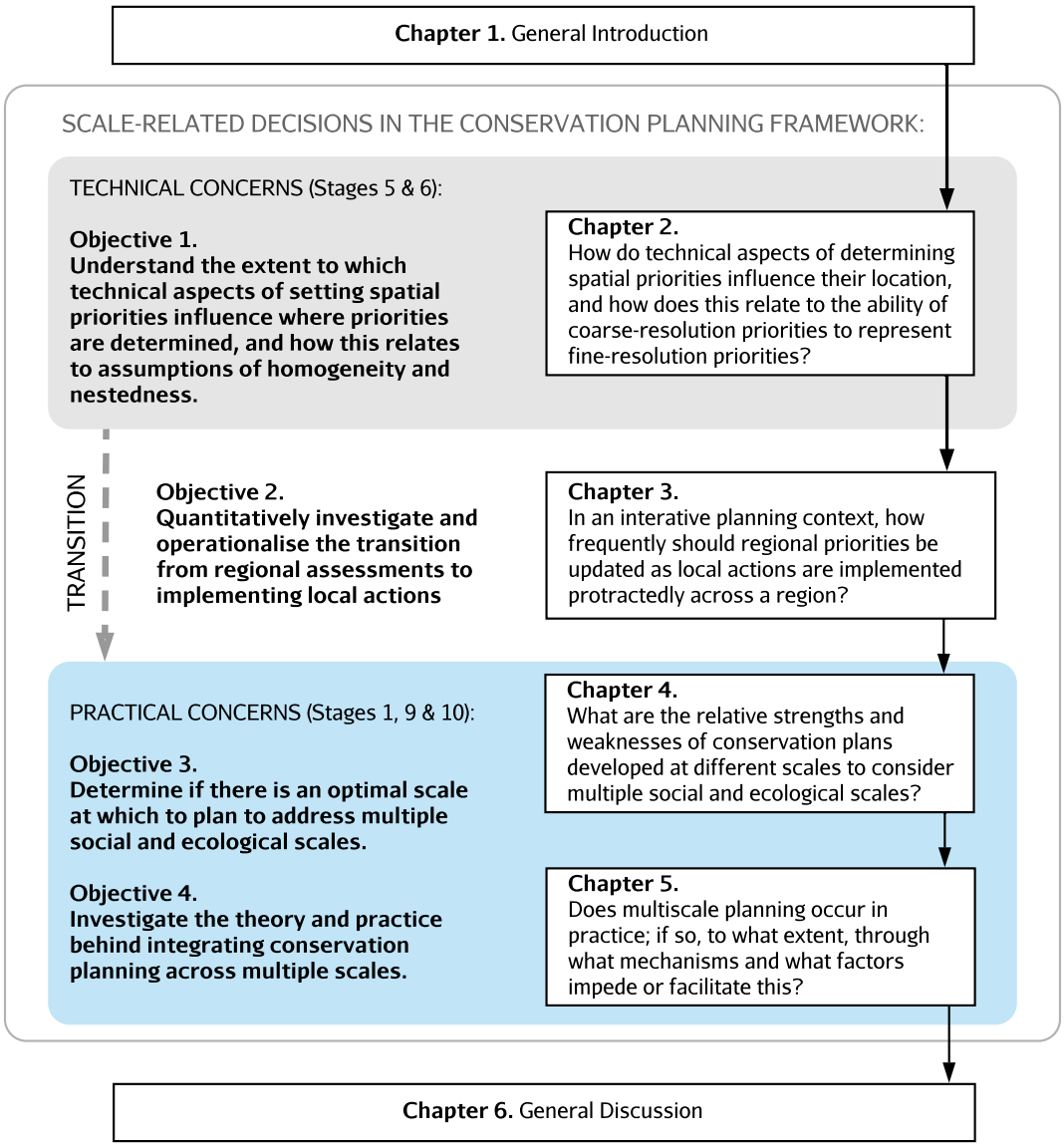


Figure 1.4 Overview of thesis structure.

In **Chapter 1** (this introductory chapter), I establish the background and context for my research. Using the different stages in the conservation planning framework that involve scale-explicit decisions as thematic groups (i.e., technical and practical concerns, and transitioning between these), I identify broad research objectives that pertain to scale within each of these groups and specific related knowledge gaps.

In **Chapter 2** where I address Objective 1, I determine the extent to which technical aspects of setting spatial priorities for marine conservation influence how and where priorities are determined. I examine three prioritisation factors: (1) the size of planning units, (2) thematic resolution of reef-class maps, and (3) the spatial variability of socioeconomic cost data used, using the conservation prioritisation software, Marxan, and Fiji and Micronesia as case study regions. Further, I examine whether there are any interaction effects occurring between these three factors in influencing where spatial priorities are determined. Finally, I assess how these prioritisation factors influence the extent to which coarse-resolution priorities are able to incidentally represent fine-resolution priorities. The results from this chapter have important implications for conservation planning strategies that rely on broad-scale, regional conservation assessments to guide subsequent assessments at smaller extents and finer-resolutions.

Chapter 3 addresses Objective 2 and is situated in the context of transitioning from conservation assessments formulated at broad extents to implementing conservation actions at local levels. Within this broader context of iterative planning, I specifically investigate how frequently regional priorities should be updated as local actions are implemented protractedly across the region, using Fiji as a case study. To do this, I use the programming language, *R*, and prioritisation software, Marxan, to simulate the process of iterative planning for the first time in the field of conservation planning. For these simulations, I designed specific parameters and rule sets to reflect the relevant decision-making steps that would be involved in applying conservation actions at local levels. For example, only a certain proportion of a prioritised planning unit would have conservation actions applied within it, to reflect the real-world constraint of spatial mismatch between planning units and management units. Other important rule sets incorporated were to apply conservation actions on the basis of the relative importance of targeted conservation features, and to ensure that applied actions in one planning iteration could not be spatially adjacent to each other. This last rule was included to emulate the common situation in which creating extensive contiguous reserves is not pragmatic and unlikely.

For **Chapters 4** and **5** which address Objectives 3 and 4, I investigate research questions to broaden our understanding of what it means to integrate conservation planning across multiple scales explicitly. To achieve this, I first explore the relative strengths and limitations of

conservation plans developed at different scales (**Chapter 4**), and use this understanding to help inform the theory and practice of conservation planning across multiple scales (**Chapter 5**). In **Chapter 4**, I identify the strengths and weaknesses of conservation plans developed at different jurisdictional levels in terms of adequately considering multiple social and ecological scales. To do this, I collate conservation plans developed at all levels (patch, local, regional, and international) across the CT region and evaluate each plan using an explicitly multiscale social-ecological systems framework. In **Chapter 5**, I explore the scope of multiscale planning in practice using Papua New Guinea and the Solomon Islands as case study regions. I first establish the extent to which multiscale planning is occurring in practice in these regions. I then evaluate each identified case study to determine any factors that impede or facilitate multiscale planning.

Finally, **Chapter 6** demonstrates in detail how each of my data-based chapters (**Chapters 2-5**) addresses the four research objectives of my thesis. In addition, I synthesise all my findings to demonstrate how my thesis contributes to filling the critical knowledge gaps related to problems of scale in the conservation planning process, highlighted in this chapter.

The data-based chapters of this thesis (**Chapters 2-5**) have been submitted for publication to international peer-reviewed journals. **Chapters 2** and **3** are published in *PLoS ONE*, and *Diversity and Distributions*, respectively. **Chapters 4** and **5** are currently in review, in *Ecology and Society*, and *Global Environmental Change*, respectively.

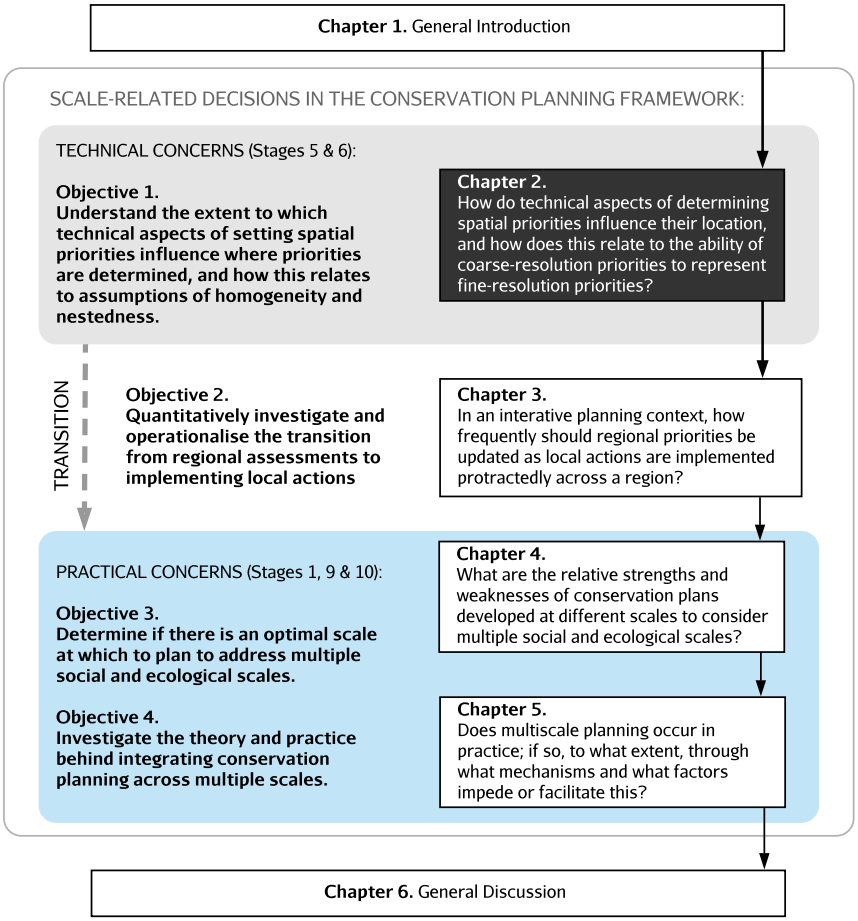
Chapter 2

Sympathy for the devil: detailing the effects of spatial prioritisation factors on priorities for marine conservation

2 Sympathy for the devil: detailing the effects of spatial prioritisation factors on priorities for marine conservation

In Chapter 2, I examine the individual and interacting effects of three spatial prioritisation factors on priorities for marine conservation: (1) planning-unit size, (2) thematic resolution of reef classes, and (3) spatial variability of socioeconomic costs. This chapter contributes to existing knowledge on the technical considerations of scale in conservation planning processes, outlined in Chapter 1: first, the significant influence of socioeconomic cost data on where conservation priorities are determined and its ability to interact with other data layers, and second, the implicit assumptions made when relying on coarse-resolution prioritisations to guide subsequent finer-resolution prioritisations. I conceptualised the research, curated and analysed the data, and wrote the chapter. Pressey and Weeks provided advice in conceptualising the research and assisted with analyses and structuring and editing the manuscript. Andréfouët provided the data and assisted with editing the manuscript. Moloney assisted with curating and processing the data and editing the manuscript.

As version of this chapter has been published as: Cheok, J., R. L. Pressey, R. Weeks, S. Andréfouët, and J. Moloney. 2016. Sympathy for the devil: detailing the effects of planning-unit size, thematic resolution of reef classes, and socioeconomic costs on spatial priorities for marine conservation. *PLoS ONE* 11(11):e0164869.



2 Sympathy for the devil: detailing the effects of spatial prioritisation factors on priorities for marine conservation

2.1 Abstract

Spatial data characteristics have the potential to influence various aspects of prioritising biodiversity areas for systematic conservation planning. There has been some exploration of the combined effects of size of planning units and level of classification of physical environments on the pattern and extent of priority areas. However, these data characteristics have yet to be explicitly investigated in terms of their interaction with different socioeconomic cost data during the spatial prioritisation process. I quantify the individual and interacting effects of three factors on spatial priorities for marine conservation, in typical marine planning exercises that use reef classification maps as a proxy for biodiversity: (1) planning-unit size, (2) thematic resolution of reef classes, and (3) spatial variability of socioeconomic costs. I assess these factors by creating 20 unique prioritisation scenarios involving combinations of different levels of each factor. Because output data from these scenarios are analogous to ecological data, I applied ecological statistics to determine spatial similarities between reserve designs. All three factors influenced prioritisations to different extents, with cost variability having the largest influence, followed by planning-unit size and thematic resolution of reef classes. The effect of thematic resolution on spatial design depended on the variability of cost data used. In terms of incidental representation of conservation objectives derived from finer-resolution data, scenarios prioritised with uniform cost outperformed those prioritised with variable cost. Following my analyses, I make recommendations to help maximise the spatial and cost efficiency and potential effectiveness of future marine conservation plans in similar planning scenarios. I recommend that planners: employ the smallest planning-unit size practical; invest in data at the highest possible resolution; and, when planning across regional extents with the intention of incidentally representing fine-resolution features, prioritise the whole region with uniform costs rather than using coarse-resolution data on variable costs.

2.2 Introduction

Conservation planning can be described in stages from stakeholder engagement through to the application and maintenance of conservation actions (Margules and Pressey 2000, Pressey and Bottrill 2009). Spatial prioritisations, a key stage in conservation planning processes, are important in guiding efficient investment of limited resources to design protected areas and off-

park interventions for conservation (Margules et al. 2002, Carwardine et al. 2008). Prioritising allows planners to quantitatively assess the importance of sites for conservation action, while also explicitly considering aspects of their socioeconomic context. Prioritisations are typically based on data on biodiversity and, more recently, socioeconomic costs, coupled with predefined quantitative objectives for environmental classes, species, or processes of interest (Rouget et al. 2006, Green et al. 2009). By 'environmental classes' I refer to spatial subdivisions of terrestrial, freshwater, or marine environments, based on physical, climatic, and/or biological variables, with the aim of deriving environmental surrogates (sensu Margules and Pressey 2000) for conservation planning. Once the planning region is subdivided into planning units and intersected with data on biodiversity and socioeconomic costs, conservation objectives are formulated and the accumulated information is analysed by decision-support tools that determine low- or least-cost conservation designs (Watson et al. 2011).

A number of studies, mostly terrestrial, have explored the influence of planning-unit size and thematic resolution of environmental classes on prioritisation (e.g., Pressey and Logan 1995, 1998, Rouget 2003, Payet et al. 2010; and see Richardson et al. 2006, Van Wynsberge et al. 2012, Hamel et al. 2013 for marine studies). While there is a growing body of evidence for the influence of aspects of resolution on prioritisation outputs, studies so far have mainly focused on the individual effects of spatial resolution of data, thematic resolution of environmental classes, and size of planning units, with few examining the effects of combinations of these factors. Importantly, no studies have yet considered how socioeconomic cost data can also interact with all these other factors to influence the selection of prioritised areas. With our increasing recognition of the importance of considering socioeconomic costs in prioritisations (Naidoo et al. 2006, Carwardine et al. 2008, Ban et al. 2009a, 2013), and the potential for conservation designs to guide conservation actions, it is necessary to investigate the potential interactions that can occur between planning-unit size, thematic resolution of environmental classes, and spatial variability of cost data.

As discussed in Chapter 1, another question related to spatial scale that remains unexplored is the degree to which conservation priorities defined at fine resolutions are likely to be nested within priorities defined with coarse prioritisations. Despite the shortcomings associated with coarse prioritisations (Pressey and Logan 1998, Rouget 2003, Richardson et al. 2006, Payet et al. 2010), including the likely invalid assumptions of homogeneity and spatial nestedness, coarse priorities continue to be produced because of the desire for broad views of conservation priorities (Poiani et al. 2000, Rouget et al. 2006, Klein et al. 2009, Selig et al. 2014, Beger et al. 2015). The two studies, to my knowledge, that have investigated the assumption of nestedness have found that coarse assessments can represent many finer-resolution priorities, except in heterogeneous or

fragmented areas (Pressey and Logan 1995, 1998, Rouget 2003, Payet et al. 2010). However, key gaps apparent from these studies are that the findings were not consistent (fine-resolution priorities were represented to varying extents), and data were for terrestrial environments only. Moreover, neither of these studies considered the role of socioeconomic cost data, and how these could influence the nestedness of fine-resolution priorities within coarse ones.

With so many aspects of data interacting in the prioritisation process across multiple extents or resolutions, the devil is in the detail. Important details that remain to be explored are the potential interactions between three prioritisation factors: size of planning units, thematic resolution of environmental classes, and spatial variability in socioeconomic costs, and how these together influence prioritisation outputs, particularly in terms of the spatial configuration of priorities when different cost layers are used. These interactions likely have important implications for identifying effective prioritisation strategies. Here, I investigate the interacting effects of these three factors on conservation prioritisations in marine planning contexts, where environmental classes are used commonly as proxies for biodiversity. Specifically, I use geomorphological reef classes (hereafter, 'reef classes'). I assess the relative effect of each of these factors and, for the first time to my knowledge, interactions occurring between all three factors, on: (1) the total extent and cost of reserve solutions, and (2) spatial configurations of priority areas. I also assess the ability of coarse prioritisations to adequately represent finer-resolution priorities, in terms of: (1) the spatial nestedness of priorities determined at different resolutions, and (2) the extent of incidental representation of reef classes at high thematic resolution by coarse-resolution priorities. Using case studies from marine environments provides insights to support the rapidly increasing number of marine protected areas in response to targets under the Convention on Biological Diversity (CBD; Edgar et al. 2014). In doing so, I help to understand the devil in the detail of marine conservation prioritisation.

2.3 Methods

2.3.1 Study regions

Two regions were used as case studies: Fiji and Micronesia (consisting of the Mariana Islands, Marshall Islands, Palau, Guam, and the Federated States of Micronesia; Figure 2.1). The individual Micronesian nations were considered as one region for the purposes of my analyses to provide a large, regional extent. The total extent of the planning regions for Fiji and Micronesia were approximately 24,439 km^2 and 32,168 km^2, respectively. The focus of this study was on all coral reefs contained within each country's exclusive economic zone. Aspects of planning-unit size and thematic resolution of reef classes (hereafter, 'thematic resolution') are especially relevant in archipelagic and developing nations, where there is a pressing need for conservation

action but the mismatch between regional-level planning and local-level implementation is large (Weeks et al. 2010b). This disparity between regional and local perspectives is primarily attributed to the often coarsely-defined prioritisations (Roberts et al. 2002) and the often fine (devolved) spatial resolutions of governance in these developing regions, with complex social, economic, and political factors shaping conservation decisions (Weeks et al. 2010b, Mills et al. 2010, Hamel et al. 2013, Horigue et al. 2015).

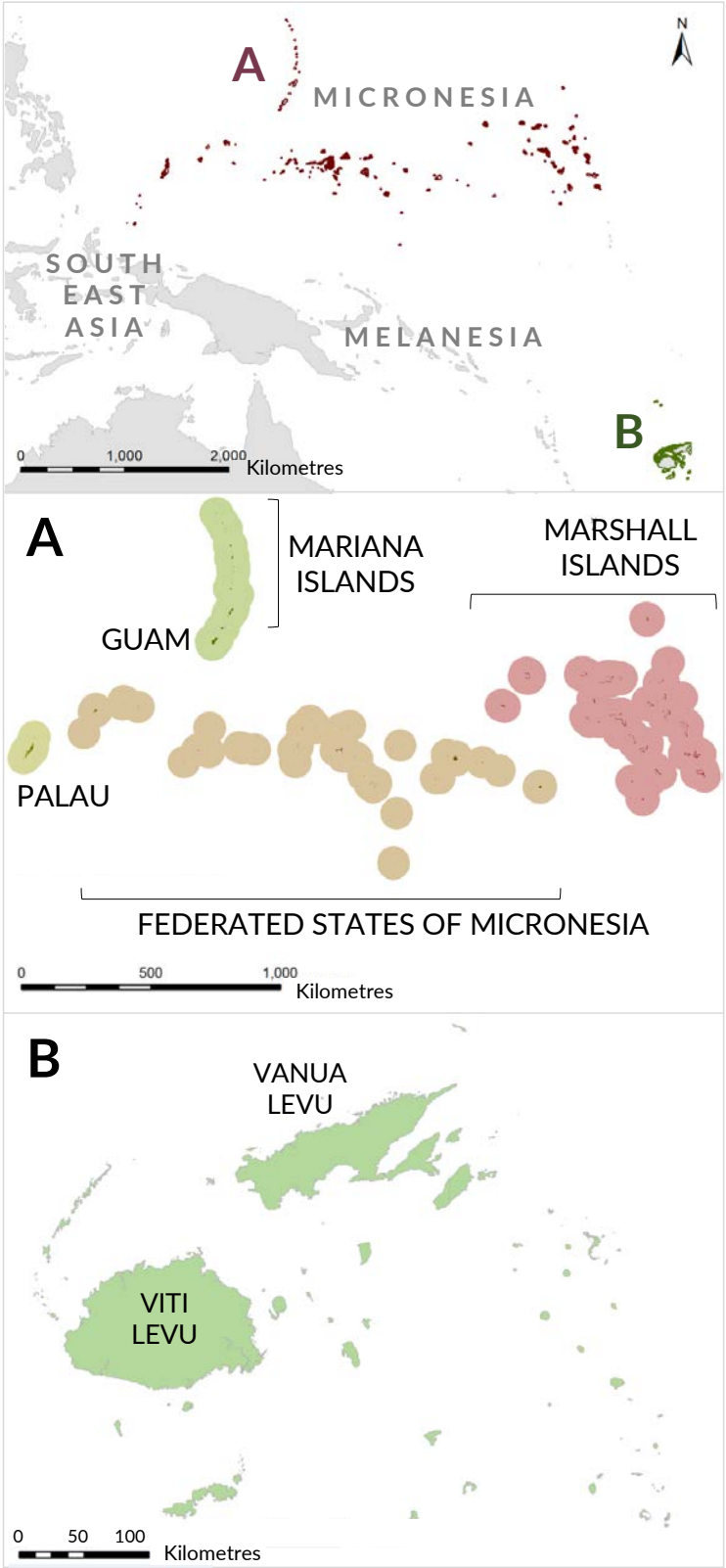


Figure 2.1 Regional context and enlarged maps of the two study regions: (A) Micronesia and (B) Fiji. Buffers are shown around Micronesian nations to increase visibility of the numerous small coral islands and atolls.

Though my analyses are grounded in real data, they are demonstration exercises, not intended to inform real-world conservation action in these study regions. For this reason, I did not consider existing marine protected areas in Fiji or Micronesia. The benefits of using empirical rather than modelled data (in the case of the reef-class maps) are that the results from this study will be a more realistic representation of outcomes expected in real-world applications.

2.3.2 Study design

I examined three prioritisation factors: planning-unit size (analysed at two levels), thematic resolution (five levels), and spatial variability of socioeconomic costs (two levels). A full factorial design (giving a total of 20 unique prioritisation scenarios; Figure 2.2) was employed to determine the influence of each of these factors in combination with the others. I used a scenario coding system (Table 2.1) to facilitate interpretation of subsequent results.

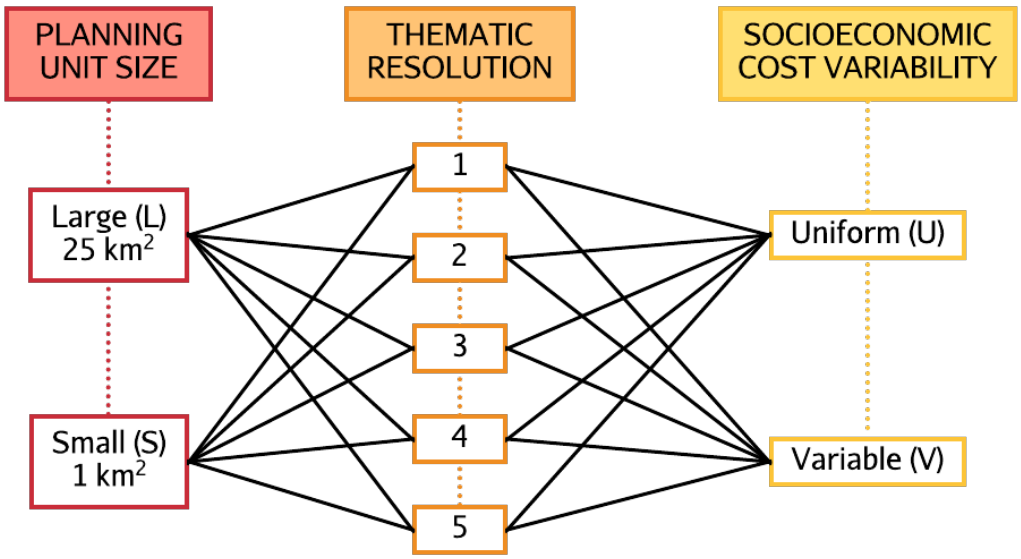


Figure 2.2 Study design showing tested factors, factor levels, and the 20 unique combinations between all levels.

2.3.3 Planning-unit size

Two planning-unit sizes were explored: 1 km^2 ('small') and 25 km^2 ('large'). These totalled 29,781 small and 1,845 large planning units for Fiji, and 38,019 small and 2,339 large planning units for Micronesia. Square planning units were used so that the smaller planning units spatially nested within the larger ones (Figure 2.3A). The sizes were determined by a review of marine prioritisation exercises to realistically gauge 'small' and 'large' planning-unit sizes, relative to real-world contexts. Planning units were complete squares, except where they occurred on the

edges of reefs. Edge planning units were trimmed to reef edges so that the area of each planning unit was equal to the extent of the reef it contained. This was important for calculating uniform cost from reef area for each planning unit (see Section 2.3.5 below).

Table 2.1 Coding system to identify individual scenarios. Codes are assigned to each level of each prioritisation factor. An example scenario code based on this coding system is 'L1U': large planning units (L), first level of thematic resolution (1), and uniform cost (U).

	Prioritisation-factor level	Level code
Planning-unit size	Large	L
	Small	S
Thematic resolution	1 (coarse)	1
	2	2
	3	3
	4	4
	5 (fine)	5
Variability of socioeconomic cost	Uniform	U
	Variable	V

2.3.4 Thematic resolution

Reef classes of Fiji and Micronesia were mapped using high spatial resolution Landsat 7 ETM + satellite images (30 metres) as part of the Millennium Coral Reef Mapping Project (Andréfouët et al. 2006). However, a mapped minimum discernible unit is much larger, around 10,000 m^2 (or a cluster of about 10 pixels). The reef classes were mapped separately for each Micronesian country, with some reef classes in common between countries. For my analyses, I considered each reef class from each country as unique, regardless of nominal thematic overlap with the other Micronesian countries. In other words, an atoll in Palau is considered here as a different reef class than an atoll in Marshall Islands or FSM. Practically, this means that prioritised areas were forced to be spread between all the different Micronesian countries. Doing so was necessary to realistically reflect the individual objectives that the separate countries would have in such transnational-scale planning exercises, while still providing a large regional extent for the prioritisation scenarios. Distinguishing reef classes between countries was also precautionary for the use of reef classes as biodiversity surrogates. Differences in species associated with the same reef class in different countries are likely to arise from dissimilar reef complexities between the different Micronesia regions, and the distance decay of similarity in ecological communities (Soininen et al. 2007).

The maps describe reef geomorphology in a hierarchical classification scheme with five thematic resolutions, from levels 1 to 5, with level 1 referring to the lowest resolution (2 reef classes in Fiji; 4 reef classes in Micronesia), and level 5 to the highest resolution (280 reef classes in Fiji; 181 reef classes in Micronesia). The 4 and 181 reef classes for level 1 and level 5, respectively, in Micronesia reflect subdivision of reef classes by country. Example maps are shown in Figure 2.3B,C.

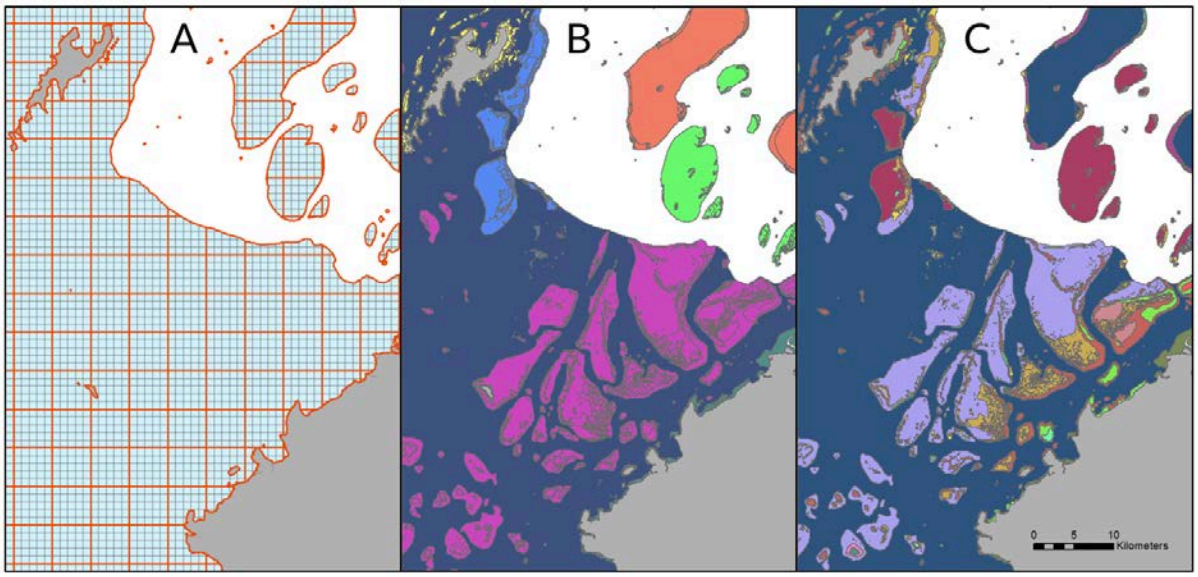


Figure 2.3 Example maps of planning-unit sizes and thematic resolutions explored in the Fiji dataset. All maps represent the same spatial extent and location; grey polygons represent Fiji terrestrial areas (islands). (A) Planning-unit sizes: 'small' (1 km^2; blue squares) and 'large' (25 km^2; red squares); 25 small planning units are nested within each large, non-edge planning unit. Note that both planning-unit grids were clipped to all reef areas, resulting in irregular planning units on the perimeters. (B) and (C) Examples of two of the five levels of thematic resolution: (B) level 2 (11 reef classes total), and (C) level 4 (43 reef classes total).

2.3.5 Socioeconomic cost data

The two layers of socioeconomic opportunity cost (Naidoo et al. 2006) used in my analyses were: spatially uniform, with planning-unit cost proportional to area (i.e., amount of reef contained within); or spatially variable, modelled to represent a proxy of opportunity cost to fishers (Figure 2.4). Since opportunity cost data did not exist for the whole of either study region, I created spatially variable cost layers for both regions using weighted linear distance from fisher populations (derived from census information; Economic Policy Planning and Statistics Office 1999, Office of Planning and Statistics 2005, Fiji Bureau of Statistics 2007, CNMI Department of Commerce 2010, Federated States of Micronesia Division of Statistics 2010, U.S. Census Bureau 2010) to the furthest reef areas (details in Appendix 1 Text A1.1). Distance measures have

commonly been used as a proxy for socioeconomic cost in previous studies (Naidoo et al. 2006, Weeks et al. 2010c), although I acknowledge that they are limited predictors of actual opportunity costs to fishers (Weeks et al. 2010c). However, my aim here was to contrast two different cost layers, not to guide conservation planning for implementation.

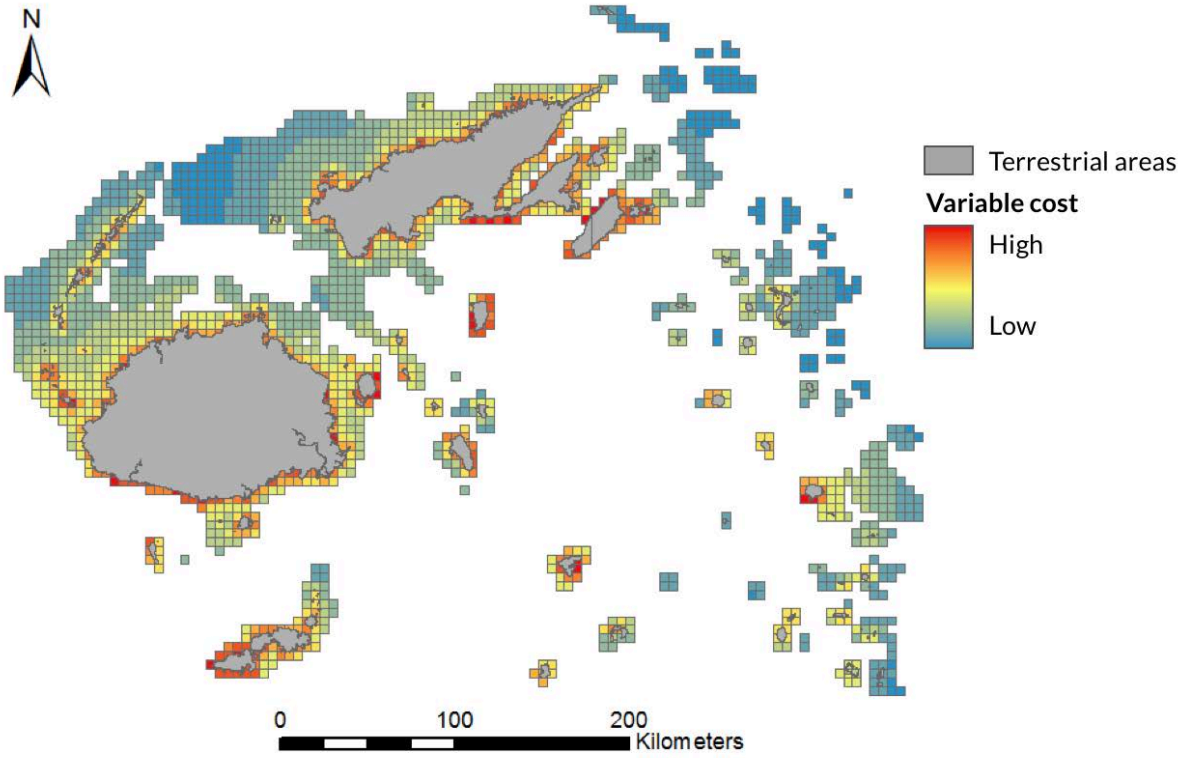


Figure 2.4 Example map showing distribution of cost variability across the Fiji planning region. Values, shown here for large planning units, are based on distance to fisher populations as a proxy for opportunity cost.

2.3.6 Priority-setting tool parameters and calibrations

Across all scenarios, the conservation objective was to protect 30% of each reef class (both Fiji and Micronesia have made commitments to protect 30% of their inshore waters; Adams et al. 2011, The Micronesia Challenge 2012). To identify sets of planning units that achieved this objective, I used the decision-support software Marxan (Ball et al. 2009). Marxan uses a simulated annealing algorithm with iterative improvement to find spatial reserve designs that meet biodiversity objectives for the least socioeconomic cost.

For each scenario, I ran Marxan to produce 100 solutions (i.e., giving a total of 100 replicates). With 20 scenarios, this produced a total of 2000 individual solutions for comparison. Marxan also produces a 'selection frequency' output, which records the number of times each planning

unit was selected across multiple solutions. Planning units with higher selection frequency are more likely to be needed to achieve a set of conservation objectives. The 'Species Penalty Factors' in Marxan were calibrated for each scenario so that all solutions achieved the overall objective of 30% of each reef class (minimum proportion met > 0.999), following the Marxan Good Practices Handbook (Ardron et al. 2010).

2.3.7 Output comparison and statistical analyses

Multiple output comparisons and analyses were performed to achieve each aim of this study (Table 2.2). All statistical analyses were performed using the statistical software package, *R* (R Core Team 2014). To compare configurations of solutions from scenarios with different planning-unit sizes, each large planning unit was converted to its component small planning units (maximum number, 25). In the case of selection frequencies, the smaller parts of large planning units were given the same values as their larger planning unit; so if a large planning unit had a selection frequency of 50, each of the (up to) 25 smaller planning units nested within it also had a value of 50. For comparison of individual solution outputs, if a large planning unit was selected, all of the smaller planning units nested within it were also considered selected.

Table 2.2 Summary of output comparisons and statistical analyses for each research aim.

Research aims	Output comparison / statistical analyses
1) Assess relative effect of each factor	• Total reserve extent and cost of solutions
2) Assess interactions between factors	• Spatial configuration of priority areas
3) Assess the ability of coarse prioritisations to represent finer-resolution priorities	• Spatial nestedness of priorities at different resolutions • Extent of incidental representation of fine thematic-resolution objectives by coarse-resolution priorities

Research aims 1 & 2: total reserve extent and cost of solutions

There are two fundamental metrics from Marxan reserve solutions: the overall extent of the reserve solution and total cost of the selected areas. These metrics reflect the overall spatial and cost efficiency of the reserve solutions. I compared these two metrics between all scenarios. Costs were compared as proportions of maximum possible cost (of the whole planning region) to allow direct comparisons of scenarios using different cost layers.

Research aims 1& 2: spatial configuration of priority areas

Output data from Marxan are analogous to ecological data (Figure 2.5). Therefore, all individual scenario solutions (2000 in total, 100 per scenario) were compiled into a data matrix for statistical analyses with prioritisation scenarios or 'sites' as rows and planning units or 'species' as columns. Another data matrix was created for all selection frequency outputs (20 in total, one per scenario). Another analogy between Marxan output data and community data is that the matrix contains many zero entries for 'species' (or planning units). The data matrices were therefore Hellinger-transformed, which allowed meaningful use of parametric ordination methods (which are Euclidean-based), while circumventing the problems associated with Euclidean distance to analyse matrices with many zeros (see Legendre and Gallagher 2001 for details). I used parametric ordination methods because of the higher level of statistical power possible with such tests.

"Sample species"

	Planning unit ID									
Prioritisation scenario	1	2	3	4	5	6	7	8	9	...
Solution 1	0	0	1	1	0	1	0	0	0	0
Solution 2	1	0	1	0	0	0	1	1	0	1
Solution 3	1	1	1	0	0	0	1	0	1	0
...										

"Sample site"

"Species presence/absence"

Scenario 1 selection frequency	39	37	41	33	19	20	36	50	35	40
Scenario 2 selection frequency	41	22	43	25	25	32	47	18	34	25
...										

"Species abundance"

Figure 2.5 Example of Marxan output data. For calculation of dissimilarity, I regarded Marxan individual solutions as analogous to biological sampling sites and planning units as analogous to recorded species. For single Marxan solutions, planning units were either selected ('present') or unselected ('absent'). For selection frequencies across 100 replicate solutions in a scenario, entries for planning units were equivalent to species abundance data.

Once the data matrices were transformed, a Euclidean dissimilarity matrix was calculated ('vegan' *R* package; Oksanen et al. 2015). This involved measuring, pair-wise, the dissimilarities (or distance) between all 2000 solutions (presence-absence) or all 20 solutions (selection frequencies). From these dissimilarity matrices, I used an average-linkage hierarchical cluster analysis to determine, through visual interpretation, the spatial similarity between solutions and whether any of the tested factors (i.e., planning-unit size, thematic resolution, and variability of

costs) appeared to influence similarity ('sparcl' *R* package for plotting; Witten and Tibshirani 2013). The cluster analysis on the selection frequency outputs allowed me to check whether the clusters derived from individual solutions were affected by spatially idiosyncratic solutions.

The dissimilarity matrix was also used to conduct a redundancy analysis (RDA), or constrained ordination ('vegan' *R* package), to quantify the extent to which the tested factors explained the clusters, since this information is absent from the cluster analysis itself. I used an RDA because it focuses only on the variation that can be explained by the 'environmental' variables (in this case, the prioritisation factors: planning-unit size, thematic resolution, and cost variability). To test whether the results from the ordination model were statistically significant, I ran a permutation test on the RDA ('vegan' *R* package).

Research aim 3: spatial nestedness of priorities

I determined the degree to which fine-resolution priorities (small planning units, high thematic resolution) were spatially nested within all coarse-resolution priorities (large planning units), some of which were based on mapping at coarse thematic resolutions. I achieved this by comparing priorities produced from each scenario with large planning units (10 scenarios in total) with two of the finest-resolution prioritisation scenarios: S5V (small planning units, thematic resolution level 5, variable costs) and S5U (small planning units, thematic resolution level 5, uniform costs) (Figure 2.6). Since the research aim here was to ascertain the ability of coarse-resolution priorities produced with large planning units to spatially capture fine-resolution priorities produced with small planning units, I examined the two prioritisation factors of potential influence: thematic resolution and cost variability. My scenarios with large planning units were intended to reflect, to varying degrees, the coarse prioritisations usually necessary over large extents. My two 'test' scenarios with small planning units were intended to reflect the higher-quality data usually available only after closer study of relatively localised parts of large planning regions (except that my high-quality data covered whole planning regions in this instance).

To assess the extent of spatial nestedness, I examined the amount of spatial overlap between high-priority areas defined at two levels: planning units with selection frequencies of 50 and 75 (Figure 2.6). Extent of spatial nestedness was indicated by the percentage of small high-priority planning units from each of the two test scenarios that overlapped with high-priority large planning units from the ten scenarios being tested.

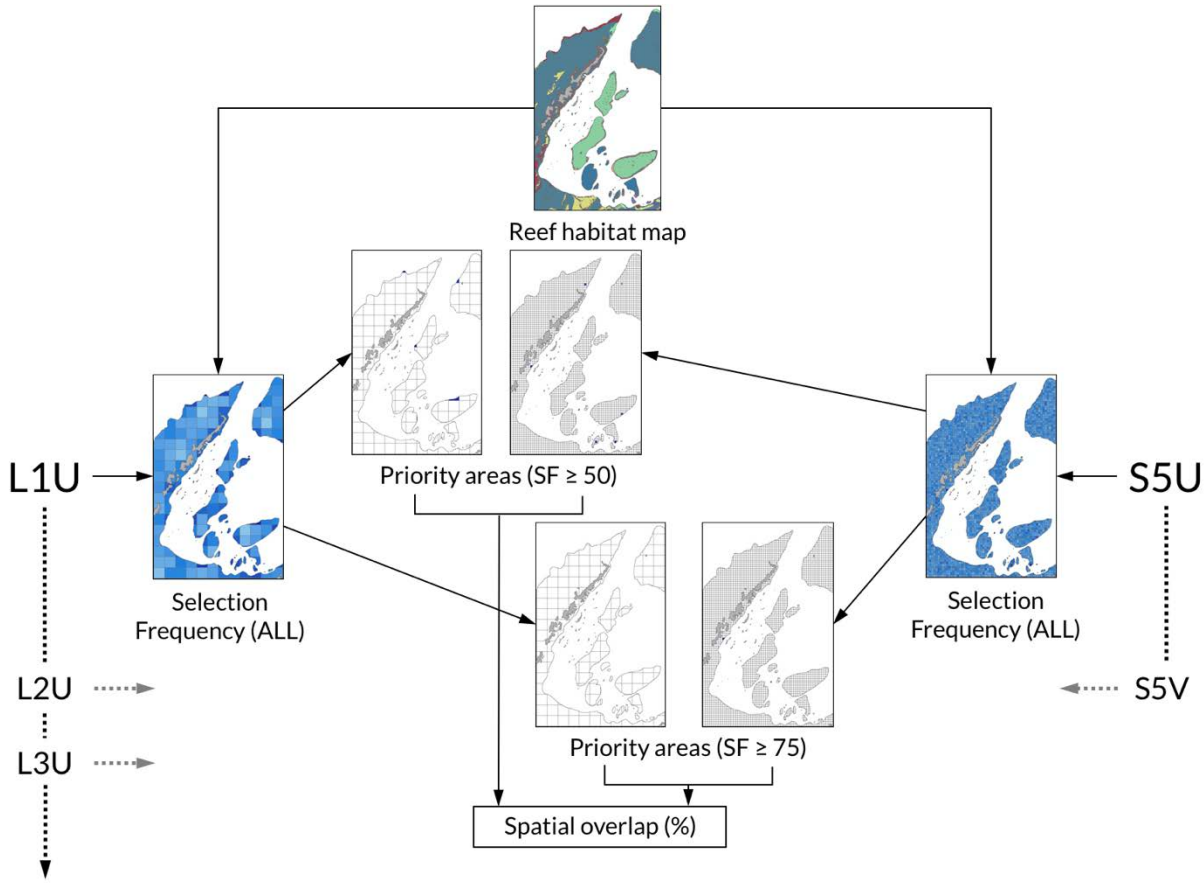


Figure 2.6 The flow of analyses related to spatial nestedness. Analyses described for spatial nestedness, defined here as extent of overlap of high-priority areas based on large planning units (left) with small planning units with high thematic resolution and both cost layers (right). Dotted arrows indicate repetition of the same analyses across all other coarse scenarios performed with both of the test scenarios, S5U and S5V.

Research aim 3: incidental representation of fine thematic-resolution objectives by coarse-resolution priorities

To complement the analyses of nestedness, I wanted to know the degree to which the conservation objective (i.e., 30% of each reef class) for the highest thematic resolution (level 5) would be incidentally achieved by priority areas identified by the ten coarse scenarios using large planning units. The planning-unit selection frequencies for all scenarios using large planning units were converted to probabilities of being selected (see Table 2.3 for example; method modified from Lombard et al. 2003). When summed across all planning units, this method gave the expected areas of each level 5 reef class that would be represented by scenarios based on large planning units. I then plotted the relationship between incidental representation of level 5 reef classes and their rarity in my study regions, with rarity determined by the extent of each reef class relative to that of the whole study area, expressed as a percentage: [1 – (total reef class

extent / total planning extent)] × 100. This formula gave large percentages to very restricted reef classes and smaller values to more extensive reef classes.

Table 2.3 Method calculating the expected areas of level 5 reef classes that would be selected for reservation in each of the scenarios using large planning units.

Planning unit ID (PUID)	Selection frequency	Probability of selection (*P*)	Level 5 reef-class code	Reef-class area (km^2) (*A*)	Expected area (km^2) of level 5 code selected by PUID (*P**A*)
1	45	0.45	1	0.395	A1,1 = 0.178
1	45	0.45	5	0.375	A1,5 = 0.169
1	45	0.45	7	0.230	A1,7 = 0.104
2	33	0.33	2	0.012	A2,2 = 0.004
2	33	0.33	7	0.988	A2,7 = 0.326
3	21	0.21	7	0.132	A3,7 = 0.028
3	21	0.21	9	0.868	A3,9 = 0.182
…					

2.4 Results

All analyses indicated similar influences and interactions of factors for both regions (Micronesia and Fiji). Therefore, only results for the Fiji case study are presented here. Results for the Micronesia case study are presented in Appendix 1 Figures A1.2-6.

2.4.1 Individual effects of prioritisation factors

Total reserve extent and cost of solutions

I found a general trend of increasing total selected reserve extent with increasing thematic resolution, regardless of planning-unit size or variability of cost (Figure 2.7a). However, there was an obvious difference in the rate of increase in total reserve size between the different planning-unit sizes as thematic resolution increased. The rate of increase was markedly higher with larger planning units.

The total costs of solutions, expressed as proportion of maximum possible cost (both uniform and variable), generally increased with thematic resolution (Figure 2.7b). Overall, total reserve costs were substantially lower in all scenarios with variable cost. For scenarios with variable cost, the only exception to the general increasing trend in cost with increasing thematic resolution was

at level 4, for which total reserve cost decreased (i.e., scenarios L4V and S4V). This trend was less apparent in the Micronesia results (Figures A1.2).

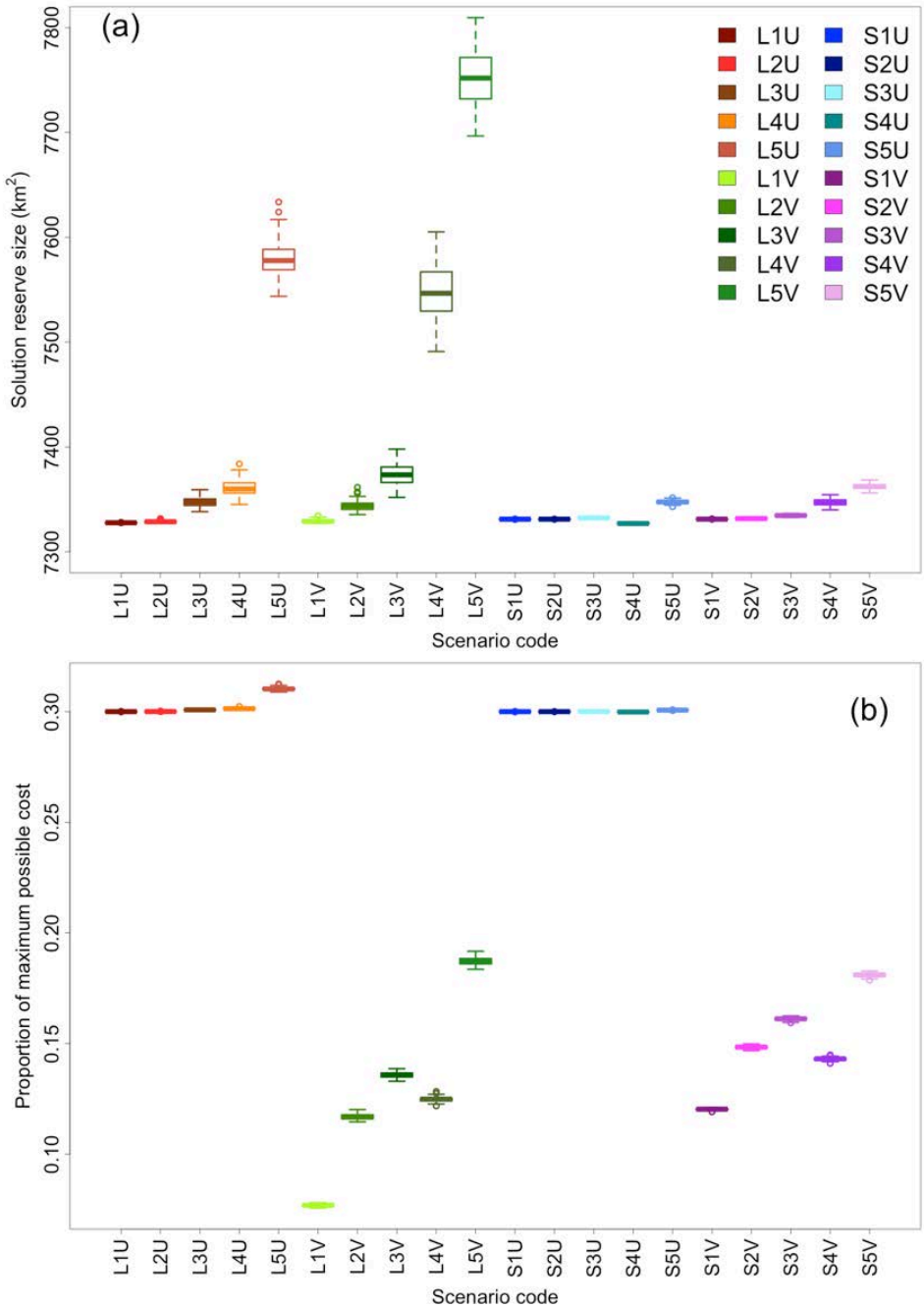


Figure 2.7 Comparisons of total reserve size and proportions of maximum possible cost. (a) Boxplots of ranges of reserve solution sizes for each scenario based on 100 replicate runs. (b) Boxplots of ranges of total costs (expressed as proportions of maximum possible cost) for each scenario based on 100 replicate runs. Each change in shade of the same colour represents the change in thematic resolution (always presented in order from level 1–5, left to right) for each combination of planning-unit size and cost variability. Colour scheme representing all scenarios remains the same throughout all figures to facilitate interpretation.

Spatial configuration of priority areas

The 2000 individual solutions were clustered mainly by cost variability and planning-unit sizes (Figure 2.8), confirming that these factors were important in determining spatial configurations. Distinct clusters first formed (lowest dissimilarity) between scenarios with variable cost, forming separate groups of clusters primarily between large and small planning units (green and purple clusters, Figure 2.8). Variability of cost data appeared to have the largest effect on spatial dissimilarity between solutions, based on the dissimilarity distances and the distinct difference in clustering of solutions with and without variable cost.

The two RDA axes explained 87.71% of the spatial variance due to the three prioritisation factors tested (Figure 2.9). Axis RDA1 represented most of the variation due to variability in cost, evidenced by the wide horizontal separation between scenarios prioritised with uniform costs (left) and variable costs (right). A marked difference between scenarios with uniform cost and those with variable cost is the clear separation of variable-cost scenarios according to thematic resolution (Figure 2.9), reflecting the clustering in the hierarchical analysis. Axis RDA2 mainly represented the variation due to planning-unit size, with large planning-unit scenarios in the lower part of the plot and small planning-unit scenarios in the upper part (Figure 2.9). Another observable difference between large and small planning-unit sizes is the amount of spatial dissimilarity between solutions for the same scenarios. Larger planning units (lower) produced individual solutions for combinations of cost variability and thematic resolution that were more loosely clustered, and therefore less similar in configuration, than the same combinations with small planning units (upper). There was no clear association between the spatial variance caused by thematic resolution with either of the plotted axes. The plotted centroids (red squares; Figure 2.9) of each prioritisation-factor level in the biplot indicate the average amount of spatial variance, relative to both axes, predicted for all solutions of the same prioritisation-factor level. The locations of the centroids for each thematic-resolution level indicate the relatively small overall influence of thematic resolution on spatial differentiation between solutions (Figure 2.9). The modelled influences of the three factors on spatial dissimilarity of all solutions from the ordination analysis were statistically significant (Table 2.4).

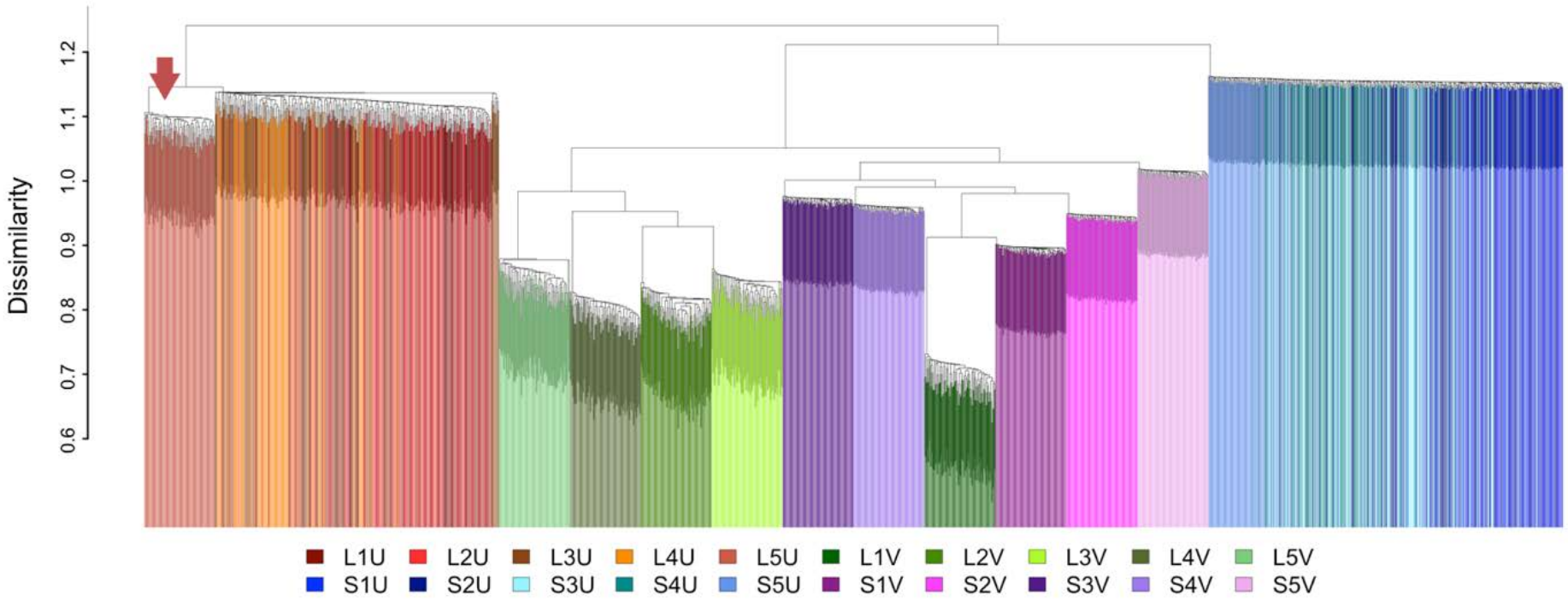


Figure 2.8 Spatial dissimilarity between all 2000 solutions for the Fiji case study. Red arrow indicates difference in spatial differentiation between solutions prioritised with the highest thematic resolution, level 5, and all other levels (1-4), for scenarios prioritised with large planning units and uniform cost.

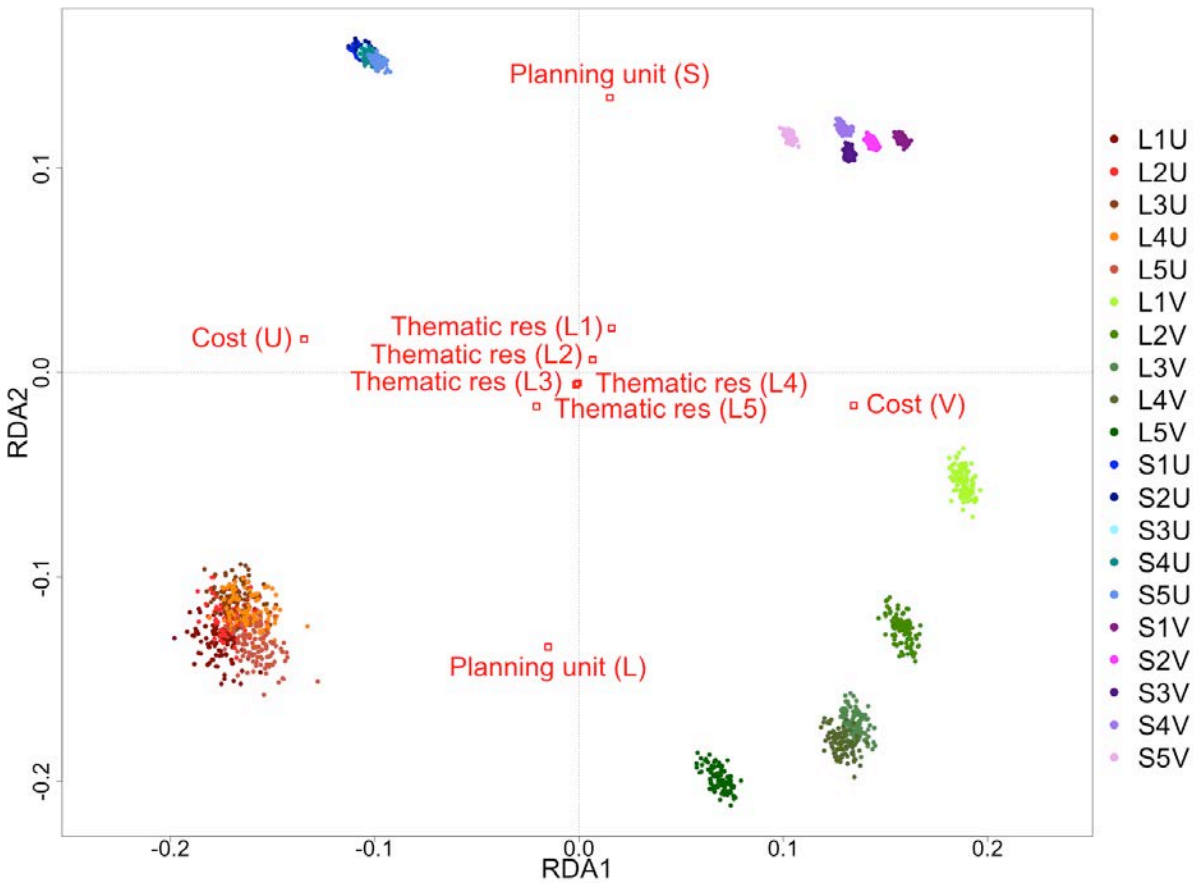


Figure 2.9 Comparison of spatial variation between all solutions produced using RDA for the Fiji case study. Cost variability mainly explains variation along RDA1, while variation along RDA2 is mostly represented by different planning-unit sizes. Red squares are centroids of the different prioritisation-factor levels, representing the average amount of spatial variance that lines up with the plotted axes.

Table 2.4 Permutation test results showing significance of each of the tested prioritisation factors in influencing the spatial dissimilarity of solutions.

	df	Variance	*F*-statistic	p-value
Cost variability	1	0.0742	260.879	<0.001
Planning-unit size	1	0.0267	93.815	<0.001
Thematic resolution	4	0.0149	13.126	<0.001

2.4.2 Interaction effects between prioritisation factors

Total reserve extent and cost of solutions

The increase in reserve size with increasing thematic resolution appeared to be amplified for scenarios involving variable cost (Figure 2.7a). Similarly, scenarios with variable cost involved

notable increases in total cost with increasing thematic resolution, while there was relatively little effect of planning-unit size or thematic resolution on the total cost of reserves for scenarios with uniform cost (Figure 2.7b).

Spatial configuration of priority areas

In the clusters of variable-cost scenarios formed in the hierarchical cluster analysis (Figure 2.8), differentiation between the different levels of thematic resolution was clear, evidenced by the largely separate clusters forming between the shades of green and purple. In other words, all the solutions for a given combination of variable cost and thematic resolution were clustered together, with very little or no overlap with the other variable-cost scenarios. The next major cluster contained solutions with uniform cost and small planning units (blue cluster, Figure 2.8). Here, there was much less spatial differentiation between solutions for different thematic resolutions, evidenced by the mixing of blue shades within the cluster. The last major cluster to form and the most spatially distinct from all other solutions, contained solutions with uniform cost and large planning units (red cluster, Figure 2.8). Similar to the scenarios with uniform cost and small planning units, there was no spatial differentiation between thematic resolutions with the exception of level 5 (red arrow; Figure 2.8). In summary, the distinctiveness of solutions for different thematic resolutions depended on whether uniform or variable costs were used.

Within the broad trend of spatial differentiation produced between solutions with uniform cost (left) and variable cost (right) in the RDA analysis, distinct clusters formed from solutions based on variable costs, composed of different levels of thematic resolution (green and purple points; Figure 2.9), as observed in the cluster analysis (Figure 2.8). Also akin to the cluster analysis, solutions produced by uniform-cost scenarios, to the left of the biplot, were not distinguished by thematic resolution (red and blue points). This is apparent from the grouping of solutions with small (blue) and large (red) planning units, regardless of thematic resolution.

2.4.3 Ability of coarse prioritisations to represent finer-resolution priorities

Spatial nestedness of S5U priorities

Spatial nestedness of S5U selected areas (calculated as percentage of S5U high-priority areas within large high-priority planning units) appeared to be influenced mainly by matched thematic resolution (Figure 2.10a,b). Nestedness of S5U areas was highest within large planning units selected to represent level 5 reef classes, for both selection frequency 50 (45–48%; Figure 2.10a) and selection frequency 75 (84–100%; Figure 2.10b).

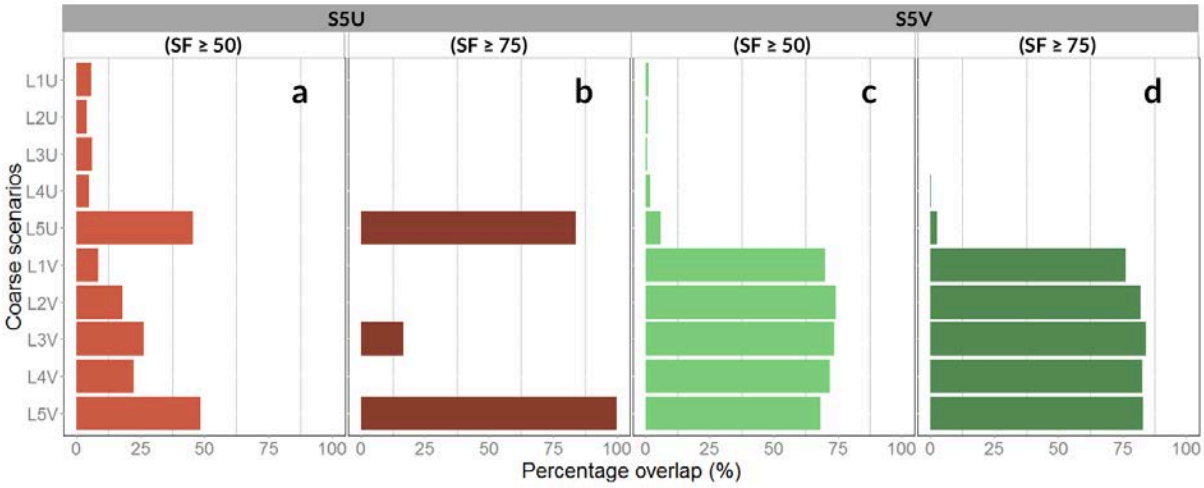


Figure 2.10 Nestedness of high-priority small planning units (test scenarios) within high-priority areas defined by large planning units. Nestedness of S5U high-priority areas, defined at: (a) selection frequency 50, and (b) selection frequency 75. Nestedness of S5V high-priority areas, defined at: (c) selection frequency 50, and (d) and selection frequency 75.

Spatial nestedness of S5V priorities

Spatial nestedness of S5V selected areas (calculated as above) was strongly influenced by variable cost (Figure 2.10c,d). Nestedness within coarse scenarios that used variable cost was high: 68–74% (high priority = selection frequency 50; Figure 2.10c) and 76–84% (high priority = selection frequency 75; Figure 2.10d). Nestedness within coarse scenarios that used uniform cost was much lower: 0.7–6% (high priority = selection frequency 50) and 0–3% (high priority = selection frequency 75). Unlike the spatial nestedness of S5U priorities, thematic resolution did not appear to influence nestedness of S5V priorities.

Incidental representation of fine thematic-resolution objectives by coarse planning-unit scenarios with uniform cost

Uniform-cost scenarios based on thematic resolution levels 1–4 had incidental representation of level 5 reef classes that trended close to the 30% objective across rarity values (Figure 2.11a), with variation above and below that value for individual reef classes (and see Figure 2.11c). For the scenario based on level 5, over-achievement of the 30% objective increased with reef class rarity (Figure 2.11a), reaching as high as 100% for some very rare reef classes, with virtually no under-achievement (Figure 2.11c).

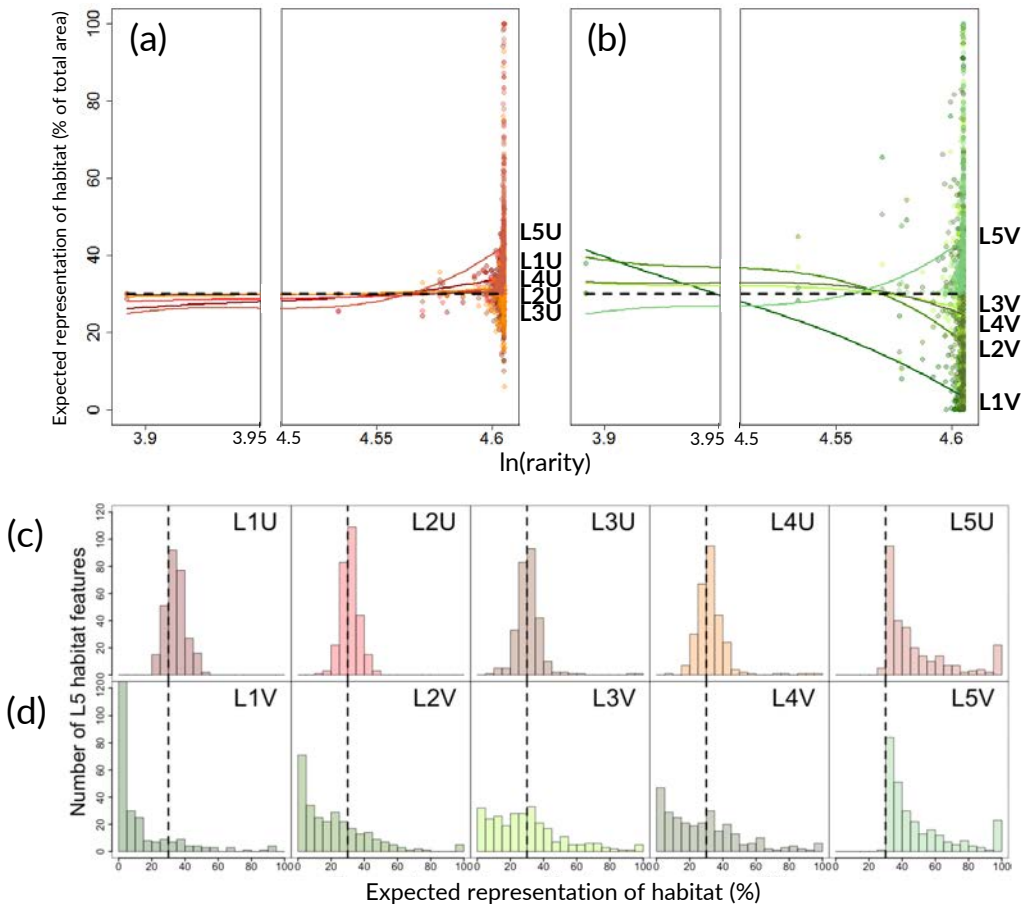


Figure 2.11 Incidental representation of level 5 reef classes by scenarios using large planning units. (a-b) Scatter plots showing expected representation of each level 5 reef class (as a percentage of total area of reef class occurrence) for each coarse scenario with (a) uniform cost and (b) variable cost, in relation to reef class rarity (transformed to natural log). Due to spread and left-skewness of rarity values, plots are shown with *x*-axis breaks where no data occur to facilitate interpretation. Local regression (LOESS) curves were fitted for each coarse scenario, indicating non-linear trends in each scatter plot. Dashed horizontal lines represent the 30% objective for level 5 reef classes. (c-d) Histograms showing the distributions of expected representation of level 5 classes for coarse scenarios with (c) uniform cost and (d) variable cost, plotted with 5% bin widths. Dashed vertical lines represent the 30% objective for level 5 classes.

Incidental representation of fine thematic-resolution objectives by coarse planning-unit scenarios with variable cost

Variable-cost scenarios based on thematic resolution levels 1–4 achieved more variable incidental representation of level 5 reef classes and much lower achievement of the overall objective than scenarios with uniform cost (Figure 2.11b,d). The level 1–4 scenarios with variable cost produced much lower values for rarer reef classes, frequently achieving no incidental representation, and there was more variation around the fitted curves than those based on uniform cost (Figure 2.11b). For the coarse scenario based on level 5 reef classes, there was the same increase in over-

achievement of objectives as for uniform cost (Figure 2.11b), also reaching 100% at high rarity, and the same lack of underachievement (Figure 2.11d).

2.5 Discussion

I sought to understand the detail underlying the influence of three factors – planning-unit size, thematic resolution of reef classes, and spatial variability of cost data – on the outputs from conservation prioritisation. My findings have implications for future marine prioritisations, particularly those that use extensive, coarse-resolution assessments as guides for conservation actions at finer resolutions.

2.5.1 Individual effects of prioritisation factors

Variability of cost data had the largest influence on planning outputs. Scenarios produced with variable cost data produced solutions that were less costly relative to all scenarios with uniform cost (in terms of the calculated proportions of maximum possible cost). This result accords with those of previous studies (e.g., Juutinen et al. 2004, Richardson et al. 2006, Weeks et al. 2010c). Spatially variable costs underlie a consistent spatial bias of selection toward cheaper planning units, where there are choices to achieve objectives.

Cost variability also had the strongest influence on the spatial configuration of solutions. It is known that the costs used in a decision-support tool such as Marxan have a large influence on the selection of planning units, and that different cost data can lead to dissimilar patterns of priorities (Ban et al. 2009b). Of the studies that have explicitly compared the spatial differences in priorities determined with different socioeconomic cost layers, all observed an influence of cost (Klein et al. 2008, Adams et al. 2010, Weeks et al. 2010c, Delavenne et al. 2012, Mazor et al. 2014, Schröter et al. 2014). My analyses further demonstrate that variable cost data had a greater overall influence on configuration of priority areas than planning-unit size or thematic resolution. This result reaffirms the importance of selecting appropriate cost metric(s) for identifying conservation priorities.

Larger planning units substantially increased total reserve extent required to meet objectives. Pressey and Logan (1998) interpreted this result as the larger above-objective representation of environmental classes with larger planning units because of reduced precision in sampling parts of classes. Rouget (2003) and Hamel et al. (2013) also found that large planning units were less efficient in terms of area needed in meeting conservation objectives. Spatial dissimilarity between all solutions was also considerably influenced by planning-unit size. This is not completely unexpected considering the potential disparities in total reserve extent attributable to planning-

unit size. Indeed, other studies support this finding with the observation that changes in planning-unit size yield spatially different priority areas (Warman et al. 2004, Shriner et al. 2006, Nhancale and Smith 2011). Different planning-unit sizes producing diverging priority areas are likely due to the spatial inflexibility, relative to features being represented, that occurs when planning-unit size increases (Nhancale and Smith 2011), reflected in the commonly observed accompanying loss of spatial efficiency (Warman et al. 2004, Justus et al. 2008).

Thematic resolution of reef classes had the least overall influence on priorities. The only consistent direct effect of this factor was the increase in total reserve extent with increase in resolution, regardless of planning-unit size or cost variability. This occurs because of the lack of fit between planning units and the boundaries of environmental classes (Pressey and Logan 1995), which is exacerbated when larger planning-unit sizes and finer, more detailed thematic resolutions are used. Despite having the least direct influence on priorities, the most significant influence of thematic resolution occurred in combination with other prioritisation factors examined in this study, highlighting the importance of exploring these interactions.

2.5.2 Interaction effects between prioritisation factors

The most notable interaction between the three factors we examined was between cost variability and thematic resolution. I found that solutions with variable cost (which were less costly overall) were more sensitive to increasing thematic resolutions than those with uniform cost. This result reflects the profiles of variable cost within reef classes. At lower thematic resolutions, there are more spatial options for achieving objectives and a higher probability that objectives can be achieved with relatively cheap planning units. At higher thematic resolutions, spatial options for achieving objectives are more constrained, and more expensive planning units are required to achieve objectives for at least some of the rarer reef classes that result from thematic subdivision. However, it is important to note that the exception to this trend was with thematic resolution level 4, which actually resulted in a decrease in total cost compared to level 3, despite having a greater number of reef classes. This is due to the nature of reef classifications used for level 4, which were not uniquely hierarchical within level 3 classes (in other words, the same level 4 classes existed under different level 3 categorical classes). Thus, the level 4 reef classes are more widespread over the planning domain, leaving more options to find lower cost solutions. I am aware of only one previous study that has also demonstrated an interaction between thematic resolution and the spatial variability of cost in terms of cost efficiency (Deas et al. 2014). This work contributes to that limited evidence base.

The change in interaction effect found when increasing thematic resolution from level 3 to level 4 demonstrates the relevance of the relationship between the spatial distribution of environmental

classes and cost values, and how this can influence the ability to achieve cost efficiency in reserve solutions. This is further supported by the fact that level 4 had a higher number of rare reef classes compared to level 3, which, all other data aspects being equal, should result in increased total reserve cost (Pressey et al. 1999). The consistent decrease in total cost seen with level 4 compared to level 3, when prioritised with variable cost only, suggests that two other factors are in play. First, the cost profiles of the rare reef classes in level 4 could be lower than those in level 3 (i.e., rare classes in level 4 can be found within less costly planning units), leaving more scope to achieve objectives in lower-cost planning units. Second, there could be higher spatial co-occurrence of rare reef classes in level 4, which would essentially increase the spatial (and therefore cost) efficiency of meeting the objectives for these rare classes in fewer planning units. A weaker signal of reduced total variable cost for level 4 reef classes was apparent in the Micronesia results, demonstrating the region-specific nature of interactions between prioritisation factors influencing planning outputs.

Another aspect of the interaction between thematic resolution and variable cost was their combined influence on spatial differentiation between priorities. The mechanism behind changes in spatially variable costs determining different spatial priorities (Adams et al. 2010, Watson et al. 2011) makes this finding understandable. A spatially variable cost layer constrains the choices of planning units (to cheaper ones, where possible) to meet the increasing number of reef class objectives with increasing thematic resolution. This constraint leads to spatially distinct sets of planning units among thematic resolutions because, at each thematic resolution, the spatial relationship between cost and reef classes is different. It should be noted again that this is likely to arise only if planning-unit options for certain classes are constrained due to the spatial pattern of expensive planning units in the cost layer. Conversely, with uniform cost, selection of planning units is more spatially flexible, leading to less distinct sets of planning units between thematic resolutions.

The other notable interaction occurred between planning-unit size and thematic resolution. Large planning units interacted with increasing thematic resolution to significantly increase total reserve extent required to achieve the overall 30% objective, particularly towards the higher resolutions. This interaction effect was considerably less evident in the scenarios with small planning units. Again, this is likely due to the greater spatial mismatch that occurs between the larger planning units and the boundaries of the finer-resolution environmental classes (Pressey and Logan 1995, Rouget 2003), resulting in less spatial precision and flexibility in achieving the same objectives.

2.5.3 Ability of coarse prioritisations to represent finer-resolution priorities

My results indicate that coarse prioritisations are unreliable guides to fine-resolution priorities unless the same socioeconomic data are used at both resolutions. This presents an inherent risk, since planning at regional scales using variable cost data will almost certainly involve coarse-resolution surrogates for cost (Giakoumi et al. 2013) that are unlikely to reflect variables important to people on the ground (Richardson et al. 2006, Weeks et al. 2010b, Mills et al. 2010, Deas et al. 2014). Put another way, nestedness of priorities would be expected only if fine-resolution priorities used the same limited, coarse-resolution surrogates for cost applied across whole regions, and this would require planners to ignore local insights into actual costs. Updating cost data on the ground as coarse-resolution priorities are investigated would inevitably change the cost data on which those coarse priorities were based (Pressey et al. 2013). While allowing local fine-tuning of priority areas, this would also undermine the very basis for the coarse priorities.

Given that achievement of conservation objectives is the basic aim of systematic conservation planning, incidental representation of finer-resolution environmental classes by coarse prioritisations is at least as important as the spatial nestedness of priorities. My analysis on the extent of incidental representation of level 5 reef classes demonstrated that all scenarios prioritised with uniform cost outperformed scenarios with variable cost. Whilst some previous studies similarly found reasonable potential for coarse prioritisations to achieve fine-resolution objectives (Rouget 2003, Payet et al. 2010), these did not consider variable socioeconomic cost data in their prioritisations. Bridge et al. (2016), however, found a significant negative relationship between planning-unit cost and incidental representation of environmental classes in the Great Barrier Reef marine reserve network, Australia. The less spatially-biased selection of planning units that occurs when no (or uniform) costs are considered allows a greater chance for incidental representation of finer-resolution objectives, with this effect being almost indistinguishable between the different levels of thematic resolution tested in this study.

With our increasing understanding of the ability of cost data to influence conservation priorities (Bode et al. 2008, Ban and Klein 2009, Adams et al. 2010), we must now recognise the imperative to use accurate and representative cost layers in conservation prioritisations. Such data rarely exist across extensive regions, due to the expense and complicated logistics involved in obtaining them. It is thus critical that we formulate strategies to help circumvent this data limitation, based on our understanding of the relative influence of prioritisation factors and the potential interactions that can occur between them. Based on my findings, one such strategy could be to prioritise with spatially uniform costs when planning across large regional extents and then update priorities as finer-resolution data on biodiversity and costs become available, as

coarse-resolution planning transitions to fine-resolution implementation (Mills et al. 2010). My results indicate that using uniform costs for initial regional-scale planning in an adaptive planning process will increase the likelihood of incidental representation of finer-resolution objectives, through the initial selection of planning units that are not biased towards planning units that appear cheaper with coarse, and probably inaccurate, regional-scale surrogates.

Importantly, I found that confidence in incidental representation was lower for fine-resolution reef classes that were less extensive in my study areas. Similar findings have come from previous studies (Kirkpatrick and Brown 1994, Lombard et al. 2003, Payet et al. 2010, Bridge et al. 2016). As rare environmental classes or species tend to be at most risk of destruction or extinction (Pimm et al. 1995, Roberts and Hawkins 1999), this is a critical point to consider regarding the ability of coarse prioritisations to represent fine priorities.

2.6 Conclusions

With increasing calls for conservation planning to incorporate socioeconomic costs (Richardson et al. 2006, Carwardine et al. 2008, Ban et al. 2009a, 2013), my findings in this chapter lead to three recommendations. First, where regular planning units are employed, the smallest practical planning-unit size will maximise spatial, and therefore, cost efficiency. Second, wherever possible, planners should invest in accurate biodiversity and socioeconomic cost data (or surrogates) at the highest resolution possible. While these two recommendations come generally at the price of practicality or expense of data acquisition, our third recommendation considers situations where the first two are not feasible. When planning across regional extents and when incidental representation of fine-resolution (or unmapped) environmental classes is desirable, it is better to prioritise the whole region with uniform costs if subsequent finer prioritisations will follow. Otherwise, spatially variable cost data can bias selection of planning units enough to reduce the likelihood of incidental representation of fine- resolution environmental classes. The importance of incorporating socioeconomic cost data is now commonly touted in the conservation planning literature. However, I have found that the cost data used in conservation prioritisation can be so influential on the configurations of selected areas, that failing to recognise appropriate scales at which to incorporate cost data can lead to negative consequences for biodiversity.

While I have shown that the influence of planning-unit size, thematic resolution, and cost variability have generally consistent impacts on conservation prioritisation outputs, interactions between these factors can lead to surprising results. Although consistent findings from both my study regions, involving very different reef complexities, suggest the findings might be generalised to other situations, it should be noted that the reef-class mapping for the two regions

originates from the same mapping project and uses the same method (Andréfouët et al. 2006). My study adds to the growing body of evidence on these interactions, but repeated studies of this nature with reef-class data that are categorically non-hierarchical and unrelated, along with different types of socioeconomic cost data, would be valuable in better understanding the potential interactions that can occur. Further understanding the details on how these problems propagate throughout the prioritisation process is relevant to achieving more effective and efficient conservation solutions in the face of expanding loss of global biodiversity and natural environments, and waning conservation resources.

Chapter 3

The plans they are a-changin': more frequent iterative adjustment of regional priorities in the transition to local actions can benefit implementation

3 The plans they are a-changin': more frequent iterative adjustment of regional priorities in the transition to local actions can benefit implementation

In Chapter 3, I investigate the implications of iterative planning processes with respect to how frequently regional conservation priorities should be updated as local actions are implemented incrementally across a region. This chapter begins to tackle the practical considerations of scale in conservation planning outlined in Chapter 1: specifically, when making the transition from regional conservation designs to locally applied actions. This work provides insights into potential benefits to updating regional priorities more frequently and trade-offs to consider regarding the frequency and cost of regularly updating priorities. I conceptualised the research, designed and assisted coding the simulation framework, analysed the data, and wrote the chapter. Pressey and Weeks assisted with conceptualising the research and analysing data, and structuring and editing the manuscript. VanDerWal provided advice on designing the simulations and data analyses, and created the management unit layer. Storlie led coding of the simulation framework, and assisted with data analyses and editing the manuscript.

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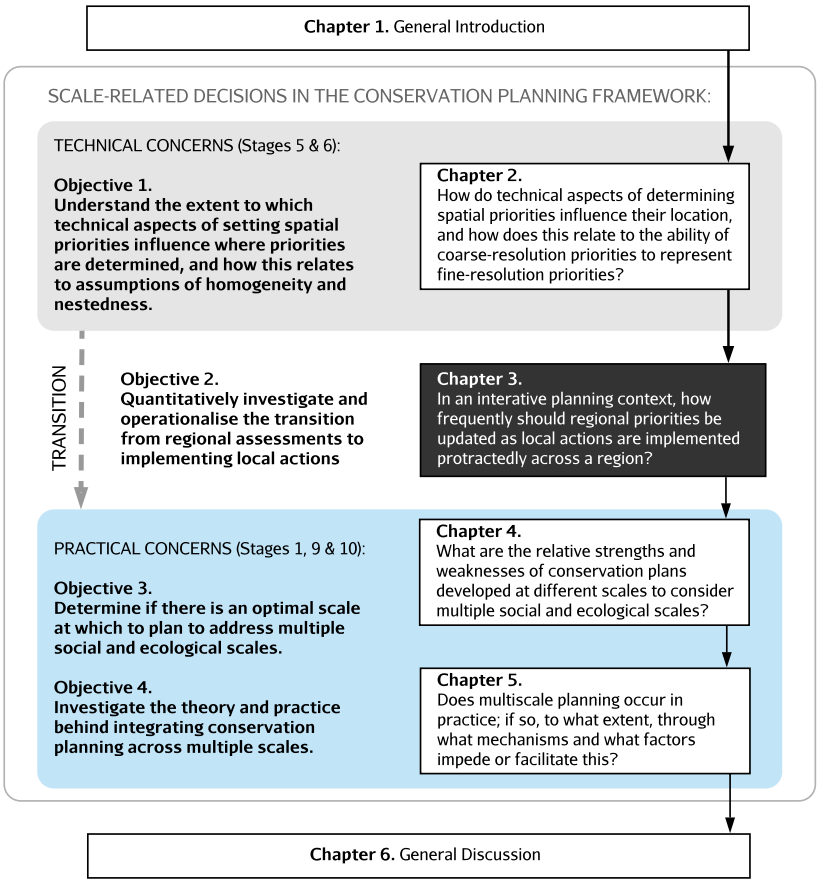

3 The plans they are a-changin': more frequent iterative adjustment of regional priorities in the transition to local actions can benefit implementation

3.1 Abstract

Regional-scale assessments are frequently conceived to guide the strategic application of conservation actions. Though changes to priority areas from initial assessments are inevitable, the transition from regional-scale assessment to implementing local actions is poorly understood. An outstanding question concerns the frequency with which regionally assessed priorities should be updated as actions are implemented. I address this gap by simulating the incremental implementation of local actions guided by regional conservation assessments, using Fiji as a case study, and explore how update frequency can influence aspects of translating regional assessments to local actions. My simulations were designed within the framework of systematic conservation planning, with implemented actions simulated on the basis of conservation value in achieving objectives and feature rarity. Other decision rule-sets were put in place to simulate on-the-ground negotiations that are often necessary when transitioning from regional-scale conservation assessments to local actions. I use these simulations to evaluate how the frequency of updating regional priorities influences: (1) total time taken to achieve objectives represented by numbers of planning units investigated, (2) total extent of final reserve systems, and (3) spatial overlap between initial regional priorities and final implemented reserves. I show that changes in the frequencies of updating did not influence the time taken to achieve conservation objectives, nor the total extent of final reserve systems. However, there was a significant difference in the number of times planning units were re-investigated for implementing actions within scenarios that involved more frequent updates. Spatial overlap between initial regional priorities and final implemented reserves increased with decreases in update frequency. I demonstrate two potential benefits to updating priorities more frequently: (1) faster achievement of objectives for high-priority features, and (2) greater potential to capitalise on areas previously investigated. My findings provide insights into trade-offs to consider regarding the frequency of updating regional assessments, which varies depending on the planning context.

3.2 Introduction

Within the last decade, literature on conservation planning has acknowledged an 'implementation crisis', referring to the preoccupation of research efforts with systematic

assessments that primarily focus on the *where* for conservation, rather than the *how* (Knight et al. 2008, Barmuta et al. 2011, Biggs et al. 2011). In part, the gap between assessment and implementation is likely due to the difficulty of the transition, including a mismatch of scales between these two parts of the planning process (Mills et al. 2010, Guerrero et al. 2013) and political constraints on expanding protection in the face of extractive activities. In applied conservation, funding models have tended to value short-term, tangible project outputs such as plans on paper, without funds allocated for implementation (Weeks et al. 2015), which can require years or even decades to complete (Pressey et al. 2013). Compounding this, the academic environment has rewarded theoretical exploration of variations on conservation assessment problems rather than feasible approaches to implementation. It is now clear that integration of the different scales involved in planning is needed for the transition from assessment to implementation (Game et al. 2011, Mills et al. 2012, 2014, Guerrero et al. 2013, Gaymer et al. 2014, Magris et al. 2014). Understanding the complementary advantages of both regional and local perspectives (Pressey et al. 2013) is necessary to successfully navigate this transition.

The assumption of progressive planning from coarse to fine scales is implicit in many regional-scale planning exercises (e.g., Klein et al. 2010, Beger et al. 2013). However, rather than a linear process, literature suggests that there need be feedback and multiple iterations of regional- and local-scale perspectives (Root and Schneider 1995, Mills et al. 2010, Holness and Biggs 2011, Pressey et al. 2013, Beger et al. 2015). In many situations, particularly where initial assessments are followed by years of incremental action (hereafter, 'protracted implementation'; Pressey et al. 2013), adjustments to assessments will become necessary as new information emerges during implementation (Pressey and Bottrill 2009, Weeks and Jupiter 2013). There are no documented examples of this occurring however, or even theoretical exploration of the implications of iterative planning cycles.

One way to operationalise iterative planning has been through regional plans that are updated at defined intervals, as a result of ongoing learning arising from implementing actions (Mills et al. 2015, Beger et al. 2015). However, a trade-off exists between resources and time invested in repeated, updated assessments and those invested in implementation. As a starting point to better understand this trade-off and to help inform decisions about how frequently assessments should be updated, I explore how the frequency with which regional priorities are updated affects some of the interactions between assessment and implementation.

I simulate the incremental, protracted implementation of local actions guided by regional assessments. In these simulations, I focus on the transition from arbitrary units of prioritisation (planning units) to units within which actions can feasibly be applied (management units). Conservation actions can refer to numerous management approaches; in this instance, I refer to

implementing marine reserves. As reserves are implemented over time and planning units are converted to unaligned management units, the configuration of the initial regional assessment is progressively updated to account for over- or under-achievement of conservation objectives. I vary the frequency with which the regional assessment is updated, and assess the effect this has on: (1) total time taken to achieve objectives, (2) total extent and cost to achieve objectives, and (3) spatial overlap between the initial regional assessment and the final implemented reserves.

3.3 Methods

In a hypothetical planning initiative, a national planning body undertakes a regional conservation assessment to produce a marine reserve network that achieves explicit biodiversity objectives. Areas of interest that emerge from this assessment direct funds towards engaging local communities in those areas, which results in some conservation actions being applied. After a number of actions have been applied, the regional assessment is updated to account for lack of fit between planning units in the assessment and management units for on-ground action (Govan et al. 2009) and consequent over- or under-achievement of objectives.

To simulate the transition between regional assessments and local implementation (Figure 3.1), I first used the decision-support tool Marxan (Ball et al. 2009) to develop a regional-scale conservation assessment that would achieve representation objectives for all coral-reef features (Step 1, Figure 3.2), using Fiji as a case study. Fiji is a good example of the context of protracted implementation: the primary mechanism for managing coral reefs is through implementation of locally managed marine areas, requiring regional-scale prioritisations to be adapted to particular local contexts (Mills et al. 2011). As described in Chapter 2, the main outputs from a Marxan prioritisation are a least-cost assessment of priority areas that together achieve all conservation objectives, and the selection frequency of planning units in achieving objectives across multiple alternative assessments. I used the selection frequency values to determine the order of planning units within which to implement reserves (hereafter, 'investigate') in the simulations (Step 2, Figure 3.2).

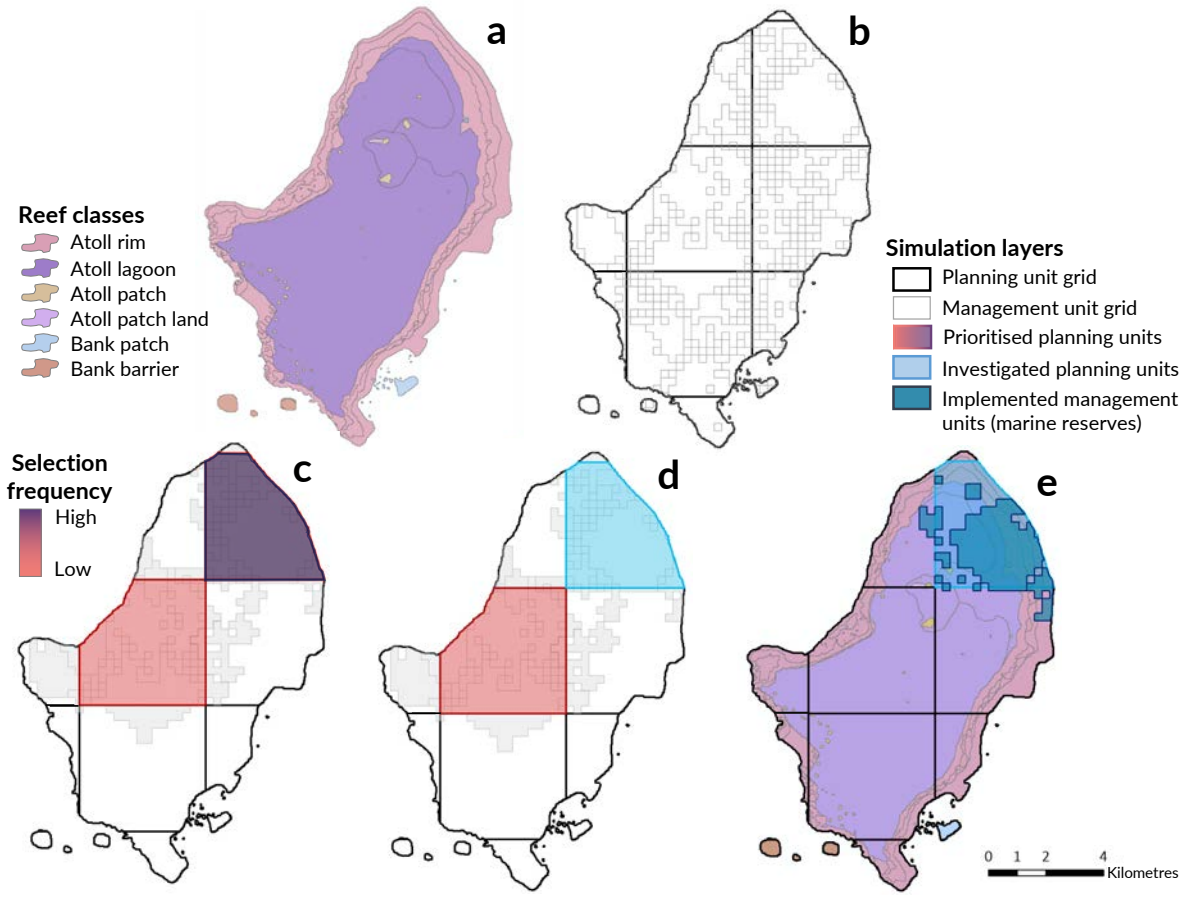

Figure 3.1 Transitions from regular prioritised planning units used in assessment to irregular management units used to apply actions. Map shows a small part of the study area, defined by coral reefs in Fiji. (a) Distribution of reef classes (targeted conservation features). (b) Layout of the planning unit grid (black), clipped to contain reef areas only, overlaid with the simulated management units (grey). Management units are irregular and their boundaries and size differ from planning units, producing a spatial mismatch between the two grids. (c) Planning units prioritised in the initial regional assessment (coloured planning units; red to purple shades reflect differences in selection frequency values between prioritised planning units). Note that only empirical investigation of prioritised planning units enables identification of management units suitable for implementing marine reserves (shaded in grey, indicating learned information on existing management units intersecting the investigated planning units). (d) During each iteration, highest-priority planning units are selected for investigation (light blue) for investigation of implemented actions. (e) Conservation actions implemented as marine reserves (dark blue) with management units that intersect the investigated planning units. Discrepancies between the extent of conservation features within the investigated planning units and implemented management units prompt a revision of the regional assessment at varying frequencies, defined by numbers of investigated planning units. The revision and reprioritisation process is iterated until all conservation objectives are met.

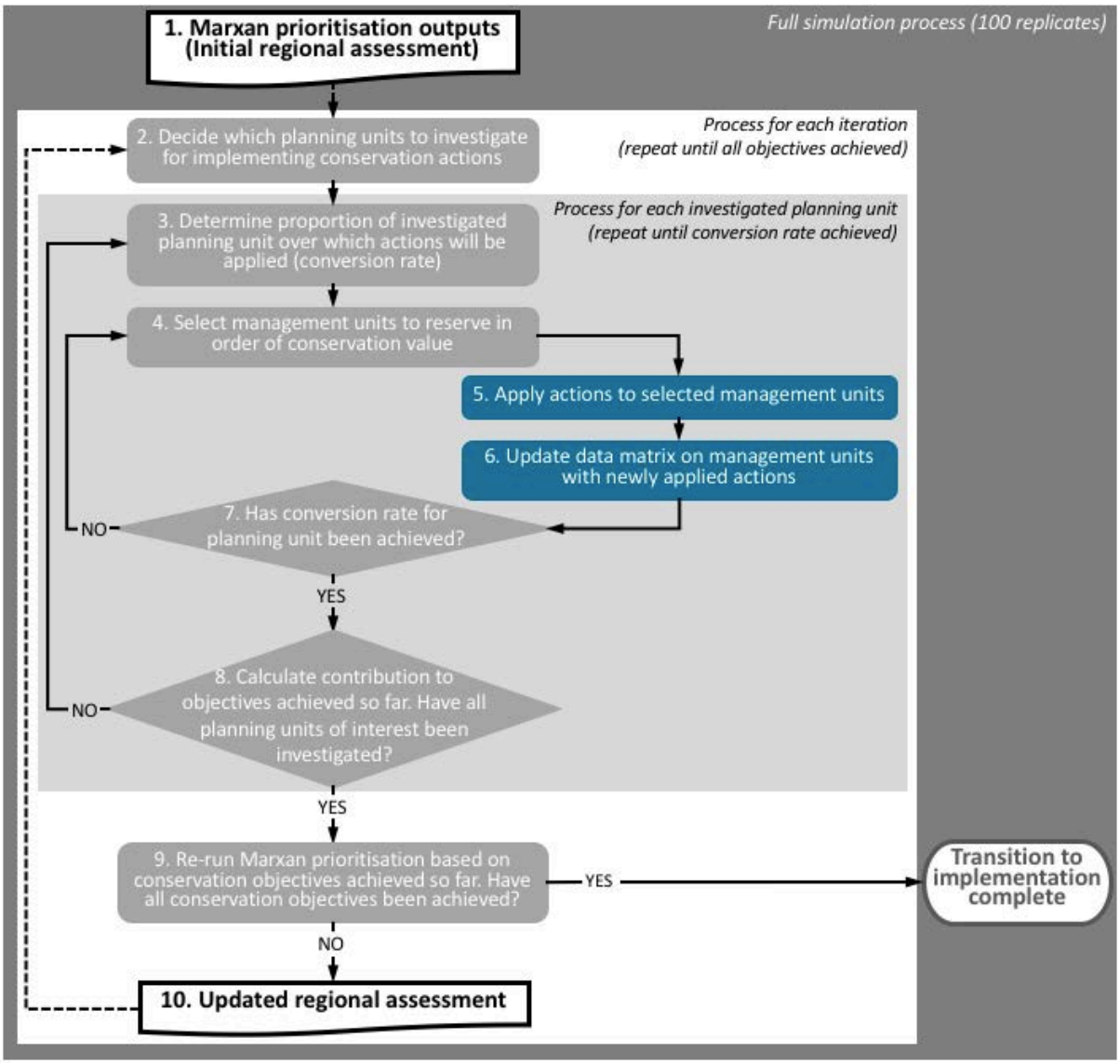

Figure 3.2 Steps in the process of simulating the transition between regional assessment and local actions. Solid arrows indicate action steps; dashed arrows indicate assessment inputs. Grey steps concern planning units. Blue steps concern management units. Note that decisions about planning units and management units might be undertaken by different organisations or teams. Innermost shaded box indicates steps repeated for each investigated planning unit until the assigned conversion rate to implemented reserves is achieved. White box indicates the steps involved in each iteration (Table 3.1) of the simulation, repeated until all conservation objectives were achieved. Each simulation (signified by outermost dark grey box) was replicated 100 times.

At each iteration (terms referring to parts of the simulation are defined in Table 3.1), marine reserves consisting of a number of management units that overlapped with high-priority planning units (Table 3.1), were implemented according to the framework outlined in Figure 3.2 (Steps 3-7; specific parameters and rule sets defined for the relevant decision-making steps in the simulation model are detailed in Appendix 2 Text A2.1). Management units intersecting prioritised planning units were selected for reserve implementation in order of their relative

conservation value, determined as the sum of the products of rarity and remaining objective to be achieved for the reef classes present within their boundaries (Appendix 2 Table A2.1). Management units continued to be selected until a pre-determined proportion of the prioritised planning unit had been converted to reserves (Appendix 2 Text A2.1). To avoid the eventual production of very small planning unit slivers as management units are sequentially converted to reserves, if the remainder of the planning unit was less than 50% of its original size (i.e., < 12.25 km^2) it was merged with a neighbouring planning unit with a shared boundary (Appendix 2 Text A2.1). The value of 50% was chosen to ensure that planning units being used for the prioritisations would still be considered 'large', typical of regional prioritisations.

Table 3.1 Definitions of key terms used to describe the simulations.

Key term	Definition
Assessment	The design phase of conservation planning (stages 1-9 of Pressey and Bottrill 2009), including spatial prioritisation.
Spatial prioritisation	The output from a regional assessment; biogeographic-economic spatial analyses used to identify important biodiversity areas to achieve conservation objectives efficiently (Kukkala and Moilanen 2013).
Selection frequency	A measure of the relative importance of the planning unit in contributing towards achievement of objectives. Represents how often a given planning unit is selected as part of a final reserve system across a series of Marxan runs.
Simulation scenarios	The four simulation scenarios examined in this study varied in the frequency with which the regional assessment was updated, modelled as the number of planning units that were investigated at each iteration. Across the simulations, this number was 25, 50, 75, or 100 planning units.
Planning units	Often regular in shape, used in prioritisations as a spatially explicit array of areas for assessment and comparison for conservation priorities.
Management units	Areas to which conservation actions (e.g., marine reserves) are applied locally. These are typically small and irregularly shaped in Fiji and other developing countries (Mills et al. 2010, Weeks and Jupiter 2013) and, on land, include property boundaries in developed countries.
Iteration	One cycle of the simulation (Steps 2-10; Figure 3.2).
Objectives	Explicit, quantitative amounts of each conservation feature (reef classes in this study) to be represented in conservation areas.
Reef classes	Geomorphological classes reflecting reef structures (Andréfouët et al. 2006).

Data matrix	The relational table containing all spatial information on the relevant data layers: planning-unit grid, management-unit grid, implementation status, and reef classes. The data matrix was continuously updated throughout the simulation.
Conversion rate	The percentage of each investigated planning unit for conversion to management units. This parameter was essential to reflect the fact that whole planning units are rarely implemented because negotiations are needed to apply management within boundaries defined by local communities, which tend to be irregular and unrelated to the size of planning units (Mills et al. 2011, Weeks and Jupiter 2013).

Once reserves had been implemented within all investigated planning units, information on the amount of each feature protected so far was used to update the regional assessment, to account for under- and over-achievement of objectives arising from the spatial mismatch between planning units and management units (Figure 3.1). The regional assessment was then re-prioritised with Marxan with implemented reserves 'locked in' (Step 9; Figure 3.2), and Steps 2-10 (Figure 3.2) were repeated until all conservation objectives were achieved. I coded the simulations using the programming language, *R* (R Development Core Team 2016), with necessary packages for handling large spatial and tabular data ('SDMTools' v1.1-221 package, VanDerWal et al. 2014; 'data.table' v1.10.0 package, Dowle and Srinivasan 2016).

For my case-study application, maps of reef classes used to inform conservation objectives came from the Millennium Coral Reef Mapping Project (Andréfouët et al. 2006), which contained 31 unique reef classes in Fiji. Planning units (n = 1182) used were regular 24.5 km^2 square grids, clipped to the boundaries of coral-reef classes (see Appendix 2 Text A2.2). This planning-unit size was selected based on the 'large' planning units determined for Chapter 2. Management units were not known *a priori* in Fiji (or in most other regions) because they are determined by local communities during implementation (Jupiter and Egli 2011). Thus, a management unit layer of irregular polygons was simulated, using the size distribution determined by that of existing fisheries closures in Fiji (see Appendix 2 Text A2.2).

In all scenarios, prioritisations sought to achieve a conservation objective of 30% of each reef class. I assumed no existing marine reserves in the simulations. Each Marxan prioritisation was set to produce 100 unique reserve assessments (i.e., 100 replicate runs), and each simulation scenario was replicated 100 times to account for stochasticity. Planning unit costs were assigned uniformly (equal to area). I ran prioritisations with homogeneous costs because socioeconomic cost data available across national extents are typically of coarse resolution and thus likely to be poorly correlated with true opportunity costs. This decision was also informed from my results in

Chapter 2, where I found that costs can have a significant influence on biasing spatial priorities, as well as reducing the likelihood of coarse prioritisations to represent finer-resolution priorities.

To explore the effects of updating frequency of regional assessments, I altered the number of planning units to investigate for implementation of reserves at each iteration of the simulation (Step 2, Figure 3.2), while assuming a constant rate of investigation of 25 planning units every two years. This investigation rate was chosen based on an assumption that priorities would not be updated more frequently than once every two years, and an approximate calculation of the number of planning units that would need to be investigated each year to achieve all conservation objectives within 30 years. I explored four simulation scenarios (Table 3.1) in which the regional assessment was updated after 25, 50, 75, or 100 planning units were investigated (hereafter identified as, 'scenario 25', 'scenario 50', etc.). This equates to an update of the regional assessment once every 2, 4, 6, or 8 years.

I used three measures to assess the influence of update frequency on the iterative transition between regional assessments and local actions: (1) the total time taken to achieve all objectives, (2) the total extent of marine reserves required to achieve all objectives, and (3) the spatial overlap between the final, implemented system of marine reserves (as management units) and the initial prioritised regional assessment (as planning units). Because the rate of investigation was constant across all scenarios, the total time taken to achieve all objectives in each run of the simulation was represented by the total numbers of planning units investigated. For example, if one completed simulation run involved the investigation (and subsequent conversion to management units) of 1200 planning units in total, then this would translate to 24 years taken to achieve all objectives, based on the rate of investigation assumed. Since planning units that were investigated could leave remnant planning units (> 50% of original size) for the next iteration of regional prioritisation, it was possible that the same planning unit could be 'reinvestigated' in subsequent iterations of the simulation. Thus, the total numbers of investigated planning units calculated for each simulation run could be assessed in terms of both absolute (total numbers of units investigated, irrespective of which specific planning units) and unique (total unique planning units investigated) numbers. The total extent of finally implemented marine reserve systems was calculated in square kilometres. Spatial overlap was calculated as the proportion of the first regional assessment (i.e., extent and configuration of the initially prioritised planning units) that were implemented as reserves at the end of the simulations.

3.4 Results

3.4.1 Total time taken

There was no significant difference between scenarios in the time taken, indicated by total numbers of planning unit investigations (not accounting for reinvestigated planning units), to achieve all conservation objectives ($F_{3,396} = 1.88$, $p = 0.13$; Figure 3.3, and Appendix 2 Figure A2.2 for summary of calculated averages across 100 replicates for each scenario). However, when accounting for planning units investigated twice or more during the simulations, there was a significant difference in the number of *unique* planning units investigated between all scenarios ($F_{3,396} = 2238$, $p < 0.001$; Tukey's post-hoc, all p-values < 0.001; Figure 3.4a). Reinvestigations of individual planning units occurred most frequently in scenario 25, decreasing towards scenario 100, with the proportion of planning units investigated just once throughout the simulations increasing from 0.24 to 0.39 between scenarios 25 and 100 (Figure 3.4b). Planning units with the highest numbers of reinvestigations (13-15) were also observed in scenarios 25 and 50.

Representation objectives for rarer features (see Appendix 2 Text A2.3 for method used to calculate feature rarity) were consistently achieved earlier in the simulations than those for more common features, reaching similar extents, on average, of over-achievement across scenarios by the time all objectives were finally achieved (Figure 3.3 and Figure A2.2). Objectives for rarer features were achieved earlier in the simulations with more frequent updating (e.g., scenario 25 and 50), compared to those with less frequent updating (Figure 3.3 and Figure A2.2).

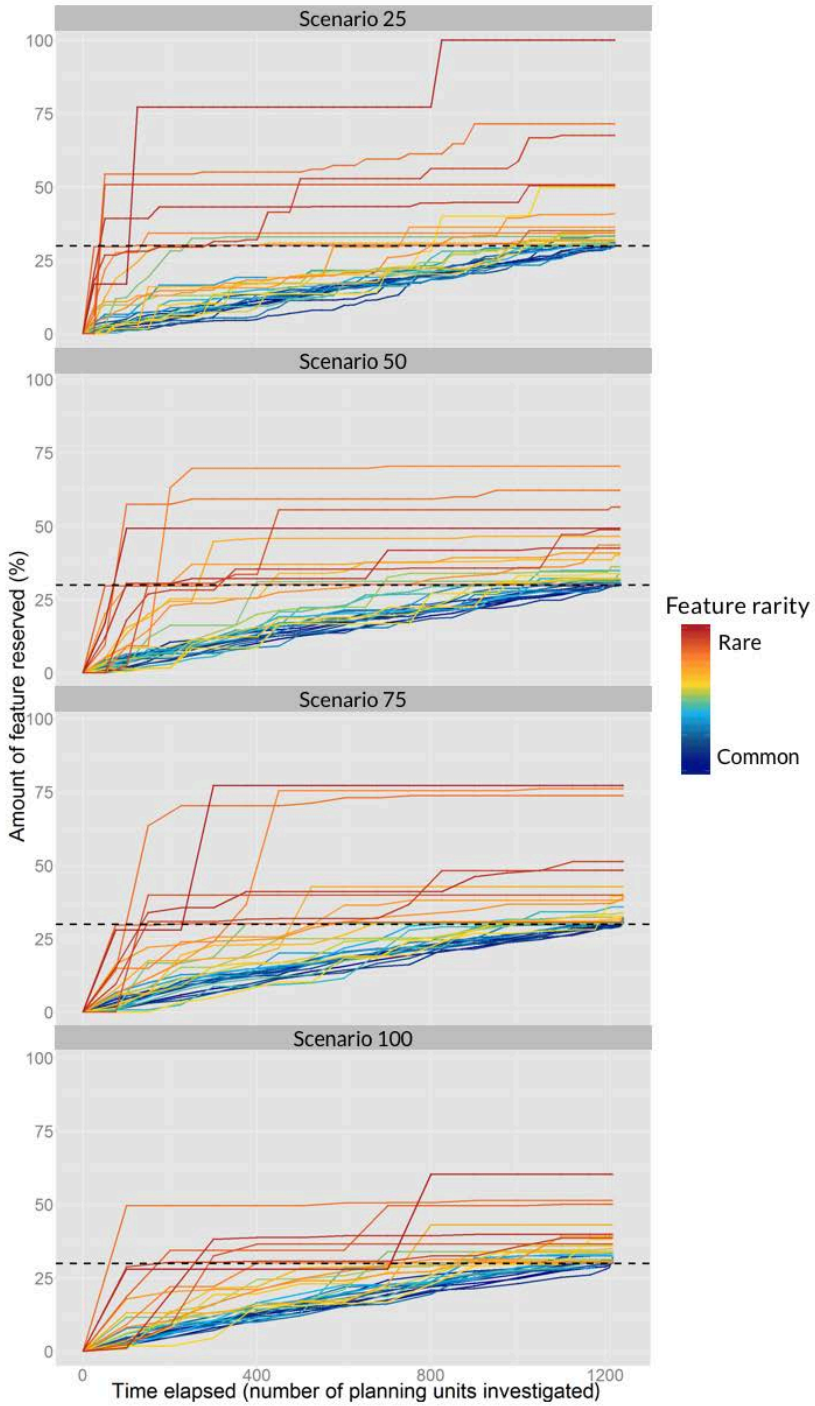

Figure 3.3 Total times taken to achieve all conservation objectives across scenarios, indicated by total numbers of planning unit investigations throughout the simulations. The graphs show one replicate out of 100 for each scenario, to demonstrate the increases in achievement and overachievement of objectives during the simulation, with replicates selected to be representative of the average length of time taken. The rate of implementation was equal across scenarios; for example, the regional assessment in scenario 100 was updated four times less frequently than scenario 25 across the length of the simulations. The 31 reef classes are coloured by gradient to reflect the order of rarity; blue indicates the most common feature, red, the most rare.

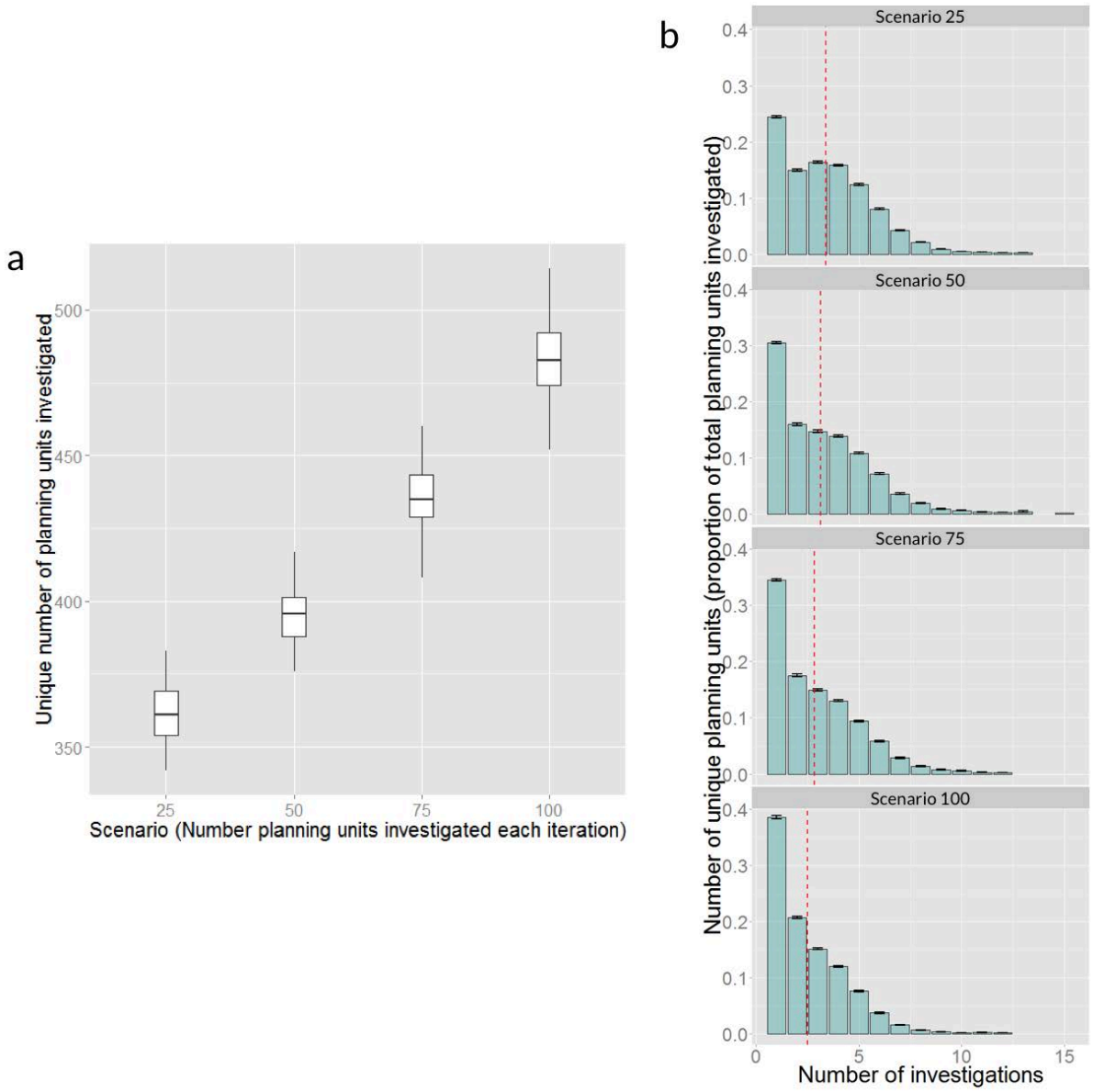

Figure 3.4 Numbers of unique planning units investigated and reinvestigated during the simulations. (a) Unique numbers of planning units investigated for each scenario, showing variation across 100 replicates. Boxes indicate median, lower quartile, and upper quartile values; vertical lines represent bottom and top quartiles. (b) Number of investigations of unique planning units in each scenario, expressed as proportions of total unique planning units investigated. Proportions are averaged across 100 replicates for each scenario; error bars indicate standard error across replicates. Red dashed lines indicate the mean numbers of investigations of unique planning units between scenarios (note slight shifts to lower mean numbers of investigations as frequency of regional updates decreases).

3.4.2 Total extent

The total extent of implemented reserve systems was similar across all scenarios (Figure 3.5a), with no significant differences apparent ($F_{3,396} = 2.45$, $p = 0.06$). Variation in total extent of reserve systems across 100 replicates within each scenario was consistently very small, with standard deviations ranging from 0.34% to 0.46% of the means for each scenario.

3.4.3 Spatial overlap

Mean proportion of spatial overlap between initial assessments and final systems of marine reserves varied from 0.326 to 0.357, with standard deviation between 0.012 and 0.015. Spatial overlap decreased as regional assessments were updated more frequently (Figure 3.5b). There were significant differences in spatial overlap between all pairs of scenarios ($F_{3,396} = 90.4$, $p < 0.001$; Tukey's post-hoc, all p values < 0.001).

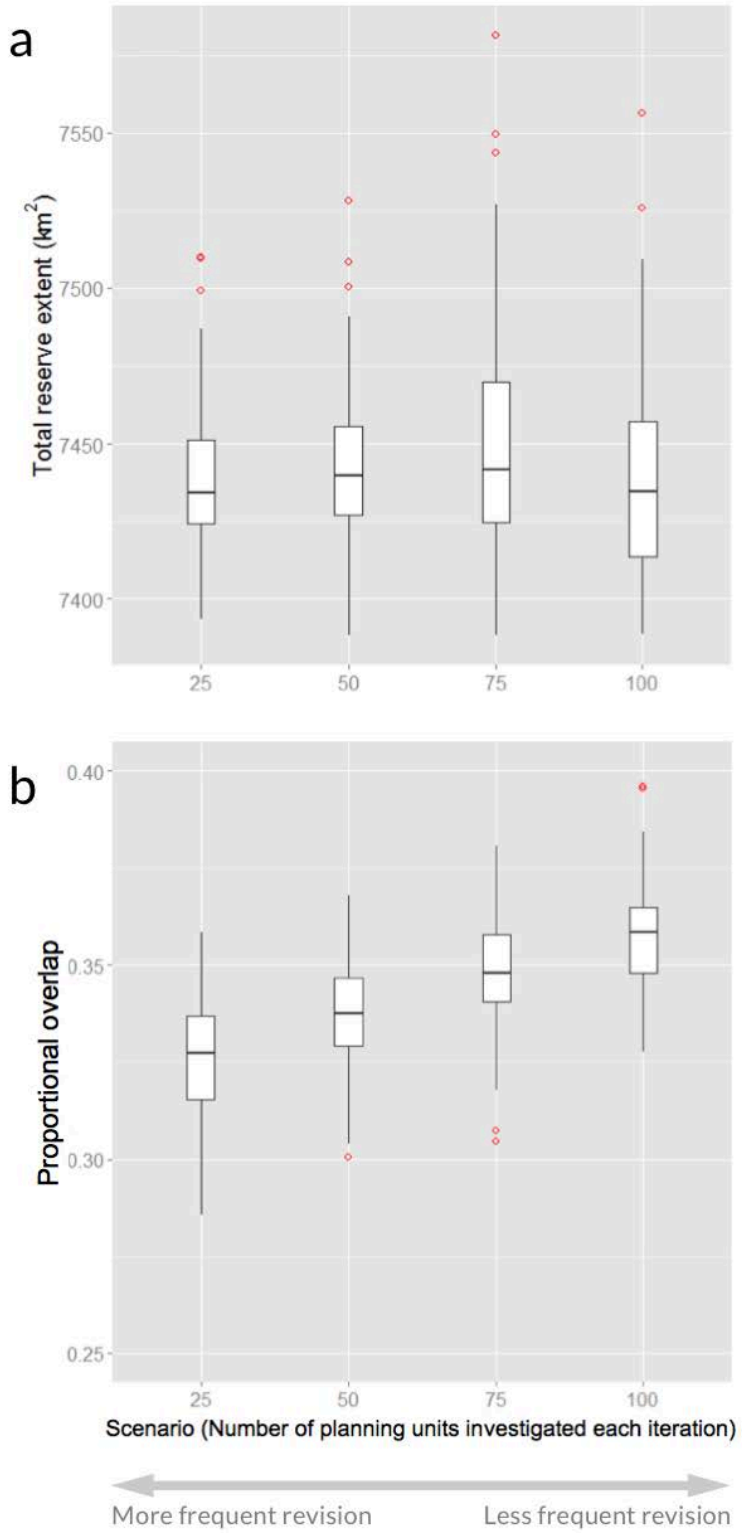

Figure 3.5 Spatial results for implemented marine reserves. Total extent to achieve all objectives (a) and spatial overlap between initial regional assessments and final reserve systems (b) across scenarios. Results for each scenario are shown across 100 replicates. Boxes indicate median, lower quartile, and upper quartile values; vertical lines represent bottom and top quartiles. Red circles represent outlier values.

3.5 Discussion

The simulation scenarios analysed here represent a spectrum of frequencies with which regional assessments for marine reserve systems were updated. For clarity, I discuss comparative results from scenarios 25 and 100; where differences between scenarios were apparent, the intermediate results of scenarios 50 and 75 were gradational between the two extremes. I acknowledge that updating frequencies outside my tested range could be expected, although extrapolation of my results to these frequencies could be unreliable.

The frequency with which regional assessments were updated did not influence the time taken to achieve all objectives in my simulations, or the total extent of implemented reserves. However, different results might be observed in regions with different patterns of feature rarity (e.g., a larger proportion of rare features, or less variable distributions of rarity). Representation objectives for rarer features were achieved more quickly across all scenarios because of the decision rules to investigate planning units in order of selection frequency (Step 2, Figure 3.2) and initiate implementation with reserves containing rarer features (Step 4, Figure 3.2). Notably, achievement (and subsequently overachievement) of objectives for rarer features occurred earliest in scenario 25. This scenario investigated only the 25 highest priority planning units at each iteration, of which a higher proportion contained rare features. In contrast, scenario 100 investigated further down the priority list of planning units at each iteration, effectively placing less emphasis on early representation of the rarer features.

Earlier achievement of objectives for rarer features with more frequent updating has implications for real-world conservation planning. My simulations did not model the loss or degradation of conservation features in parallel to the protracted application of management actions (cf., Visconti et al. 2010). To ensure protection occurs before opportunities for conservation are lost, features that are more threatened should be prioritised during negotiations regarding implemented actions (equivalent to my decision-rule in Step 4, Figure 3.2), and more frequent revision of regional assessments might be warranted.

Although the absolute number of planning unit investigations was consistent across scenarios, scenarios updated more frequently presented more opportunities to revisit previously investigated planning units. In scenario 25, fewer unique planning units were investigated, but reinvestigations led to a greater proportion of each, on average, being converted to reserves. As a result, the total final reserve system extents were not distinguishable across the simulation scenarios.

That planning unit reinvestigations were more common in scenarios with more frequent updating also has real-world implications. In some contexts, the transaction costs (Naidoo et al. 2006) of returning to previously investigated areas could be reduced, compared to establishing relationships with new communities. Reinvestigations could therefore be a more efficient approach to achieving representation objectives, provided that they result in expansion or growth in number of existing reserves. If attitudes towards existing reserves are positive, communities might be willing to increase the area under protection (Weeks and Jupiter 2013); however, frequent local planning could also lead to process fatigue, and expectations of increased reserve extent might not be met if acceptable upper limits on reserve area are reached. Benefits from reinvestigations are likely to be greatest where initial implementation is less extensive and sufficient time between 'investigations' is allowed for management benefits to be observed (e.g., Cinner et al. 2005, Ferraro and Hanauer 2015). Whilst my simulations do not account for heterogeneous willingness of communities to engage in conservation (e.g., Knight et al. 2010), rules to disallow planning unit reinvestigations could be added in future explorations of this research area.

In my simulations, I assumed that planning unit costs were equal to the area of habitat within their boundaries. However, heterogeneous planning unit costs could influence the efficiency of iterative planning processes across multiple scales. Ideally, simulations including heterogeneous costs would also simulate the transition from coarse-resolution cost proxies during national-scale prioritisation, to higher-resolution and more accurate opportunity cost data that would become available during local-scale investigations to guide decision rules on which management units are converted to marine reserves.

The greater spatial overlap between initial regional assessments and final reserve systems when priorities are updated less frequently likely occurred because larger proportions of the initial and successive regional assessments were implemented before priorities were updated, restricting the spatial flexibility of each newly assessed set of priority planning units. In contrast, frequent updates, combined with earlier achievement of objectives for rarer features, increased the spatial flexibility and scope for departures from the initial assessment. Aside from these differences, spatial overlap between initial assessments and implemented reserves was relatively small, averaging about one-third for all scenarios. This supports the expectation that, even without new data becoming available during implementation, departures from initial assessments should be expected during the transition from assessment to implementation (Pressey et al. 2013). Inevitable application of management actions outside of prioritised areas by initiatives independent of systematic assessments (not incorporated in my simulations) would also drive further departure from initial assessments.

It is important to note that even though eventual actions depart spatially from initial priorities, this should not discount the process of planning (Laurian et al. 2004). Previous work has demonstrated the efficiency of systematic approaches over ad hoc expansion of reserve systems (Mills et al. 2012). My simulations add to this by demonstrating the value of frequent updates of systematic assessments in more quickly achieving objectives for rare (threatened, or otherwise prioritised) features.

Regional-scale conservation assessments (e.g., Beger et al. 2013, Mazor et al. 2014), usually produced with coarse-resolution proxies for biodiversity and cost data, should be viewed only as starting points to guide implementation of actions at local scales (Pressey et al. 2013). Whilst this perspective has been articulated (Groves et al. 2002, Green et al. 2009), there has been little investigation of how large the necessary departures from initial assessments might be, or how iterative planning across spatial scales might be undertaken. Real-world examples in which prioritisations are updated in response to implemented actions, and consequent achievement (or underachievement) of objectives, remain scarce (Pressey et al. 2013, Mills et al. 2015). In reviewing the extent to which marine conservation plans have been adaptive, Mills et al. (2015) found that some spatial plans did contain an explicit expectation that they would be revised or updated, for example in response to monitoring of the effectiveness of implemented marine protected areas, new data, or emerging threats. However, there was no explicit mention of frameworks in place for updating spatial prioritisations specifically.

Plans for KwaZulu-Natal, South Africa, and Kubulau District, Fiji, both specified a five-year review period (Jupiter and Egli 2011, Harris et al. 2012), though the Kubulau plan was revised within this timeframe after three years (Weeks and Jupiter 2013). The zoning plan for the Great Barrier Reef (GBR), Australia, proposed a minimum seven-year interval before the plan could be reviewed or amended, to provide stability for businesses and communities and allow sufficient time for ecological communities to respond to marine protected area establishment (Mills et al. 2015). While the Kubulau and GBR plans could be considered fully implemented (rather than part-way through an incremental application of actions, as I simulated here), the KwaZulu-Natal plan states that reserve targets are to be achieved over a 20-year period, using five-yearly updates to incorporate new information (Harris et al. 2012). In other instances where regional prioritisations have been updated whilst implementation is still ongoing, updates have been more ad hoc, occurring as a result of refreshed national commitments to planning, or funded interests from environmental non-governmental organisations (e.g., in Papua New Guinea; Government of Papua New Guinea 2015).

Overall, the benefits from more frequent updating of regional priorities aimed at guiding conservation actions were smaller than I anticipated. Based on my results, I identify two

potential benefits to more frequent updates: faster achievement of objectives for high-priority features, and greater potential to capitalise on previous groundwork in areas that are reinvestigated. The assumption of a trade-off between investment in updating regional assessments versus implementing them may in fact be inaccurate, given that, based on the skillsets required, the teams involved in either process are likely to differ. Arguments against more frequent updates to regional assessments include the difficulty in conveying the dynamic nature of conservation priorities to many stakeholders. For example, governments or NGOs who secure funding for priority areas may be committed to implementing actions within those areas alone. In this regard, my finding that less frequent updates do not significantly impact the time taken to achieve objectives should provide some encouragement.

My study represents first steps in understanding how to operationalise the iterative transition between regional priority assessments and implementation of local actions, and some of the factors that can influence the efficiency and effectiveness of this process. However, there remains much to explore in terms of other factors that may influence these outcomes. Future areas of research could involve examining how my results might change depending on the size of planning units, thematic resolution of ecological data, and the spatial heterogeneity or resolution of cost data that are incorporated and updated as regional plans are implemented through time.

Chapter 4

Identifying the strengths and weaknesses of conservation planning at different scales: the Coral Triangle as a case study

4 Identifying the strengths and weaknesses of conservation planning at different scales: the Coral Triangle as a case study

In Chapter 4, I evaluate the ability of conservation plans, developed at different jurisdictional levels (e.g., local, national), to adequately address multiple social and ecological scales. This chapter addresses further practical concerns of scale in conservation planning outlined in Chapter 1: specifically, understanding whether there is an optimal scale at which to plan. This work begins to tackle the notion of explicitly considering multiple scales in conservation planning and adds understanding of the different kinds of limitations associated with conservation planning at lower (e.g., local) versus higher (e.g., national) levels. On the basis of these limitations, this chapter makes specific recommendations that might increase vertical integration between planning levels and help to overcome their respective limitations. I conceptualised the research, collated and analysed the data, and wrote the chapter. Weeks and Pressey provided advice in conceptualising the research and assisted with structuring and editing the manuscript.

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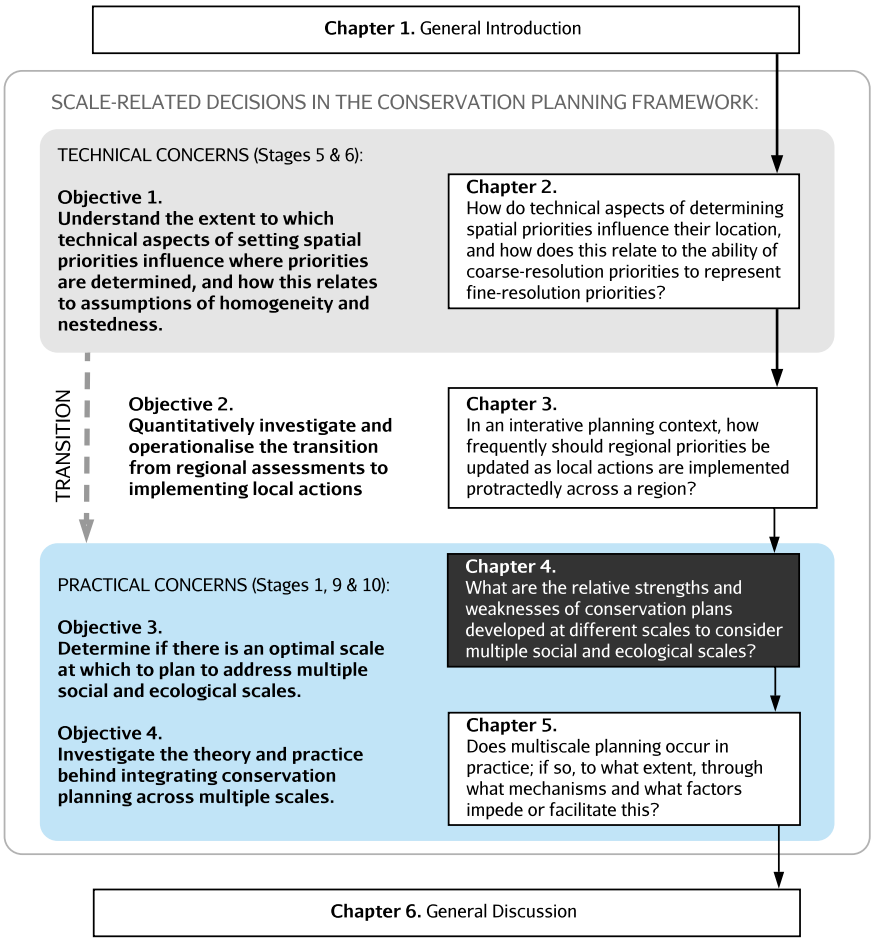

4 Identifying the strengths and weaknesses of conservation planning at different scales: the Coral Triangle as a case study

4.1 Abstract

Each year, hundreds of conservation plans are developed to direct limited resources towards conservation in priority areas. Conservation plans are developed at levels (defined here as points on a range of spatial extent) varying from global to local, but approaches to effectively integrate plans across scales remain elusive. To most effectively plan across multiple levels, the relative strengths and weaknesses of planning at different levels must be understood. Taking the Coral Triangle as a case study, I apply an adapted social-ecological systems (SES) framework to assess the 'scalar coverage' of conservation plans: the ability of plans developed at one level to adequately consider social and ecological components (i.e., resource units, resource systems, governance systems, actors) across all levels. No conservation plans I assessed had complete cross-level coverage. Plans most adequately addressed social and ecological components at the same level of planning and, to a lesser extent, lower levels. In line with previous literature suggesting social factors are most relevant at local levels, I found that local-level plans engaged with the greatest number of stakeholder groups, while higher-level plans more adequately addressed ecological components. To enhance overall cross-level coverage, I suggest that conservation plans focus not only on components at the same level, but also consider components at adjacent levels. Given that it appears more practicable for higher-level plans to consider components at lower levels, the onus should fall on higher-level planning to link to lower-levels. Achieving complete cross-level coverage will require vertical interactions between planning processes at different levels, and conceiving of planning processes across all levels as connected 'planning systems'. I demonstrate how an adapted SES framework can be utilised by conservation planners to assess the cross-level coverage of their own plans and to formulate appropriate conservation objectives to address social and ecological components at different levels.

4.2 Introduction

Problems of scale in conservation planning often relate to mismatches in scale between social and ecological systems. To account for these scale mismatches, it is clear that we need to consider multiple scales explicitly during any conservation planning process (Lengyel et al. 2014, Weeks et al. 2014, Guerrero and Wilson 2017). While various methods to consider multiple

scales in conservation planning have been suggested (as outlined in Chapter 1), currently, there remains uncertainty around the extent to which lower-level (e.g., local) plans are able to address higher-level (e.g., national) features and processes or vice versa. This information could be critical to understanding whether scaling down (whereby planning incorporates patterns or processes at progressively finer scales, within areas of interest identified initially at broader scales; Groves et al. 2002), or scaling up (whereby separate local planning processes are coordinated and placed within a broader context; Lowry et al. 2009), is more effective in integrating conservation planning across scales.

Applications of Ostrom's (2009) SES framework have frequently considered multiple social and ecological scales in environmental management (e.g., Guerrero et al. 2013, Cumming et al. 2015, Virapongse et al. 2016, Guerrero and Wilson 2017). Cumming et al. (2015) extended Ostrom's (2009) framework to consider multiscale and cross-scale interactions and feedbacks explicitly, and applied this adapted framework (hereafter, 'SES framework'; Figure 4.1) to retrospectively assess how these factors influence the spatial resilience of established protected areas. The question remains, however, as to how we can purposefully facilitate successful outcomes across multiple social and ecological scales in conservation planning, and whether there is an 'optimal' scale of planning to achieve this. The principle of subsidiarity, whereby decisions are made at the lowest institutional level capable of executing them sufficiently, has been suggested to guide community-based natural resource management generally (Marshall 2008), and in the Coral Triangle specifically (Fidelman et al. 2012). However, the capabilities of different institutional levels with respect to conservation planning remain unclear. To address these questions, we must first understand the relative effectiveness of conservation plans developed at different levels to address different components of an SES.

Here, I apply the SES framework depicted in Figure 4.1 (Cumming et al. 2015) to assess the ability of plans developed at one level to adequately consider social and ecological components that exist across levels from international to patch (hereafter, 'scalar coverage'). By components, I refer to resource units, resource systems, governance systems, and actors of an SES (Ostrom 2009); as Cumming et al. (2015) demonstrate, each component comprises different elements, which occur at a range of levels. For example, the ecological component of resource units can comprise the elements of species and habitats, which occur and function at different levels. The social component of actors can involve the elements of communities, governments, or non-governmental organisations (NGOs), which also operate at different jurisdictional levels.

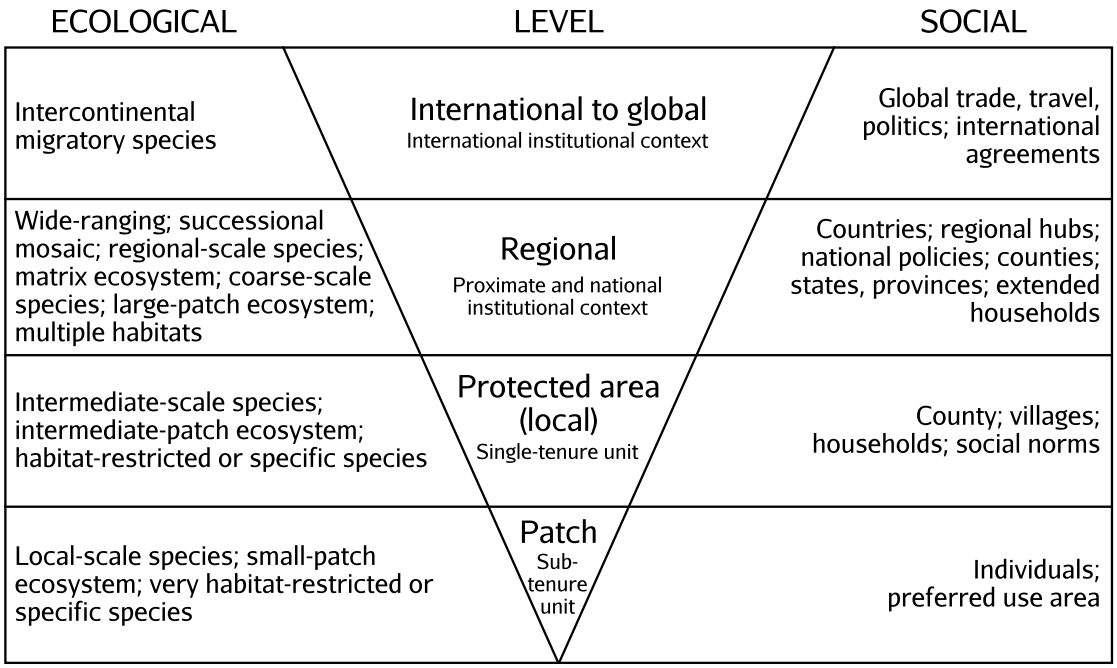

Figure 4.1 SES framework reproduced from Cumming et al. (2015) summarising patterns and processes at different social and ecological scales. The SES framework integrates Ostrom's (2009) SES framework and Poiani et al.'s (2000) ecological components of a functional landscape.

The adequacy with which ecological or social elements are considered in conservation plans can be inferred from analysing associated conservation objectives (Magris et al. 2014). Explicit conservation objectives are interpretations of broad conservation goals (representing values and beliefs) that guide the selection of areas for conservation action and serve as benchmarks to assess progress towards successful implementation or outcomes (Pressey and Bottrill 2009). For example, ecological objectives might be proportions of certain habitat types (representing the resource system component; e.g., "protect 20% of fringing reef habitats"). Socioeconomic objectives tend to be more qualitative in nature and, for example, could address community livelihood concerns (representing the actor component; e.g., "maintain sustainable livelihoods for artisanal fisheries").

I use the Coral Triangle region as a case study for my analyses. This region includes six countries: Indonesia, Papua New Guinea (PNG), Philippines, Solomon Islands, Timor Leste, and Malaysia. This region is of particular interest to conservation planners because of its global biodiversity importance coupled with highly varied socioeconomic and political complexities (Mills et al. 2010, Fidelman et al. 2012). All but two of the six countries (Malaysia and Timor Leste) have some form of decentralised natural resource governance (decision-making power devolved to local governments or customary clans; Fidelman et al. 2012), for which problems of scale mismatches are known to be acute (Mills et al. 2010).

Ideally, conservation plans would have a large scalar coverage, regardless of the level at which they were undertaken; for example, international-level plans would adequately consider social or ecological elements occurring at patch levels, or vice-versa. I consider this unlikely however, and hypothesise that: (1) plans conducted at any particular level will state objectives more adequately for socioeconomic and ecological elements at the same level (hereafter, 'intra-level objectives') than for those above or below the level of planning (hereafter, 'extra-level objectives'), (2) local-level plans will engage with a greater number of stakeholder groups and consider social factors in more detail than planning at higher levels, and (3) higher-level plans will more adequately address ecological elements compared to planning at lower levels. More adequate consideration of intra-level objectives is likely since social and ecological elements occurring at the same level of planning will be easier to conceive and relate to, compared to those of other levels (Ostrom et al. 1999, Wyborn and Bixler 2013). My second and third hypotheses reflect the frequently cited benefits of management stemming from local and regional levels, respectively (Mills et al. 2010, Gaymer et al. 2014).

4.3 Methods

I first identified marine conservation plans developed at different levels (from patch to international; Figure 4.1) from all six Coral Triangle countries through searching the peer-reviewed and grey literature. My criterion that defined a conservation plan was documentation that reflected the core purposes of conservation planning (Margules and Pressey 2000), considered here as including explicit socioeconomic and/or ecological objectives (qualitative or quantitative), and identifying spatial boundaries delineating area(s) for some form of management action or as priority areas for future action. Planning levels were categorised on the basis of jurisdictional level rather than spatial extent, since the extents of the same levels of jurisdictions (e.g., provinces) can differ greatly between Coral Triangle countries and it is at different jurisdictional levels that conservation actions are applied.

For each conservation plan (Table 4.1), I first extracted stated ecological and socioeconomic objectives, and identified the corresponding ecological and social elements of the SES framework addressed by each objective. The level of each ecological and social element addressed by each stated objective was then classified independently (see Figure 4.2 for an overview of the analytical process). The scalar coverage of each plan was determined by assessing the adequacy with which all SES levels and elements were addressed (detailed below). A plan with optimal scalar coverage would include and adequately specify socioeconomic and ecological objectives that represent SES elements occurring from patch to international levels. Case study assessments

were summarised and visualised in *R* (R Development Core Team 2016) using the 'fmsb' (v0.6.0; Nakazawa 2017) and 'ggplot2' (Wickham 2009) packages.

4.3.1 Assessing the adequacy with which ecological objectives are addressed

Each stated objective was first classified in terms of the relevant ecological elements and associated level(s) to which it pertained (Step 3, Figure 4.2). Classifications were conducted by myself and discussed with my co-authors for verification where there was uncertainty. I assigned these on the basis of Poiani et al.'s (2000) framework for biodiversity conservation at multiple scales. It can be difficult to assign an ecosystem or species to an exact level because region-specific life-history information is unavailable for most species; moreover, ecological features and phenomena operate at multiple scales (Levin 1992). Thus, Poiani et al.'s (2000) framework defines the extents of ecological scales generally, with overlapping values between levels to account for regional differences. Informative and specific descriptions of targeted ecosystems or species (e.g., species name, specific habitat information) were missing from many of the conservation plans I assessed. Ecological elements were therefore assigned on the basis of described habitats or processes, across the range of levels that may have been encompassed by those described. For example, a common ecological objective was, "Conserve 20% of shallow marine and coastal habitats (coral reefs, estuaries)". Since coral reef and estuarine habitats in these regions could span patch to local (protected area) levels (Figure 4.1; Poiani et al. 2000), this objective would be classified as addressing both patch and local levels.

Table 4.1 Summary of all conservation plans and associated reports collated for evaluations of scalar coverage.

Conservation plan	Plan level	Country	Lead planning organisation(s) †	Reference(s)
Kakarotan Island Mane'e	Patch	Indonesia	Kakarotan community	Cinner et al. (2005)
Muluk Village Traditional Closure	Patch	PNG	Muluk community	Cinner et al. (2005)
Nino Sanis Santana Marine National Park	Local	Timor Leste	Timor Leste Ministry of Agriculture and Fisheries, Northern Territory Government, Charles Darwin University	Edyvane et al. (2012)
Nusa Penida MPA	Local	Indonesia	CTC	Ruchimat et al. (2013)
Sinub Island Wildlife Management Area	Local	PNG	WI-O	Jenkins (2002)
Tubbataha Reef Natural Park	Local	Philippines	WWF, CI	Department of Environment and Natural Resources (2014)
Tun Mustapha Park	Local	Malaysia	WWF, UQ, Universiti Malaysia Sabah	Jumin et al. (2017)
Wakatobi Marine National Park	Local	Indonesia	TNC, WWF	Elliott (2001); Clifton (2011, 2013)
Choiseul Province Ridges to Reefs Protected Area Network	Regional	Solomon Islands	TNC, WWF, Live and Learn	Lipsett-Moore et al. (2010)
Isabel Province Ridges to Reefs Protected Area Network	Regional	Solomon Islands	TNC, WWF, WorldFish	Peterson et al. (2012)
Kimbe Bay MPA Network	Regional	PNG	TNC	Green et al. (2007b, 2009)
Lesser Sunda Ecoregion MPA Network	Regional	Indonesia	TNC	Wilson et al. (2011)
Raja Ampat MPA Network	Regional	Indonesia	TNC, WWF, CI	Agostini et al. (2012); Grantham et al. (2013)
Roviana and Vonavona Lagoons MPA Network	Regional	Solomon Islands	University of California Santa Barbara, Tiola Conservation Foundation, WWF, Christian Fellowship Church	Aswani et al. (2005)
Land-Sea Conservation Assessment for Papua New Guinea	Regional	PNG	PNG Conservation and Environment Protection Authority; UQ; TNC	Adams et al. (2017)
Ridges to Reefs Conservation Plan for Solomon Islands	Regional	Solomon Islands	Solomon Islands Ministry of Environment; James Cook University; TNC	Kool et al. (2010)
Coral Triangle Marine Protected Area System	International	All	WWF, TNC, CI, UQ	Beger et al. (2013)
Sulu-Sulawesi Marine Ecoregion Conservation Plan	International	Indonesia, Malaysia, Philippines	WWF	Dumaup et al. (2003)

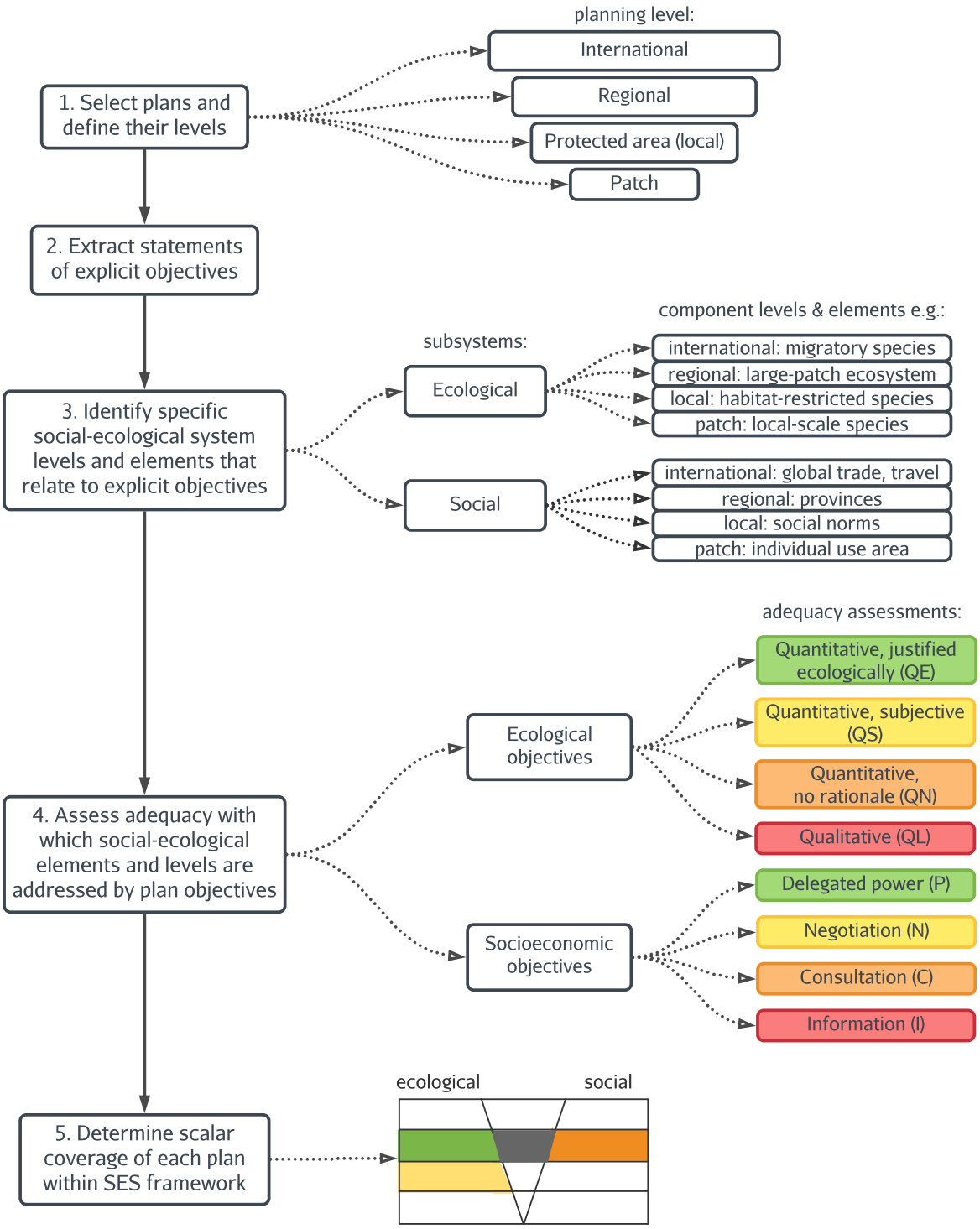

Figure 4.2 Overview of analytical process (adapted from Magris et al. 2014), depicting major steps in analysing each conservation plan against the SES framework. For ecological classifications, qualitative statements (QL) refer to statements of preferences and quantitative statements were grouped into three classes: no rationale (QN), subjective (QS), or justified ecologically (QE). For socioeconomic classifications, statements were classified to reflect the degree of stakeholder participation involved (adapted from Arnstein 1969, Pomeroy and Douvere 2008), ranging from information (I) to consultation (C), negotiation (N), and delegated power (P).

Each ecological objective was then assessed in terms of whether it was qualitatively or quantitatively articulated. Qualitative (QL) objectives were those that described the objective or target, without quantitative specification. An example is "include critical or unique sites such as areas with very high diversity, high levels of endemism or unique marine communities". Quantitative objectives involved numerical values when translating an ecological principle or estimating necessary amounts for conservation. Following Magris et al. (2014), quantitative objectives with no rationale (QN) lacked any explicit justification; e.g. "30% of each shallow marine habitat (coral reefs, mangroves, seagrass, and estuaries) and its sub-class". Subjective quantitative (QS) objectives were based on the opinions of experts, stakeholders, the authors, or on previous work or models but without explicit quantitative ecological justification; e.g. "aim to include at least three representative examples of each habitat type in different locations, distributed over a large area to reduce the chance all would be negatively impacted by a single environmental or anthropogenic event at the same time". Ecologically justified quantitative (QE) objectives were based on empirical data, ecological theories, or models employed with supporting ecological information; e.g. "aim for MPAs to be spaced 100-200 km apart to maintain genetic connectivity … (McLeod et al. 2009)". In our evaluations, QE represented the most adequate level of addressing ecological objectives and QL the least adequate.

4.3.2 Assessing the adequacy with which socioeconomic objectives are addressed

The socioeconomic objectives of each plan were categorised in terms of the relevant social elements and associated level(s) that each addressed according to the SES framework (Figure 4.1). Socioeconomic objectives were often stated ambiguously, thus potentially spanning multiple levels of actors and governance. For example, "protect high potential tourist sites", could address socio-political and economic components at multiple levels (e.g., tourist satisfaction occurring at patch and local levels; national revenue generated at regional levels; Cumming et al. 2015). Because of this ambiguity, and because authentic stakeholder involvement underpins the achievement of any conservation objective in societal settings (Pomeroy and Douvere 2008), my assessment of socioeconomic objectives focused on the extent of stakeholder engagement across the different levels. I considered stakeholder groups as engaged in my assessments only if stated socioeconomic objectives explicitly referred to them, or engagement with them was explicitly described in the planning documents or reports.

To assess the degree of stakeholder engagement in planning processes related to stated socioeconomic objectives, I used a classification scheme based on Arnstein's (1969) ladder of citizen participation, adapted in the context of Pomeroy and Douvere's (2008) review of stakeholder participation in marine spatial planning. Stakeholders are defined here as, "individuals, groups or organisations who are, in one way or another, interested, involved or

affected (positively or negatively) by a particular project or action toward resources use" (Pomeroy and Douvere 2008). I reviewed all stated socioeconomic objectives in each conservation plan to identify the relevant stakeholder groups and assign respective levels: national government, local government, international NGOs, local NGOs, remote academics, local academics, mining and shipping industry, tourism sector, aquaculture sector, commercial fisheries, subsistence fisheries, traditional leaders, and local communities. These stakeholder groups and corresponding levels were then categorised in terms of their degree of participation: 'information' (I), essentially nonparticipation, where flow of information is unidirectional from the managing institution(s) to other stakeholders only; 'consultation' (C), where stakeholders are consulted but whose ideas and feedback are not necessarily considered in the planning process; 'negotiation' (N) where there is genuine dialogue and negotiation between stakeholders but final decision-making still rests with the managing institution(s); and 'delegated power' (P), where full decision-making power is delegated to all stakeholders involved in the planning process.

4.4 Results

I identified a total of 18 conservation plans: two each at the patch and international levels; six at the local level; and eight at the regional level (Table 4.1). Five plans were from Indonesia, four each from PNG and Solomon Islands, and one plan each from Malaysia, Philippines, and Timor Leste.

4.4.1 Strengths and weaknesses of plans to address intra-level versus extra-level objectives

In general, plans included intra-level ecological objectives and those below the level of the plan (hereafter, 'sub-level'). There was only one exception: a case study that did not address ecological objectives at all sub-levels (SI National Plan, Figure 4.3a; Kool et al. 2010). Across all case studies, intra-level ecological objectives were addressed more adequately than, or just as adequately as, sub-level objectives from the same plan (Figure 4.3a). In the few instances (n = 3) where higher-level (i.e., supra-level) ecological objectives were included, these were addressed least adequately (qualitative; Nino Sanis, Sinub, and PNG National Plans; Figure 4.3a). Socioeconomic objectives only ever referenced engagement with stakeholder groups at the same level of planning or below, or none at all (Figure 4.3b).

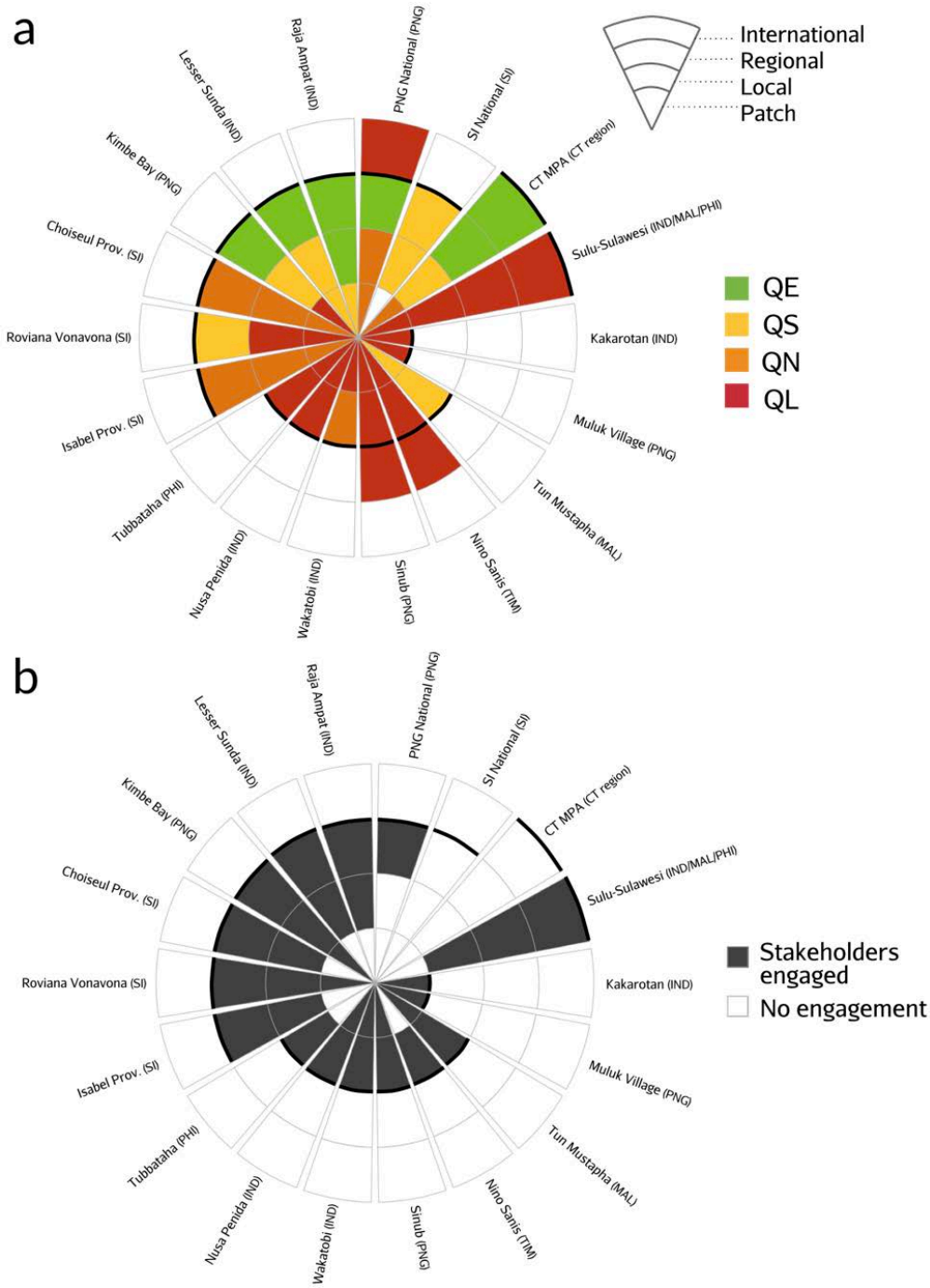

Figure 4.3 Differences in ecological and social levels addressed by conservation plans developed at different levels. (a) The maximum ecological adequacy achieved is shown at each SES level, based on all ecological objectives stated in each plan. Classifications of ecological adequacy: qualitative (QL), quantitative with no rationale (QN), quantitative subjective (QS), and quantitative ecologically justified (QE). (b) Presence of stakeholder engagement at each SES level across conservation plans developed at different levels. In both parts, inner to outer plot circles represent the progression of SES levels, from patch to international, respectively. Solid black lines indicate respective planning levels for each conservation plan assessed, ordered from patch to international case studies (clockwise, from 80°). Diversity and extent of stakeholder engagement

was highly variable within and across SES levels; thus, a more detailed representation is shown in Figure 4.4.

4.4.2 Strengths and weaknesses of plans in addressing ecological and socioeconomic objectives

Of all plans assessed, 16 included both ecological and socioeconomic objectives. The remaining two plans (that addressed ecological objectives only) were conducted to review existing reserve systems and update spatial priorities (Kool et al. 2010, Beger et al. 2013). Plans typically had the greatest number of ecological objectives formulated at the intra-level (see Appendix 3 Table A3.1 for details). In contrast, socioeconomic objectives were generally most numerous at the local level, irrespective of the level at which the plan was developed (Table A3.1). While the majority of ecological objectives were stated qualitatively across all plans, there was an inverse relationship between percentage of qualitative objectives and planning level (i.e., percentage of qualitative objectives decreased with increase in planning level; 100%, 83%, 52%, and 50%, from patch to international levels). With the exception of patch-level plans, there was variation in the adequacy with which intra-level objectives were addressed by different plans developed at the same level (Figure 4.3a). Ecological objectives were most adequately addressed (QE) at higher levels of planning (i.e., regional and international; Figure 4.3a).

I found that socioeconomic objectives formulated at different levels were concerned with resource use by, and benefits to, the stakeholder groups that were most relevant at each level. For example, socioeconomic objectives at local levels were primarily concerned with resource use by and benefits to local communities (e.g., "protect areas of cultural importance"), while those at regional levels involved broader economic concerns, often relating to resource use by and benefits to industry (e.g., "support low-impact environmentally friendly industries that are compatible with MPAs"). All socioeconomic objectives across all conservation plans were stated qualitatively.

Plans regularly included multiple ecological objectives at a single level. For example, in the Wakatobi Marine National Park (Indonesia) plan, two ecological objectives addressed local-level elements, with different degrees of adequacy: (1) "effective management of coral reefs, cetaceans, together with undefined ecologically valuable marine species" (QL); and (2) "maintaining existing levels of hard coral cover which are estimated to be around 35-40%" (QN). To highlight the *potential* for plans to address SES elements at different levels, I illustrate the maximum degree of adequacy with which ecological objectives were addressed at each level (Figure 4.3a).

In general, I found that across my collated case studies, governments had decision-making power, non-governmental institutions and industries tended to be negotiated with, and local communities and subsistence fisheries were more variably informed, consulted or negotiated

with (Figure 4.4). Exceptions to the latter trend occurred in conservation plans from countries with customary resource ownership, where local communities had delegated power (e.g., patch- and regional-level plans in Papua New Guinea and the Solomon Islands; Figure 4.4 and Appendix 3 Table A3.2). Local-level plans engaged with a more diverse range of stakeholder groups than any other level of planning, but the degree of engagement varied between different groups. At the lowest level (i.e., patch), engagement occurred with the least number of stakeholder groups and was less varied compared to other levels of planning (Figure 4.4).

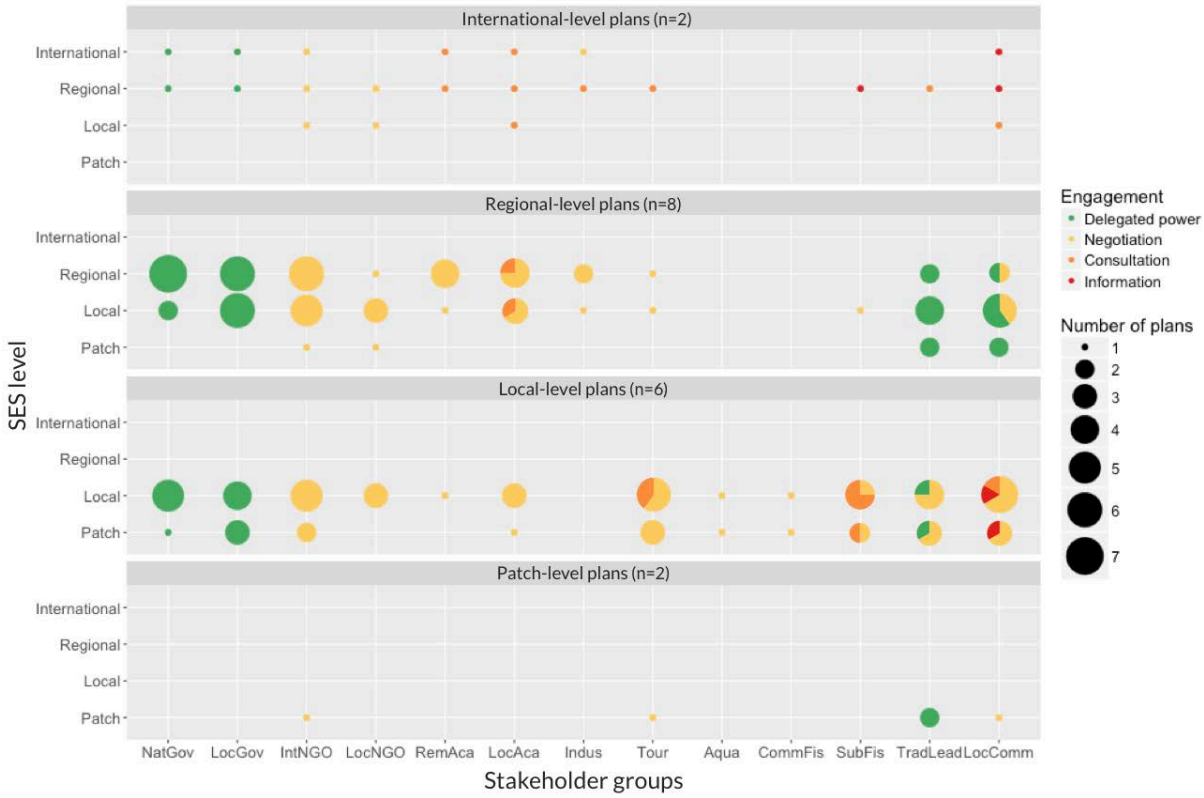

Figure 4.4 Differences in breadth and extent of stakeholder engagement across the four planning levels. Classification used to assess stakeholder engagement: information (I), consultation (C), negotiation (N), and delegated power (P). Pie-chart bubbles reflect variation in engagement of stakeholder groups between case studies at the same planning level, showing the proportions of the number of case studies and their respective degrees of engagement. Stakeholder groups (ordered by approximate scale of power or operation, from left to right): national government (NatGov), local government (LocGov), international NGO (IntNGO), local NGO (LocNGO), remote academic (RemAca), local academic (LocAca), shipping and mining industry sector (Indus), tourism sector (Tour), aquaculture sector (Aqua), commercial fisheries (CommFis), subsistence fisheries (SubFis), traditional leaders (TradLead), local communities (LocComm).

4.5 Discussion

None of the conservation plans I evaluated had complete scalar coverage. Nevertheless, regional-level plans achieved the greatest scalar coverage, in terms of both social and ecological components. Given that I found no one level of planning that adequately considered all social and ecological levels, my results provide insights into how we can more comprehensively plan for multiscale SES.

My hypothesis that conservation plans will better address intra-level socioeconomic and ecological elements compared to those at extra-levels was substantiated to a certain extent by my results. While all plans I evaluated consistently addressed intra-level objectives more adequately than extra-level objectives, 10 plans addressed sub-level objectives equally well. My second hypothesis was similarly validated with qualifications. Rather than finding that local-level plans considered social factors in more detail, I found that plans developed at any level (except for patch-level plans) consistently considered the greatest number of socioeconomic objectives at local and regional levels. I also found that local-level plans engaged the greatest number of stakeholder groups. However, the extent of engagement was highly variable across stakeholder groups in all levels of planning except patch-level. Finally, my hypothesis that regional-level plans would address ecological objectives more adequately than local- and patch-level plans was validated.

4.5.1 Apparent strengths and weaknesses of lower-level planning

My finding that socioeconomic objectives were generally greatest in number at local levels likely reflects the more directly, and therefore easily, observable social factors and impacts at this level compared to higher ones (Ban and Klein 2009). The framing of socioeconomic objectives at local levels also better captured, relative to other levels, social elements representing lower-level actors and governance systems (e.g., socioeconomic concerns relevant to local resource users and communities).

While equitable engagement across all key stakeholders who are not final decision-makers is generally ideal (Pomeroy and Douvere 2008), the extent of engagement that is most appropriate across different stakeholder groups is complex and highly context-specific (Gopnik et al. 2012, Fox et al. 2013, Sterling et al. 2017). Nonetheless, since my results suggest that local-level plans are better able to engage a greater number of stakeholder groups than other levels, it is clear that some form of local-level planning is essential. Conversely, patch- and local-level plans generally did not address ecological elements at higher levels and were less able to adequately address

these at lower levels (though they were able to address intra-level ecological objectives in a subjective quantitative way), suggesting that local-level planning alone will not be sufficient.

Weaknesses in planning at different levels can be described as either conceptual or technical. Lower-level plans generally did not include objectives for ecological elements at higher levels, suggesting a conceptual limitation, in the ability of planners at lower levels to perceive ecological features and processes that occur at broader extents (e.g., Wyborn and Bixler 2013, Charlie et al. 2013). A typology of scale mismatches between spatial scale (in our context, representing ecological elements occurring at regional and international extents) and institutional level (here, of local protected areas), suggests this type of mismatch produces a lack of local management capacity to address ecological elements at larger extents (Maciejewski et al. 2015). Technical limitations were suggested by the poor adequacy with which ecological elements were addressed, most evident in lower-level plans. This technical limitation of lower-level planning, typically led by local NGOs in Coral Triangle countries, relates to the lack of technical expertise and capital resources commonly faced by these organisations (Green et al. 2011).

4.5.2 Apparent strengths and weaknesses of higher-level planning

Regional- and international-level conservation plans demonstrated greater capacity to adequately address ecological elements. This is likely because organisations that are leading planning at higher levels (in Coral Triangle countries, typically international NGOs) tend to have greater access to technical resources and expertise, which are necessary to incorporate empirically justified ecological objectives (see Kool et al. 2010, Agostini et al. 2012, Beger et al. 2013). As with local-level plans, higher-level planning was important in addressing socioeconomic elements at the same (i.e., regional and international) levels (e.g., concerning development of industries). This suggests that conservation planning across both lower and higher levels is necessary to ensure that the scalar coverage of social components is maximised for improved outcomes across all levels (Ban et al. 2013).

A common technical limitation of higher-level planning is the ability to obtain fine-resolution data necessary to adequately address ecological elements at local levels (Mills et al. 2010). While the higher-level conservation plans I evaluated frequently mentioned this caveat (e.g., Green et al. 2007, Lipsett-Moore et al. 2010, Beger et al. 2013), I still find that these plans had greatest scalar coverage. However, while my assessment scheme for ecological adequacy favours quantitative objectives, I was unable to determine whether the data used to address stated quantitative objectives were appropriate or accurate. It is thus possible that the adequacy with which higher-level plans considered regional and international elements is overestimated.

Nevertheless, my results suggest that it may be possible to overcome the technical and conceptual limitations to some degree in higher-level planning.

The greater scalar coverage achieved with higher-level plans may be a reflection of the hierarchical scales that relate to management, social networks, and knowledge (Cash et al. 2006), where higher levels inherently contain within them all entities at lower levels (e.g., Lebel et al. 2008). Moreover, national institutions are supposed to be designed to include and affect factors relevant to lower levels (e.g., flows of capital, economic policies). In contrast, local institutions are seldom, if ever, capable of coping with developments occurring at national or international levels (e.g., civil war, international trade markets; Ostrom et al. 1999).

4.5.3 Recommendations to overcome limitations associated with single-level planning

From my assessment of conservation plans against the SES framework, the need to plan at multiple levels and integrate across these is evident. Planning at regional levels or higher appears to have greater scalar coverage than plans developed at lower levels, suggesting some support for scaling down processes, where sequential planning occurs at progressively lower levels. Because the limitations of scaling down primarily relate to the lack of consideration of local contexts (Gaymer et al. 2014), I suggest that, rather than strictly scaling down, a more effective strategy to integrate planning across scales may involve initiating planning at a high level (e.g., regional or international) and then iteratively cycling between higher and lower levels of planning (e.g., Pressey et al. 2013). This would ensure that higher-level plans do not proceed to identify priorities and conservation interventions without consideration of relevant local conditions.

There is now significant evidence demonstrating that scale mismatches often decrease functioning of SES (Epstein et al. 2015): the larger the magnitude of mismatch, the more likely is a greater loss in system resilience (where resilience is defined as a loss of critical components or functioning, or where inefficiencies occur; Maciejewski et al. 2015). Given the clear mismatch between the different planning levels and their ability to consider elements occurring at other levels, one way to overcome their respective limitations is to ensure that planning processes interact and effectively inform one another across levels. This would increase the alignment in scale-related perspectives between different planning levels, which has been asserted as a means to overcoming mismatches between spatial scale and institutional level (Maciejewski et al. 2015).

My results suggest that the limitations of planning at both lower and higher levels are conceptual and technical in nature, but differently so. Lower-level plans are constrained by their ability to conceptualise as well as technically address elements at higher levels, while higher-level plans are primarily limited by their capacity to technically address lower-level ecological elements and

conceptualise socioeconomic objectives at higher levels. Conceptual and technical limitations might be overcome by different types of interactions between planning processes conducted at different levels. Conceptual limitations could be overcome through workshops designed to share and learn from differences in perspectives between planning levels (e.g., identifying socioeconomic and ecological objectives that are important at different levels). Technical limitations might be mitigated through exchanges of data, information or individuals possessing the appropriate expertise between processes. Such information-sharing between levels needs to be institutionalised for long-term success (Berkes 2009). These interactions across planning levels would be facilitated if planners have greater awareness of conservation plans developed at other levels, in the same region and elsewhere (to broaden their perspectives to understand the different conceptualisations of socioeconomic and ecological objectives at other levels). This could be accomplished through a standard database system in which all conservation plans and pertinent planning information (e.g., objectives, data used, socioeconomic and ecological context, implementation strategies) are recorded.

Integrations of SES theory and conservation planning remain limited, mostly theoretical, and largely motivated from a social science perspective (Ban et al. 2013, Mills et al. 2014, Bodin 2017). I argue that the SES framework can also be used as a practical tool for conservation planners to understand the social and ecological elements that may be most relevant to the planning level being undertaken, and elements pertinent at extra levels that should also be considered. The SES framework could also be used by planners to assess the scalar coverage of plans (past or in progress) and identify weaknesses in considering intra-level and extra-level social or ecological elements, as I have done here. This has relevance to multiple stages of the conservation planning process; beyond setting conservation objectives, the SES framework can also be used to inform the initial scoping stage of planning processes, as well as evaluating planning outcomes (Pressey and Bottrill 2009).

The main difficulty I found in applying the SES framework to evaluate the strengths and weaknesses of conservation plans was in allocating appropriate levels to the different social and ecological elements addressed by conservation objectives. As these elements do not fall into discrete categories of spatial or jurisdictional scales, they need to be assigned to numerous levels. I also found assessing the adequacy of socioeconomic objectives to be challenging because, while stakeholders do represent different jurisdictional levels, they can often operate on and influence multiple other levels of an SES. This difficulty was largely related to the ambiguous and qualitative way in which socioeconomic objectives were commonly articulated in conservation plans. A potential improvement to the framework in future applications would involve more explicit inclusion of the elements of governance systems relevant at each SES level. Governance

is an essential constituent of successful conservation (Armitage et al. 2012), and it is likely that different governance systems will inherently better address different SES levels (Termeer et al. 2010). A more explicit understanding of governance systems operating at different levels may reveal insights into how the scalar coverage of conservation plans can be influenced from a multilevel perspective (e.g., Lebel et al. 2008).

4.5.4 Limitations

My study was limited by the number of case studies found, particularly at the patch and international levels. A number of factors contributed to this. Malaysia and Timor Leste each had only one case of systematic conservation planning at the time of my analysis (Edyvane et al. 2012, Jumin et al. 2017). In the Philippines, where there are upwards of 1,200 marine protected areas (Horigue et al. 2012), only one case study could be identified. This is likely due to a combination of my criterion for defining conservation plans as requiring documented objectives, and the fact that established protected areas in the Philippines are typically ad hoc, patch-level decisions that are not well documented (Alcala and Russ 2006). My analyses are also limited in that my evaluations relied on planning documentation and reports alone. Causality regarding the presence or absence of objectives had to be inferred, as well as the authenticity of reported stakeholder engagement; further empirical investigation with planners involved in conservation plans was beyond the scope of this study. Nonetheless, plan documentation plays a vital role in the accountability and transparency of the systematic conservation planning process (Margules and Pressey 2000) and the tracking of progress towards achieving objectives, and should thus represent a reliable source on the intentions of any planning process.

4.6 Conclusions

Though no conservation plans had complete scalar coverage, higher-level plans demonstrated the greatest capacity for scalar coverage. For conservation planning to be successful in adequately considering social and ecological components that exist across all scales, I suggest that, in addition to primarily addressing intra-level elements, planners should seek to identify and specify objectives for relevant elements at adjacent levels. My approach of using the SES framework to understand the extent to which conservation plans adequately address multiple social and ecological scales can facilitate this. The responsibility for ensuring that planning processes are vertically integrated across planning levels lies with the individuals and organisations leading these processes. Achieving this requires, at minimum, two critical ingredients. The first is an awareness of other plans developed at different levels, as well as their respective strengths and weaknesses in addressing SES elements (as I have demonstrated and identified here). Second, further research is needed to understand how planning processes at different levels can and do

inform one another over time to overcome the technical and conceptual limitations associated with planning at a single level.

Chapter 5

Scalar capital as ingredient of success in multiscale conservation governance: evidence from Melanesia

5 Scalar capital as ingredient of success in multiscale conservation governance: evidence from Melanesia

In Chapter 5, I explore the theory behind multiscale conservation planning and then analyse how it is conducted in practice. This chapter addresses the remaining practical concerns of scale in conservation planning outlined in Chapter 1: specifically, investigating the potential to conduct conservation planning across multiple scales to more explicitly consider scale. This work draws upon multiple fields, such as collaborative governance, policy and social network theories and makes a number of novel contributions to current theoretical and empirical understanding of integrating conservation planning across multiple scales. Finally, this chapter also builds on the insights gained from Chapter 4, where I demonstrated the respective limitations of lower- and higher-level planning and the crucial need for vertical integration across planning levels. Chapter 5 identifies specific avenues through which vertical integration can and does occur.

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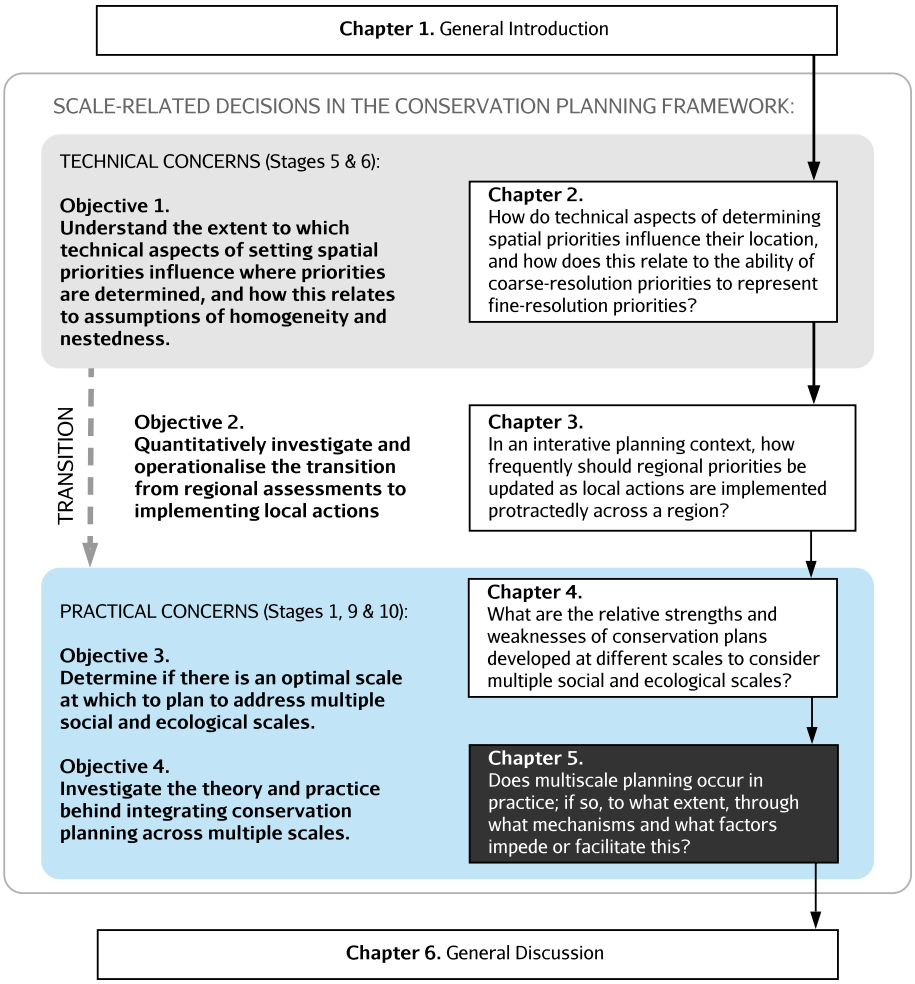


5 Scalar capital as ingredient of success in multiscale conservation governance: evidence from Melanesia

5.1 Abstract

Problems of scale abound in the governance of complex social-ecological systems. Conservation governance, for example, typically occurs at a single scale, but needs to inform governance and action at other scales in order to be effective in conserving multiscale social and ecological components. This process is conventionally conceived as unidirectional – comprising either scaling down or scaling up – in the way it both exploits and creates natural, social, human, institutional, and financial capital. Here, I analyse multiscale conservation governance and the factors that impede or facilitate its effectiveness. Comparative analysis of conservation planning in Papua New Guinea and the Solomon Islands, through in-depth document review, key informant interview and participant observation, reveals limited evidence of unidirectional processes. Instead, I observe multidirectional scaling pathways, cultivated by the following six scale-explicit characteristics of effective conservation governance: (1) multiscale understanding, (2) scale jumping, (3) leadership characteristics, (4) stakeholder engagement, (5) policy frameworks, and (6) institutional settings. While the latter four are familiar concepts, though not always recognised as explicitly scalar, we know little about the first two attributes of conservation governance. Based on this novelty, I propose a new form of capital – 'scalar capital' – to complement natural, social, human, institutional, and financial capitals as both input and outcome of effective conservation governance. I find that scalar capital facilitates different flows of resources (data, conservation objectives, practitioner experience, institutional support, and funding) in multiple directions. Critically, I present empirical evidence that conservation governance can foster scalar capital to improve outcomes across multiple scales.

5.2 Introduction

Difficulties in understanding scale have pervaded the fields of environmental governance, management, and planning since their inception (Margules and Pressey 2000, Termeer et al. 2010, Cumming et al. 2015, Morrison 2017). Complications with scale are manifest in a multitude of ways, though all are in some way a reflection of our limited understanding of social and ecological processes that operate and interact differently between scales. I use the term 'social processes' to refer to the ways in which individuals and groups act and interact to construct and adapt relationships and behaviour. These ways are continually modified and

refined through, for example, social learning and memory, institutional and organisation inertia and change, social networks, and adaptive capacity and governance (Folke 2006). I use the term 'ecological processes' to refer to the biological, chemical, and physical actions and interactions that occur between organisms and their environment. Examples are dispersal and movement of species across landscapes or seascapes through habitat connectivity (Maciejewski and Cumming 2016), environmental degradation and impacts on species community composition, and competition-colonisation dynamics (Driscoll et al. 2013).

Though my focus here is on conservation planning, the concepts I explore ultimately concern the multiscale governance of complex social-ecological systems. These problems essentially relate to the problem of fit in complex social-ecological systems (Folke et al. 2007, Bodin et al. 2014), which has historically yielded adverse outcomes for environmental governance (Crowder et al. 2006). A better understanding is crucial for the successful management of these systems (Epstein et al. 2015) and, because any ecological or social system operates over or within a range of spatial, temporal, and organisational scales (Cumming et al. 2017), this need has broad-reaching relevance across many research areas. For example, theories of collaborative governance (Ansell and Gash 2007), policy and social networks (Sandström and Carlsson 2008), and advocacy coalition (Weible et al. 2009) all consider interactions and connections between public and private stakeholders or policies that inevitably exist at multiple jurisdictional and institutional scales. Despite these considerations, explicit treatment of scale in these theoretical frameworks is a relatively recent development (Weible et al. 2011, Bodin 2017). Of particular relevance is the scale-explicit idea in social network theory of scale-crossing brokers, described as a social network position that bridges specifically across ecological scales (Ernstson et al. 2010). There is now empirical evidence of the value of scale-crossing brokers (Cohen et al. 2012, Guerrero et al. 2015b, Reid et al. 2016) in facilitating links between separate levels, along with disparate sectors of society (e.g., policymakers, communities, and researchers). Similarly, related fields of social-ecological systems and ecosystem services have also begun to more appropriately, and explicitly, conduct analyses at multiple scales of assessment (Scholes et al. 2013) or against a multiscale framework (Cumming et al. 2015).

5.3 Multiscale conservation governance

There is now much evidence to suggest that conservation governance needs to explicitly consider and integrate across multiple scales as a response to scale mismatches (Morrison 2007, Scholes et al. 2013, Cumming et al. 2015). However, despite frequent calls for integration across scales, conservation scientists, policymakers and practitioners have yet to define explicitly what this means or demonstrate how they should approach it. In order to assess the extent to which

multiscale conservation governance is occurring, and whether it does in fact lead to improved outcomes, we first need to define it. I propose that multiscale conservation governance occurs where conservation planning processes undertaken at different scales effectively inform one another and consequently, result in improved outcomes compared to processes undertaken at a single scale or at multiple scales without informing one another. Given that hundreds of conservation plans are developed every year (Álvarez-Romero et al. in press), more effective and deliberate planning across multiple scales could improve conservation outcomes and achieve greater impact with the limited resources available for conservation.

The conservation planning literature largely conceives multiscale governance as a dichotomy. Scaling down (Figure 5.1a) assumes modification of designs generated at higher jurisdictional levels to include local objectives and preferences as planning is done at progressively lower jurisdictional levels (Mills et al. 2010). In contrast, scaling up (Figure 5.1b) refers to attempts to coordinate and place separate locally driven initiatives in a higher jurisdictional context (Horigue et al. 2015). These opposing trajectories are associated with two main approaches through which conservation planning has occurred: 'top-down' centralised management or 'bottom-up' decentralised management (Ban et al. 2011). Both scaling down and scaling up have been advocated in the literature, based on their respective benefits. Top-down planning is advantageous because it can incorporate wider perspectives, such as consideration of connectivity and complementarity between biodiversity features, possible only at higher jurisdictional levels and correspondingly larger extents. This perspective leads to planning initiated at high levels, with progressive refinement through scaling down (Ban et al. 2011). An alternative perspective is that the advantages of bottom-up planning, including local stakeholder engagement, buy-in, and compliance (Gaymer et al. 2014), call for planning to be initiated at local levels, and scaled up to incorporate higher-level perspectives. Other attempts to integrate planning across scales have involved the amalgamation of different scales into a singular static assessment (Bombi et al. 2013). This is problematic, however, because the assessment still occurs at a single scale of analysis, maintaining the limitations associated with single-scale assessments (Lemos and Agrawal 2006).

I use the term 'scalar pathway' to describe the movement of different directional flows across multiple jurisdictional levels over time (Figure 5.1). Scaling-up or scaling-down pathways imply that planning processes inform one another unidirectionally through time (Figure 5.1a,b). It remains unclear in the field of conservation planning whether this perceived dichotomy actually exists in the real world, or whether other modes of scalar pathways (e.g., Figure 5.1c) occur. Public policy scholars resolved a similar argument in the 1980s by merging the best attributes of the bottom-up and top-down approaches, with the explicit distinction of applying this

combination of approaches to a longer timeframe than was the case in most policy implementation research (Sabatier 1986). While conservation practitioners recognise the complementary advantages of scaling up and scaling down (Gaymer et al. 2014), no study has offered ways to operationalise cycling between multiple scales of planning. I argue here that multiscale conservation planning likely requires more flexible scalar pathways, beyond unidirectional scaling up or scaling down.

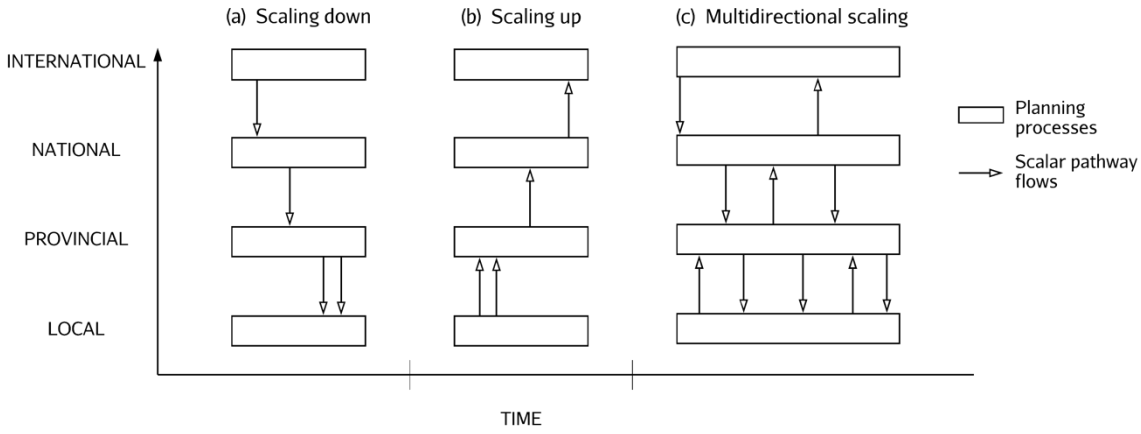


Figure 5.1 Conceptual representations of directional movement (e.g., of planning resources, such as data or funding) across multiple jurisdictional levels over time. Scaling down (a) and scaling up (b) represent the dichotomy prevalent in the conservation planning literature. Scaling down begins at higher jurisdictional levels, sometimes informed by international agreements, and involves modification of national- or provincial-level designs to include additional objectives and preferences as planning is subsequently adapted to lower levels. Scaling up moves in the reverse direction, whereby higher levels of planning inform and contextualise separate initiatives initiated at lower levels. National governments frequently co-opt these to calculate country-level progress towards achieving spatial targets set by international agreements. I observed a more realistic archetype of planning across multiple scales, named here multidirectional scaling (c). In this mode, movement between processes can occur in multiple directions, and planning processes can occur simultaneously at different levels and inform subsequent planning at higher or lower levels, occasionally bypassing the adjacent level. Tick marks on time axis denote separate timelines for each archetype.

Understanding of the factors that influence successful outcomes in conservation planning has typically been limited (Ferraro and Pattanayak 2006). To address this, Bottrill and Pressey (2012) proposed an evaluation framework adapted from Scoones' (1998) sustainable livelihoods framework, which views different types of capital (natural, financial, social, human, and institutional) as either a resource or product of investment (Table 5.1). I draw from Bottrill and Pressey's (2012) framework (hereafter, 'evaluation framework') to assess the relative success of each scalar pathway. Relative success was evaluated with respect to reported and perceived gains

in the different types of capital in each planning process that comprised the scalar pathways, and how these gains related to factors that specifically facilitated multiscale planning (e.g., where planning processes effectively informed those at other levels across each pathway).

Any efforts to foster multiscale planning will require better understanding of three central elements: (1) how planning at different levels can and do inform one another, (2) overall scalar pathways through which multiscale planning occurs in practice, and (3) factors that can impede or facilitate multiscale planning in particular contexts. Analysing these elements will elucidate how the effectiveness of multiscale planning can be influenced by specific socio-political conditions. To understand these three central elements, I evaluated conservation planning developed at different levels from Papua New Guinea (PNG) and the Solomon Islands (SI) (Figure 5.2; 14 conservation plans in total, 10 from PNG and 4 from SI). Vertical integration and coordination of conservation planning across jurisdictional levels is paramount in this region, due to the presence of customary governance regimes that necessitate local-level involvement in environmental planning. Nevertheless, though conservation planning has occurred at multiple scales in these countries (e.g., Smith et al. 2002, Green et al. 2007, Kool et al. 2010), often with the same organisation leading multiple planning processes (a single organisation, The Nature Conservancy, led all except one of the planning processes I evaluated), planning has not been deliberately multiscale.

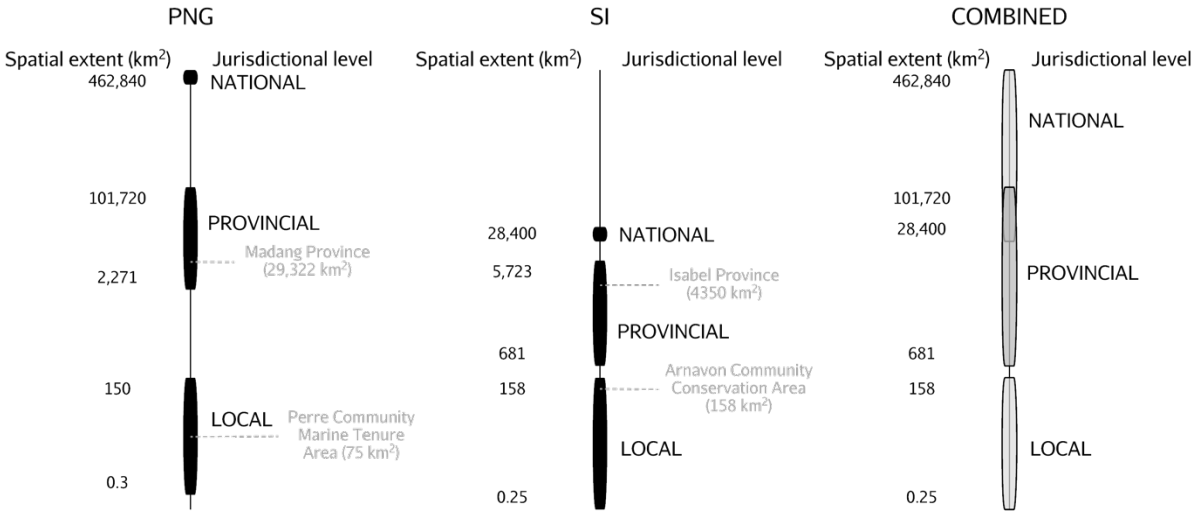


Figure 5.2 Relationships between spatial extents and jurisdictional levels in the two study regions: Papua New Guinea (PNG) and Solomon Islands (SI). Jurisdictional levels are points on the jurisdictional scale (Cash et al. 2006). Note additional jurisdictional levels exist in PNG; however, for the purposes of consistent comparison between case studies, I focused only on levels common to both countries.

5.4 Methods

5.4.1 Identifying scalar pathway case studies

Planning processes (n = 14) that comprised each scalar pathway case study (n = 3) were identified based on geography (Papua New Guinea and Solomon Islands) by searching available peer-reviewed and grey literature (Appendix 4 Table A4.1). Each scalar pathway was first constructed through process-tracing, based on a comprehensive review of all collated planning documentation on specific events, places, connections (between planning processes) and timelines explicitly mentioned. These initial pathways were subsequently refined by showing to and discussing with key informants during interviews, then completed and corroborated based on all information collected from the key informant interviews.

5.4.2 Document review

A documentary review was conducted on all reports, management plans and other relevant scientific or governmental publications (n = 38; Appendix 4 Table A4.1) on each planning process included in my case studies. Documentation was identified through searches of peer-reviewed and grey literature, as well as additional documents received from key informants. Documents were analysed for information such as, spatial extent, timeline, planners and stakeholders and specific planning context involved, general planning process undertaken, any known planning outcomes, and any reported connections, and the nature of these connections, to other planning processes.

5.4.3 Key-informant interviews and participant observation

To triangulate collated and collected data, in-depth and confidential interviews and participant observations were undertaken with key informants who were involved in planning processes across different jurisdictional levels (n = 12), and at in-country planning workshops (n = 2). Participants included governmental and non-governmental conservation practitioners, members from local communities, different levels of government, as well as different industry sectors. Since it was not possible to conduct interviews with all planners involved across all processes, sampling of interview participants was stratified to ensure that planners operating at different levels (i.e., local, provincial, and national) were represented. Twelve in-depth interviews were deemed sufficient for reaching adequate code and meaning saturation on the basis of: my study purpose, which was to identify broad thematic issues related to factors influencing multiscale planning; a relatively homogeneous population of conservation practitioners; and the level of quality in collected data with the in-depth interviews (see Hennink et al. 2017). These

respondents included planners from local and international environmental NGOs, and national government representatives. Face-to-face interviews were conducted over a two-month period (August – September 2017) and lasted 1-2.5 hours each. I audio-recorded and subsequently transcribed all interviews. Interview questions were semi-structured and focused on individual planning processes, outcomes from these processes, how individual plans related to other plans, and perceptions of planning successes. To avoid biased recollection of responses, results were corroborated with those of other interview respondents and through review of associated documentation on these processes. Discrepancies between respondents were treated as results and evaluated against the specific context of the planner and discussed accordingly in this chapter.

5.4.4 Analysis

Content analysis was conducted on interview transcripts supplemented with document-review and participant-observation data to elicit factors related to the successes and failures of individual planning processes, as well as connections between different planning processes and how these explicitly related to successes or failures across scales and jurisdictional levels. This analysis involved determining common themes and patterns from the collected and collated qualitative data and occurred in two main parts: first, identifying *a priori* themes gleaned from literature on conservation planning across multiple scales and the document review, and then analysing the data for *emergent* themes and patterns. All themes that were repeatedly identified during content analysis across majority of respondents were considered important. Success of individual processes and scalar pathways were evaluated in the context of Bottrill and Pressey's (2012) conservation planning evaluation framework, and the five forms of capital (natural, social, human, financial and institutional).

5.5 Multidirectional scaling: multiscale planning in practice

My analysis of three scalar pathways (two pathways elicited for PNG and one for SI; Figures A4.1-3) derived from 14 conservation plans in total, demonstrated that conservation planning processes inform one another through flows of data, conservation objectives, practitioner experience, institutional support, and funding (Figure 5.3). These resource flows between planning processes undertaken at different levels allowed multiscale planning to occur. Conservation practitioners involved in planning at higher levels often reused large datasets collated or collected for these plans; interviewees regarded these datasets as relatively constant through time. Objectives were frequently associated with these data flows and often originated from international-level commitments. Where flows between multiple processes involved the same personnel at different levels, they contributed flows of planning experience from other

contexts, as well as providing broader perspectives for individual planning processes. These flows of shared experience were frequently considered to result in increased efficiency in planning processes. Institutional support (i.e., policy- or governance-related support from existing formal institutions such as national ministries, or international conventions) was often associated with flows of funding from higher levels to lower levels of planning.

I did not find empirical support for either scaling-down or scaling-up pathways (Figure 5.3; see Figures A4.1-3 for empirical pathways). Instead, I found that scalar pathways demonstrated iterative, bidirectional flows between multiple levels of planning. My finding of multidirectional scalar pathways demonstrates that multiscale planning is occurring in PNG and SI, although this has been opportunistic rather than the result of deliberate high-level coordination over long timeframes.

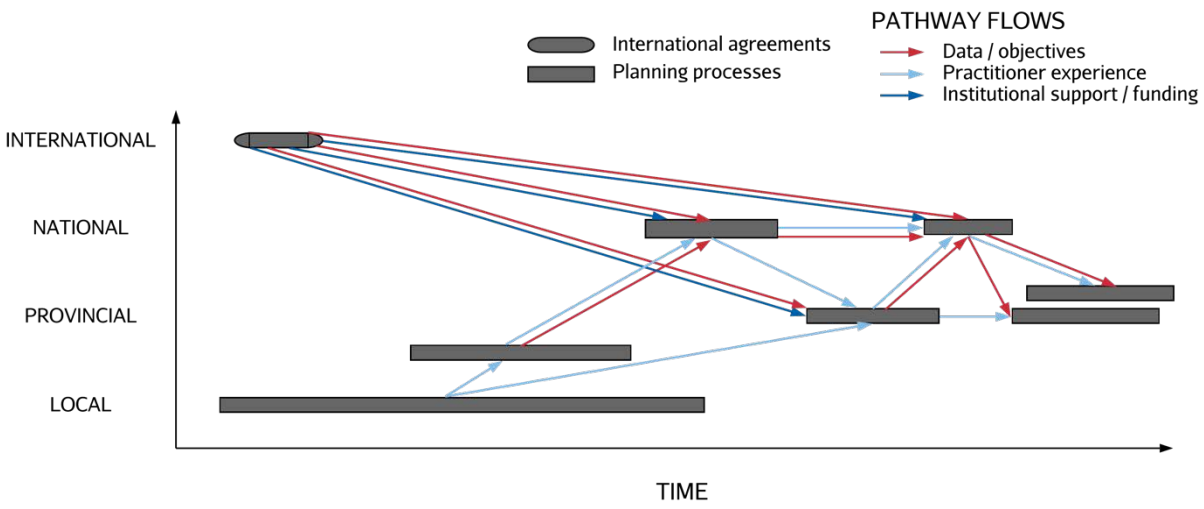


Figure 5.3 Simplified depiction of scalar pathways among published conservation planning processes in Papua New Guinea and Solomon Islands between 1995 and 2017. Pathways consist of different types of flows between levels, which are not unidirectional but vary idiosyncratically over time (see archetype in Figure 5.1c). Planning occurs infrequently at international levels; agreements and conventions between nations are a focal process at this level (e.g., Convention on Biological Diversity targets, the Coral Triangle Initiative). These agreements can provide institutional support and funding for planning at lower levels. Pathways frequently began at the local level, likely a by-product of customary tenure over resources and the strength of local governance in this region, and a lack of institutional capacity at higher levels in the early years of conservation planning. Flows of resources identified between levels of planning were composed of data, conservation objectives, practitioner experience, institutional support, or funding. Practitioner experience was the main resource flow between planning at local and provincial levels, while common datasets and related objectives as well as practitioner experience flowed between planning at higher levels (provincial and national).

Planning processes at all levels contributed some flow of planning resources to other processes, at either the same or different levels (Figure 5.3). Pathways cycled between provincial and national levels in PNG, and mostly within provincial levels in SI, with consistent flows of data or objectives and practitioner experience involved. International-level processes (e.g., Convention on Biological Diversity) supplied flows of planning objectives to conservation practitioners operating at higher levels (i.e., provincial and national), with these objectives informing which datasets to obtain. However, adjustments of objectives from higher levels to consider local preferences occurred as planning progressed to lower levels, a characteristic of scaling down pathways. Similarly, while datasets were often shared between multiple higher-level processes, practitioners updated data to finer resolutions as planning proceeded to local levels. Evidence of this is significant, because practitioners have typically assumed that new and finer-resolution data will either be collected or become available as planning is undertaken at lower levels (Pressey et al. 2013).

Less iterative cycling occurred between the local-provincial and local-national levels than between higher levels (Figure 5.3). Nevertheless, local-level planning played a pivotal role in learning that was subsequently applied by practitioners at higher levels, with flows from local to provincial (and occasionally national) levels consisting primarily of practitioner experience and occasionally data. Interview respondents repeatedly stated that the flows of information and learning between planning processes, particularly from local levels, were highly beneficial; this is supported by the well-established understanding that learning and adaptation are critical in effective conservation planning (Grantham et al. 2010). The reduced iterative cycling observed between lower levels of planning is potentially a result of the available timeline for study (i.e., the timeline of all planning processes evaluated might not be long enough to capture more local-level planning processes, which might occur after the documented provincial- and national-level processes).

Flows of planning resources occurred primarily between processes at adjacent levels (Figure 5.3). The only exception was where particular local-level planning processes achieved a high profile (e.g., in terms of importance or perceived success), leading to recognition at, and interactions with, national or international levels. For example, Kimbe Bay (PNG) emerged from international-level assessments as a regional priority for conservation action, instigating flows of institutional support and funding to local levels (Green et al. 2009). Similarly, the significance and success of the Arnavon Islands community-based conservation area (SI) influenced national-level planning through contribution of data and spatial targets towards national biodiversity commitments (Kool et al. 2010). In turn, these flows of data from the local to national level have

generated further flows of institutional support from the national government down to the local-level Arnavon islands (Foale and Wini 2017).

In all case studies, scalar pathways were initiated at the local level by a range of governmental, NGO, and community stakeholder groups involved in planning. This likely reflects two contextual features: customary governance of resources in Melanesia, and a strategy by The Nature Conservancy of trialling and learning from conservation planning at smaller extents (i.e., local levels) and applying the knowledge gained to subsequent processes at higher levels. However, a unidirectional scaling-up pathway, commonly associated with customary resource ownership and bottom-up conservation planning, was not evident in SI or PNG. Elsewhere, it is possible that scalar pathways would begin at higher jurisdictional levels in contexts with centralised resource governance and stronger institutional capacity (e.g., Yellowstone National Park, USA, in 1872; Oakerson and Parks 2011, Great Barrier Reef Marine Park, Australia, in 1975; Day and Dobbs 2013), but similarly not result in strictly scaling-down pathways.

Other external economic and socio-political conditions also influenced the direction of scalar pathways. Interview respondents indicated that planning moved away from a local-level focus to higher levels of planning following the global financial crisis in 2009. This was a deliberate strategy by planners to implement actions more cost-effectively, since higher-level planning processes involve less on-the-ground engagement, requiring less time and funds to complete than intensive local-level planning exercises. A significant political driver was conservation legislation mandated at the national level, which helped to provide institutional support to planning at lower levels. The creation of such legislation also provided incentives for further planning across all levels and increased time-efficiency in gaining institutional support and endorsement from national governments.

5.6 Scalar capital as input and outcome of multiscale planning success

The success of each planning process could be assessed with the evaluation framework (Bottrill and Pressey 2012) using the five established forms of capital (i.e., natural, human, social, institutional, and financial). For example, a local-level planning process in PNG (Keppel et al. 2012a) resulted in new legislation that enabled the formal recognition and management of conservation areas, an outcome of gaining institutional capital (Bottrill and Pressey 2012). One respondent reported: "First step was to address the issue that [people] did not have consent [over] their land. We engaged an environmental law firm […] to develop a law […] so that local-level governments could directly have a say in how the forest resources were being used". I then associated the successful outcomes of planning processes to factors that facilitated or impeded multiscale planning, through thematic analysis of key-informant interviews. Document and

participant-observation data also were used to confirm and supplement the analysis. I identified a number of themes that repeatedly emerged, all of which were notably scale-explicit. When attempting to evaluate these scale-explicit factors against the evaluation framework, it was apparent that the five forms of capital (Bottrill and Pressey 2012) did not explicitly consider scalar dimensions. Because explicit consideration of scale appears to be fundamental to multiscale planning and scale can in fact be viewed as a resource (Bebbington and Batterbury 2001), I propose a new form of capital – 'scalar capital'.

I define scalar capital as the explicit consideration and application of understanding of the important dimensions of scale, as it pertains to the governance of complex systems. I term these scale-explicit factors that influence multiscale planning, the 'dimensions' of scalar capital. In line with the framework to evaluate conservation planning outcomes (Bottrill and Pressey 2012), I propose scalar capital as an input for and product of investment, ultimately accruing flows of planning resources and benefits over time (Table 5.1). If conservation problems are inherently multiscale, then solutions must also be, making scalar capital essential to evaluations of conservation planning. I identified six principal dimensions of scalar capital: (1) multiscale understanding, (2) scale jumping, (3) leadership characteristics, (4) stakeholder engagement, (5) policy frameworks, and (6) institutional settings. The first two dimensions are concepts unfamiliar to the conservation planning literature; I describe these in detail in the following sections and discuss potential implications for future multiscale conservation planning. The literature has long recognised the remaining four dimensions, which I corroborated with the findings across my case studies. Critically, while these dimensions are recognised, I emphasise the need to ensure that they are multiscale (i.e., present across all levels of planning) to contribute to scalar capital.

Regarding the more familiar dimensions, much evidence supports the vital importance of leadership characteristics in successful conservation planning, with many arguing that a leadership approach can be intentionally managed to maximise impact (e.g., Black et al. 2011, Bruyere 2015). Similarly, there is now a very clear understanding of the necessity of genuine stakeholder engagement in any conservation planning process (see Pomeroy and Douvere 2008 for examinations of various engagement approaches, and Reed 2008 for a comprehensive review). While there are no silver-bullet policy frameworks and institutional settings that can be applied to ensure success in any given planning context, the importance of frameworks operating effectively in the context of natural resource management is obvious (Kingsford et al. 2009, Ferse et al. 2010). In PNG and SI, I found that, where certain policies and institutions existed at the relevant levels, practitioners were able to expedite planning processes and achieve greater implementation success compared to other instances where these policies or institutions were

absent. This concept is referred to in the literature as vertical policy integration (Roux et al. 2008, Adger and Jordan 2009) or cross-scale linkages (Wyborn 2014), and while typically discussed with reference to policies, applies equally to institutions (Schout and Jordan 2008). Though there is no established solution to achieving vertical integration in an uncoordinated or fragmented system (Lane and Robinson 2009), it is clear that successful integration requires policies and institutions that are consistent, coherent, and mutually supportive across jurisdictional levels, and mechanisms in place that facilitate regular exchange of information, consultation, and arbitration between all levels (Jordan and Lenschow 2008).

Table 5.1 Summary of definitions, example outcomes, and indicators for the five forms of capital relevant to conservation planning processes (natural, financial, human, social, and institutional) from Bottrill & Pressey (2012), with the addition of scalar capital as a proposed new form.

Capital	Definition	Example outcome	Indicator with example	Reference
Natural	Stock and flow of goods and services provided by ecosystems, including the diversity of species, regulating processes, and supporting services	Reduction in loss or degradation of natural values	Extent and intensity of threatening processes (e.g., deforestation; exploitation)	(Costanza and Daly 1992)
Financial	Gains or savings of cash, property or goods that represent the wealth or economic value of an individual or organisation	Leverage of additional funds or in-kind support	Proportion of additional funds received (e.g., % change in annual budget of implementing agency attributable to new donors)	(Bottrill and Pressey 2012)
Human	Knowledge or skills that enable people to develop strategies to achieve their objectives, which provide the foundation for the other four types of capital	Learning applied in future plans	Use of new knowledge or skills applied in subsequent plans (e.g., application of new decision tool by members of planning team)	(Scoones 1998)
Social	The relationships and interactions between individuals and groups with productive benefits	Trust in planning processes	Perceptions of planning process and outputs by stakeholders (e.g., % of stakeholders with positive view of plan)	(Pretty and Ward 2001)
Institutional	The capacity, structure, or functioning of institutions through formal means (e.g., laws and regulations) or informal arrangements (e.g., cultural norms applied in governing natural resource uses)	Influence on resource-use planning	Avoidance by developers of priority conservation areas (e.g., occurrence of development applications in priority areas)	(Ostrom 1990)
Scalar	**The explicit consideration and application of understanding of the important dimensions of scale, as they pertain to the governance of complex systems**	**Planning processes informing other processes at different scales**	**Scale-constrained actors gaining access to resources from levels of planning otherwise inaccessible (e.g., individuals from planning processes put into contact with processes at other levels to share knowledge)**	

5.7 Novel dimensions of scalar capital

5.7.1 Multiscale understanding

To successfully plan across multiple scales, practitioners first need to understand the purpose, strengths, and weaknesses of planning at different jurisdictional levels. I looked at realised (cf., anticipated) outcomes from individual planning processes in PNG and SI to identify the purpose of planning at different levels. Key informants linked the absence of explicit understanding of the functions and limitations of what can be realistically achieved by planning at each level, to less effective planning outcomes at these levels and, therefore, across scales. For example, where planning teams expected that protected area implementation would be a direct outcome from national planning exercises and this did not eventuate, it led to disappointment, decreased morale, and perceptions of wasted effort. At local levels, I found socially motivated objectives gained the most importance compared to other levels. Planning processes that involved expectations and objectives related to social factors were more successful in implementing actions than those that did not. It is important to consider here that outcomes observed at higher levels may be idiosyncratic due to a lower number of units than at lower levels; however, such outcomes have been demonstrated to exert significant forces in influencing community assembly in ecosystems (Terborgh et al. 2001).

A factor that inhibited understanding the purpose of planning at different scales was the widespread misconception that costs incurred to implement plans at the local level can be decreased by leveraging planning into broader scales (through 'economies of scale'). This was inhibiting because conservation planning was encouraged to move away from local levels to maximise cost effectiveness. While economies of scale are applicable in certain planning contexts (see Armsworth et al. 2011), my case studies demonstrated that actions are implemented at the local level and therefore associated costs with doing so cannot be leveraged up.

Related to multiscale understanding is the explicit consideration of how the intrinsic geography of a location relates to the different jurisdictional levels of planning, and how this can influence planning success. Aspects of geographical context that I found influential were: the potential physical restriction of actors across landscapes and consequent social networks that arise from levels of connectivity; access to available resources; and the spatial fit between jurisdictional, geographical, and pragmatic concerns. The importance of fit between governing institutions and ecological processes or problems is well established in the social-ecological systems literature ('ecological fit'; Bodin et al. 2014, Epstein et al. 2015). However, the connection between institutions and the intrinsic geography in the ecological-fit literature is often implicit; here I refer to the fit between these explicitly. For example, planning processes were perceived as less

complex and more successful in SI than PNG due to geographical differences: SI comprises small, discrete island units separated by tracts of ocean, compared to the larger contiguous landmasses of PNG. The physical separation and comparatively smaller extents of provincial jurisdictions, and consequent greater fit between provinces and spatial extents to be planned for and managed by communities in SI (Figure 5.2), contributed to more effective planning outcomes. The ability of geographical context to influence successful planning highlights the importance of understanding this dimension of scalar capital.

Explicit consideration of temporal scales is another essential component to multiscale understanding. Appropriate temporal scales appear to reflect the varying purposes of design and implementation that are relevant across different spatial extents. If we consider that spatial extent motivates the way conservation practitioners think about different objectives, and planning at different levels serves different purposes, then the expected temporal scales upon which different planning processes operate should also differ between levels and purposes. Consistent across my case studies (Figures A4.1-3), all gazetted local protected areas have required extensive periods from conception to implementation (15-25 years; also see Morrison 2009). Conversely, at higher levels (e.g., national and provincial) where planning more appropriately revolves around the design stages of conservation planning but lacks implementation, processes occurred over much shorter timeframes (2-3 years).

5.7.2 Scale jumping: types and enabling mechanisms

Placed within the context of conservation planning, scale jumping (Smith 1993) refers to the ability of actors or organisations to interact and operate vertically across multiple scales (here, across multiple jurisdictional levels), thereby enhancing the capacity of one scale from another (Morrison 2007). This is integral to multiscale planning because it creates social or institutional links between the different levels of planning, facilitating planning processes at particular levels to inform those at other levels. For example, an individual capable of scale jumping is involved in processes at multiple levels (e.g., local and national) and, as a result, is capable of connecting otherwise constrained individuals or organisations at either of these levels. Significantly, my case studies demonstrated that successful occurrences of scale jumping mechanisms and outcomes had the potential to produce positive feedback flows, through the generation of resource flows to other levels, or through the positive attitudes accumulated through such activities.

I identified five types of scale jumping pertinent to multiscale planning: (1) integrating lessons from other scales, (2) contextualising, (3) grounding, (4) forecasting, and (5) accessing exogenous and cross-level resources (Figures 5.4 and 5.5). These five types reflect different outcomes

produced through distinct enabling mechanisms (Figure 5.5). Below, I describe each of these types, the mechanisms that enabled them, and their implications for multiscale planning.

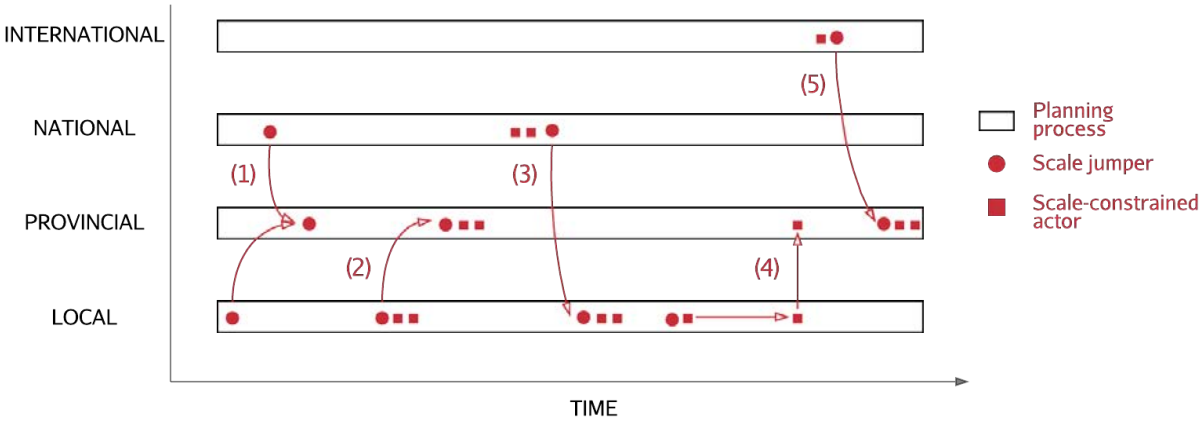


Figure 5.4 Schematic depiction of the five types of scale jumping identified, across various jurisdictional levels: (1) integrating lessons from other scales, (2) contextualising, (3) grounding, (4) forecasting, and (5) accessing exogenous and cross-level resources. Jurisdictional levels chosen to demonstrate each type are notional; all types of scale jumping may occur from and to any level.

Integrating lessons learned from conservation planning at other scales is a critical form of scale jumping (type 1, Figure 5.4). Interview respondents reported that conservation practitioners who were involved in planning processes across multiple scales were able to draw from a richer knowledge base, informed by social, human, and institutional conditions experienced in varying contexts at each level of planning. Such multilevel learning also allowed planning to be trialled at smaller, more manageable extents and learning from successful outcomes subsequently applied to higher levels and larger extents. This form of scale jumping is an important contributor to successful adaptive planning, which necessarily requires ongoing and explicit learning, as well as engagement with organisations and stakeholders at multiple levels (Mills et al. 2015). Moreover, the retention of individuals across different processes that this type of scale jumping entails has been shown to promote retention of institutional knowledge (Fox et al. 2013).

Contextualising and grounding (types 2 and 3, Figure 5.4) involve scale-constrained actors or organisations making decisions in the contexts of higher or lower levels, respectively. For contextualising (type 2), wherein decisions at lower levels are placed into the context of higher levels, scale jumpers motivate decisions by local stakeholders with a broader context (e.g., understanding the full extent of the degradation of timber resources that has occurred across the whole province). Grounding (type 3) places decisions at higher levels into a lower-level context, whereby scale jumpers mediate an international organisation to broaden their decision-making

context to consider local circumstances, around which the international organisation has stakes (e.g., identification of conflict between an allocated mining tenement site with the highest conservation priorities established by local communities). Forecasting (type 4, Figure 5.4) similarly involves a broadening of scalar contexts, but along temporal scales. In this type of scale jumping, actors or organisations constrained to thinking within short timeframes broaden these to consider processes over a wider range of temporal scales, relevant at higher social or ecological scales (e.g., Cumming et al. 2015). In this way, the jumping of spatial scales that occurs in forecasting is coincidental, arising from the expansion of temporal perspectives.

Interviewees repeatedly highlighted accessing exogenous and cross-level resources (type 5, Figure 5.4) as a factor contributing to the success of conservation planning across levels, emerging in my analysis as a significant type of scale jumping. My reference to resources here is not exclusively monetary but includes all the socially based forms of capital (social, human, and institutional; Table 5.1). I found that, across all levels of planning, required resources consistently existed at other levels, both above and below, which were not accessible without scale jumpers liaising between levels.

I discerned five enabling mechanisms (Figure 5.5) through which different types of scale jumping could occur: maintaining continuity of individuals, co-locating actors, expanding perceptions, reducing social distance, and building capacity (see Appendix 4 Table A4.2 for details of observed examples). I found that a single type of scale jumping was achievable through different mechanisms. Thus, I used a matrix (Figure 5.5) of enabling mechanisms and types of scale jumping to understand observed (by key informants) and potential combinations. Of interest are the potential combinations: these may prove useful to investigate in future multiscale planning processes to maximise this dimension of scalar capital.

I found that maintaining the continuity of individuals (individuals' involvement in planning processes across multiple levels and geographies; Figure 5.5) was fundamental to achieving many types of scale jumping. This concept is alluded to in descriptions and examples of human and institutional capital (Bottrill and Pressey 2012); however, neither the role of this concept nor its pertinence to scale has been made explicit. While I refer to individuals remaining constant between processes, individuals can be substituted with organisational memory (Walsh and Ungson 1991) but with the strong caveat of needing accurate, comprehensive, and timely recording systems within the organisation. Continuity of individuals across different levels contributes to increasing the wealth of planning experiences and knowledge gained from different institutional and geographical contexts by conservation practitioners (Fox et al. 2013), while also strengthening the capacity of these individuals to jump between different levels of planning. The significance of this role has been outlined previously with respect to successful

adaptive conservation planning (Mills et al. 2015). The connections between planning processes created through involving the same key individuals ensures that lessons learned in one process can be easily applied in others while continual learning occurs, improving efficiencies.

MECHANISMS / TYPES	**Maintaining continuity of individuals** (individuals' involvement in planning processes across multiple levels and geographies)	**Co-locating actors** (physical placement of actors from processes at different levels or geographies)	**Expanding perceptions** (provision of information relevant to levels beyond normative understanding)	**Reducing social distance** (connection of actors or organisations with greater resources operating at different levels)	**Building capacity** (spontaneous application of gained knowledge at different levels)
Integrating lessons from other scales (knowledge pertaining to increasing gains in capital)	■				
Contextualising (decisions at lower levels placed into context of higher levels)	■	■	■		■
Grounding (decisions at higher levels account for constraints and opportunities at lower levels)	■	■	■	■	
Forecasting (extension of temporal perspectives)		■	■		
Accessing exogenous & cross-level resources (external social, human, institutional, or financial resources from other levels)	■	■	■	■	■

Figure 5.5 Combinations of types of scale jumping and enabling mechanisms identified from interviews with key informants involved in conservation planning in PNG and SI. Filled cells represent observed combinations; blank cells represent potential combinations for future exploration.

Co-locating actors (physical placement of actors from processes at different levels or geographies; Figure 5.5) often had a powerful impact by allowing actors to physically step outside of constraints imposed by their original scales of operation. This enabling mechanism relates to the concept of social learning, which is understood as a central tenet in environmental management fields (Berkes 2009). Specifically, co-locating actors in scale jumping speaks to group-centred and multilevel social learning, where new knowledge is created through the transformation of experience, allowing iterative reflection that can occur when ideas and experiences are shared with others (Berkes 2009, Pahl-Wostl 2009). Furthermore, such social learning has demonstrated successful sharing of environmental memory between diverse regions, in terms of management responses to change (Matous and Todo 2018). A related enabling mechanism I identified was expanding perceptions (provision of information that represented scales beyond normative levels of understanding by scale-constrained actors or organisations; Figure 5.5). Expanding perceptions also revolves around learning, where scale-constrained actors understand perspectives and gain knowledge generated at other levels, which has been signified as important for vertical coordination (Pahl-Wostl 2009).

Crucially, I found that consequences of scale jumping can effect a positive-feedback loop whereby enabling mechanisms are facilitated, thus allowing more scale-jumping outcomes to be produced. For example, a pivotal consequence of scale jumping is the development of broad social networks, which in turn contribute to the abilities of a scale jumper in reducing social distance between unconnected actors or organisations with greater resources and others with fewer resources and at different levels (Figure 5.5). This notion of bridging ties (within and across scales) is understood in social network theory as an influential factor behind successful management (Schneider et al. 2003), due to the greater diversity of experiences and knowledge systems mobilised and fostering of trust between diverse groups that spur collective actions (Bodin and Crona 2009, Ernstson et al. 2010, Cohen et al. 2012, Guerrero et al. 2015b). Scale jumpers also trained and built the capacity (Figure 5.5) of scale-constrained actors and organisations so that they could spontaneously and autonomously apply newly acquired knowledge in such a way as to jump scales themselves. Together with reducing social distance, these were the least common enabling mechanisms I identified (Figure 5.5). These two less common mechanisms may warrant further exploration in future multiscale planning processes. Future research should also investigate causal and interactional relationships between scale jumping enabling mechanisms that we observed often occurred in tandem.

I observed one instance where scale jumping hindered the progression of a planning process (and thus potentially multiscale planning). This occurred where a provincial planning process that experienced many of the beneficial outcomes of scale jumping also experienced an overload of attention as a result. Too many individuals and organisations sought to be involved, inadvertently overwhelming the province and rendering planning processes less effective and efficient (i.e., coordination between projects became challenging and local practitioners were overworked). This consequence may be attributable to the fact that scale jumpers and their activities were unequally distributed across the region. Were scale jumping enacted more uniformly across the region and beyond the one province, outcomes may have remained productive. For example, if grounding (type 3, Figure 5.5) were occurring in local-level sites across multiple provinces, attention from higher levels would be more evenly distributed. A similar conclusion has been drawn in the context of multilevel governance for large marine commons, where less distributed decision-making in a nested system can constrain innovation and diversity (Gruby and Basurto 2014) . This has potentially important implications for 'hotspots' approaches to conservation (Myers et al. 2000) because these encourage concentrations of funds and actors into a few regions (e.g., Allen 2008).

5.8 Scope to foster dimensions of scalar capital

Given that multiscale governance is critical to maximising conservation outcomes (Lemos and Agrawal 2006, Cumming et al. 2015) and that scalar capital enables multiscale conservation planning, fostering scalar capital should be a prime consideration of conservation scientists, policymakers, and practitioners. Importantly, there is empirical evidence from my case studies that the different dimensions of scalar capital can be fostered in planning processes (see Appendix 4 Table A4.3). Based on my findings, I make other recommendations on ways scalar capital may be fostered.

In the dimension of multiscale understanding, I highlight the significance of understanding the purpose(s) of planning at different scales. Despite social objectives gaining the most importance at lower levels and the difficulties in operationalising social objectives at high levels (as demonstrated in Chapter 4), I propose that social objectives be considered by conservation practitioners at all levels. What requires change is the way that these objectives are conceptualised at different levels, being formulated to consider actors and features of the governance system that are relevant to each level. Local-level planning processes are likely essential in any multiscale planning context and I contend that limitations associated with high costs should be explicitly acknowledged to avoid failed expectations, particularly in discourses with funding donors who favour projects that appear more cost-effective (AbouAssi 2013). To overcome the inherent variation in temporal scales relevant at different levels of planning, I suggest that conservation practitioners build a 'planning system identity' (as complex systems; Cumming and Collier 2005, Folke 2006), with iterative flows and feedbacks that need to occur between each of the levels over time. I argue that conceiving multiscale conservation planning as a complex system will facilitate more effective outcomes across scales, especially through promoting multiple-loop learning between different levels or ecological scales to inform decision making (e.g., Argyris 1976, Pahl-Wostl 2009). For planning at particular jurisdictional levels to inform and align with planning at other levels, practitioners and, importantly, funding institutions, must expand the temporal scales considered to include as much of the planning system as possible (e.g., long-term institutional commitment, recording and revision of organisational memory, and continuity of personnel; Pressey and Bottrill 2009, Pressey et al. 2013). Otherwise, these organisations will waste brief but valuable efforts, when institutional responses are not synchronised with ecological or social processes (Levin et al. 2013, Epstein et al. 2015).

Across a number of research areas, significant overlap is evident in the concepts I have identified and discussed in this chapter, related to governance, geography, and interactions within and

between actors and organisations at different levels. This suggests that the ability to foster scalar capital is of broader relevance and interest across the fields of collaborative governance, policy and social network theory, political geography, social-ecological systems theory, and institutional fit. In understanding policy and social networks in environmental governance, certain networks have been identified as less effective for multi-actor collaboration (e.g., Mills et al. 2014, Sayles and Baggio 2017). The specific types and enabling mechanisms of scale jumping I have identified may be utilised in such cases, to influence the social or policy networks and improve the capacity for collaborative governance. Moreover, empirical studies of social networks that analyse explicit cross-scale relations have identified only the relative positions of scale-crossing brokers within the network that provide integral links between levels, or areas within the network where cross-scale links are lacking (Cohen et al. 2012, Guerrero et al. 2015b). Here, I move towards a deeper understanding of the distinct mechanisms that enable different outcomes of scale jumping, as a basis for fostering these cross-scale links in environmental governance. Though I identified the continuity of individuals across processes at different levels as a fundamental enabling mechanism to scale jumping, this does not mean that planning teams should remain constant. New individuals bring novel perspectives, while those entrenched within a particular aspect of the planning process may not be able to observe any flaws or inefficiencies. This is supported by collaborative governance and social network theory, which emphasizes the importance of bridging ties to other subgroups in enhancing productivity and innovation (Bodin and Crona 2009).

Building capacity (Figure 5.5) for implementation and management is particularly relevant at local levels, where jurisdictional distance from national governments is greatest, and institutional capacity is often weakest, particularly in developing nations (Cuthill and Fien 2005). Thus, for long-term success in multiscale conservation planning, capacity building should be an integral component of planning processes at levels where institutional capacity is low. Moreover, conservation planning is globally under-resourced (Halpern et al. 2006), which makes accessing exogenous and cross-level resources (type 5, Figure 5.4) essential to the success of multiscale planning processes. In the related fields of social-ecological systems theory and the prevalent problems of fit (Epstein et al. 2015), fostering the dimensions of scalar capital may contribute by aligning the planning and institutional systems, and temporal scales considered, closer to ecological systems.

5.9 Conclusion

The intrinsic role of scale in any social-ecological system means that scientists, policymakers and planners must explicitly consider multiple scales in the successful understanding or management

of these systems. I am the first to demonstrate empirically how multiscale conservation planning occurs in practice and that the perceived dichotomy of scaling down and scaling up in conservation planning may not in fact be representative of real-world multiscale governance. Despite highly decentralised governance systems in Melanesia, evidence of multiscale planning was not strictly unidirectional (i.e., scaling up) and involved multidirectional flows of planning resources between different levels.

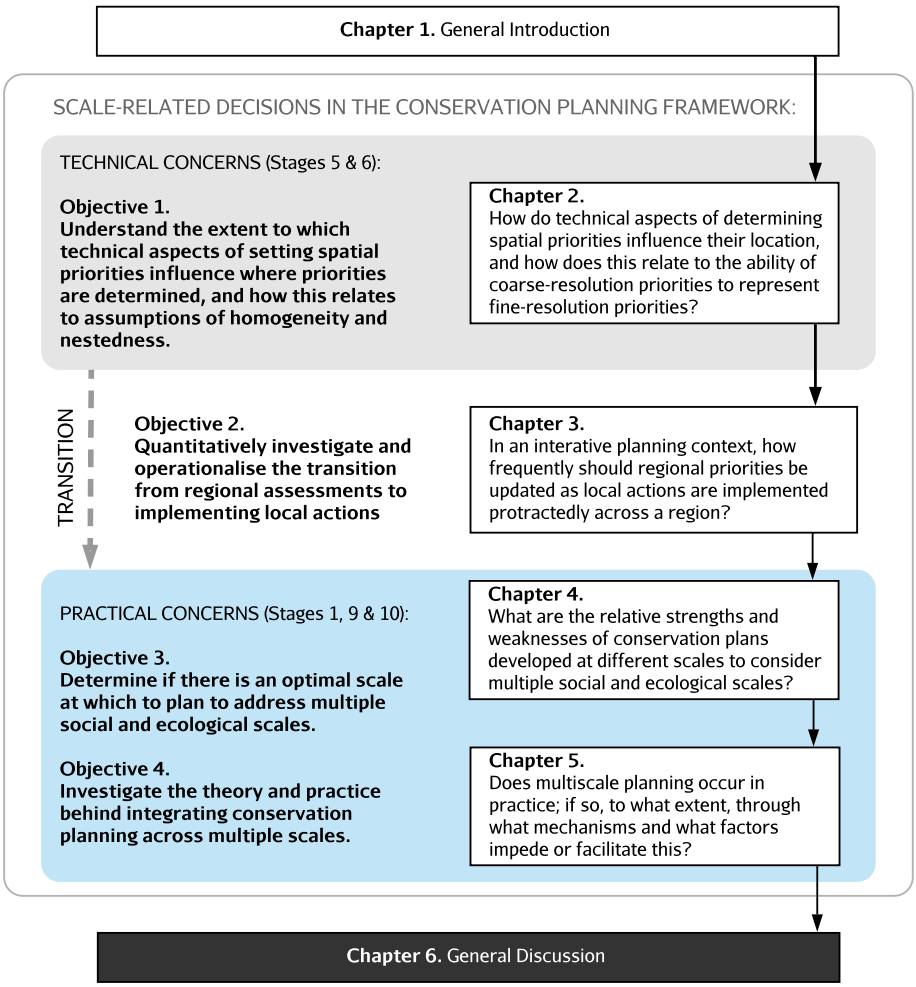
I define the concept of scalar capital and highlight its necessity for effective multiscale governance. Two novel dimensions of scalar capital, multiscale understanding and scale jumping, appear equally critical in successfully integrating and coordinating conservation governance across multiple scales. I propose that scientists, policymakers, and planners integrate scalar capital into existing evaluation frameworks for conservation planning and governance to improve explicit considerations of scale in these processes, and ultimately, multiscale outcomes. Critically, I also present empirical evidence illustrating ways to foster scalar capital. Scope remains to explore these concepts further and understand the extent of their applications in more detail. While inputs of scalar capital into multiscale conservation planning have thus far been inadvertent, I suggest that conservation scientists, policymakers, and planners should invest in generating scalar capital within and across processes, and intentionally design multiscale planning as a way to improve conservation outcomes across multiple scales.

Chapter 6

General Discussion

6 General Discussion

In Chapter 6, I discuss how Chapters 2-5 address the overall goal and four research objectives of my thesis outlined in Chapter 1. I highlight how this thesis contributes new knowledge towards our current understanding in the theory and practice of conservation planning, as well as consider the main shortcomings of my research, remaining knowledge gaps, and scope for future research.



6 General Discussion

6.1 Thesis summary

Despite continually increasing commitments to environmental protection around the globe (Watson et al. 2016), there is significant evidence that many protected areas, particularly marine protected areas, fail to deliver beneficial ecological and social outcomes (Gill et al. 2017). Exacerbating this problem is the influence of scale in understanding and managing natural systems (Holling 2001, Cash et al. 2006), which has long hindered the ability of conservation practitioners and researchers to successfully undertake conservation planning and governance (Berkes 2002, 2006, Folke et al. 2007, Hill and Engle 2013, Beever et al. 2014). This, coupled with the overwhelming complexity of ecosystem management today (DeFries and Nagendra 2017) and ubiquitously constrained resources for conservation (Halpern et al. 2006) highlights the serious urgency of conducting effective conservation planning across multiple scales.

As described in Chapter 1, considerations of scale throughout the 11-stage conservation planning framework (Pressey and Bottrill 2009) have been limited and largely implicit (Mills et al. 2010), and can be broadly categorised into technical (e.g., spatial prioritisation assessments; stages 5 & 6) and practical (e.g., implementing conservation plans; stages 1, 9 & 10) considerations. The overall goal of this thesis was to understand the different influences of scale on outcomes of the conservation planning framework, to ultimately make specific recommendations that enable conservation practitioners to account for scale more explicitly throughout planning processes. To accomplish this goal, I formulated four broad research objectives, which were designed to address the key knowledge gaps that pertain to understanding scalar influences throughout the conservation planning framework. I first examined the influence of spatial resolution and heterogeneity of planning-unit size and socioeconomic cost data, respectively, and the thematic resolution of reef classes, on spatial prioritisation assessments for conservation (Chapter 2). I then investigated the implications of operationalising iterative planning processes (Chapter 3), which are often proposed or assumed to overcome the limitations associated with extensive, coarse-resolution assessments but have yet to be considered quantitatively or explicitly. Taking as a case study conservation plans from the Coral Triangle, I identified the strengths and weaknesses of plans developed at different scales to consider multiple social and ecological scales (Chapter 4). Finally, I evaluated the theory and practice of multiscale conservation planning (Chapter 5).

Despite the rapidly developing recognition of the significance of scale in conservation planning and governance outcomes (Wyborn and Bixler 2013, Lengyel et al. 2014, Guerrero et al. 2015b, Maciejewski and Cumming 2016, Cumming et al. 2017, Sayles and Baggio 2017), many aspects of scalar influences in complex systems are still poorly understood due to the intractable nature of the concept. In the following sections, I describe how each of my data-based chapters addresses the research objectives, highlighting how they contribute new knowledge towards our current understanding in the theory and practice of conservation planning, and outline specific contributions and recommendations pertaining to the knowledge gaps identified in Chapter 1. Further, I discuss the main shortcomings of my research, remaining knowledge gaps, and the scope for future research that can expand on this work.

6.2 Achievement of objectives and thesis contributions

6.2.1 Technical concerns of scale in conservation planning (stages 5 & 6)

Objective 1. Understand the extent to which technical aspects of setting spatial priorities for marine conservation influence where priorities are determined, and how this relates to assumptions of homogeneity and nestedness.

To address objective 1, I quantified the individual and interacting effects of three spatial prioritisation factors in Chapter 1: (1) planning-unit size, (2) thematic resolution of reef-class maps, and (3) spatial variability of socioeconomic cost data. To quantify the effects of these factors in planning extents large enough to determine scale effects, I conducted these evaluations using two case study regions of large extents, Fiji and Micronesia (the Mariana Islands, Marshall Islands, Palau, Guam and the Federated States of Micronesia). I evaluated the influence of resolution on spatial priorities through quantitative comparison of different prioritisation aspects: (1) total extent and cost of prioritised areas, (2) spatial configuration of priority areas, (3) spatial nestedness of priorities determined with coarse- and fine-resolution planning units, and (4) extent of incidental representation of fine-resolution priorities by coarse-resolution priorities.

Two important knowledge gaps I identified in Chapter 1 that pertain to technical concerns in conservation planning are: (1) understanding the individual and interacting effects of different levels of resolution of prioritisation factors on marine spatial priorities, and (2) examining the ability of coarse prioritisations to guide finer-resolution assessments. My research in Chapter 2 contributes to filling these gaps, first, through demonstrating the significant influence of the socioeconomic cost data layer on how priority areas are determined, and as a result, its ability to interact with other prioritisation data layers (e.g., biodiversity surrogates, such as reef-class maps) to alter reserve configurations; and second, by identifying how coarse prioritisations can be

conducted to maximise the likelihood of representing finer-resolution priorities in marine environments. I found that all three prioritisation factors influenced where spatial priorities were determined to some extent, with the spatial variability of socioeconomic cost data having the greatest influence. Importantly, this research also found an interaction effect between two prioritisation factors: the thematic resolution of reef classes and the spatial variability of cost data used. I confirmed my prediction that the assumptions of homogeneity and nestedness are largely invalid, with poor spatial nestedness of conservation priorities achieved unless the same socioeconomic data were used in both coarse- and fine-resolution assessments. Surprising, however, was my finding that considerably greater extents of incidental representation of fine-resolution objectives were achieved by coarse prioritisations run with spatially uniform socioeconomic costs, as compared with coarse prioritisations run with spatially variable costs.

Novel contributions from this research were identifying an interaction effect between apparently unrelated prioritisation factors, and being the first study, to my knowledge, to test both assumptions of homogeneity and nestedness in using coarse prioritisations to guide subsequent finer-resolution assessments that incorporate different socioeconomic cost layers. Importantly, this research invalidates the assumptions implicit in the strategy commonly presupposed to overcome the shortfalls of conducting coarse prioritisations to identify priorities (such as relying on these coarse priorities to direct more detailed analyses across the Coral Triangle region; Beger et al. 2013). While the ability of cost data to influence spatial conservation priorities is well known (Richardson et al. 2006, Carwardine et al. 2008, Adams et al. 2010, Weeks et al. 2010c), this research has extended understanding on the degree and scope of this effect. In particular, I identified the potentially influential role of region-specific contexts to interact or compound the influence of cost on spatial priorities (i.e., the spatial relationship between the distribution of reef classes and the variability in socioeconomic costs across space).

Crucially, Chapter 2 led to three practical recommendations to consider resolution more explicitly in the spatial prioritisation process. While the first two were intuitive (i.e., use the smallest practical planning-unit size and highest-resolution biodiversity or socioeconomic data where possible), the last recommendation (i.e., when conducting conservation assessments across regional extents to guide subsequent finer-resolution prioritisations, do not bias priorities with inaccurate, coarse-resolution socioeconomic cost data), has significant implications. The importance of including socioeconomic cost data in conservation prioritisations is now widely recognised (Ban and Klein 2009, Ban et al. 2013, Gurney et al. 2015); this, coupled with the common approach of using coarse prioritisations to guide finer-resolution ones (Larsen and Rahbek 2003, Fjeldså 2007), or assuming that planning will progress from coarse to fine scales (Klein et al. 2010, Beger et al. 2013) may in fact result in perverse outcomes for conservation. I

demonstrate that incorporating spatially variable socioeconomic cost data in regional assessments will strongly bias priority areas to large 'low-cost' planning units, which are most likely based on inaccurate data, since it is at local levels that socioeconomic costs to resource users are most accurately measured (particularly in regions of poor data availability; Ban et al. 2009). A similar finding of this negative relationship between planning-unit cost and incidental representation of environmental classes has been demonstrated in the Great Barrier Reef marine reserve network (Bridge et al. 2016). I add to this evidence base by showing similar outcomes in geomorphologically and socioeconomically distinct regions (i.e., Fiji and Micronesia). Finally, this result from Chapter 2 has critical implications for iterative planning processes, which were explored in Chapter 3, since such processes rely on the ability of coarse regional prioritisations to reasonably represent finer-resolution priorities.

6.2.2 Practical concerns of scale in conservation planning (stages 1, 9 & 10)

Objective 2. Quantitatively investigate and operationalise the transition from regional conservation assessments to implementing local actions.

To address objective 2, I simulated the process of iteratively transitioning between regional conservation assessments (large, arbitrary planning units) and implementing local conservation actions (small, irregularly-shaped management units) in Chapter 3. This is the first study to operationalise and quantitatively investigate iterative planning processes through simulations. Thus, the approach I used was novel and the first of its kind; the simulations were coded with the programming language *R* and were designed to execute spatial prioritisation assessments (with Marxan) on its own accord at appropriate stages within each run of the simulation. With the plethora of factors causing necessary changes as regional plans are implemented (Pressey et al. 2013), I chose to focus this research specifically on the question of how frequently regionally assessed priorities should be updated to account for new information that emerges, as local actions are implemented protractedly across the region. I used Fiji as a study region and examined the influence of four different rates of update frequency to regional priorities.

Two key knowledge gaps I identified in Chapter 1 that pertain to practical concerns associated with this research objective are: (1) investigating the frequency with which regional assessments should be updated in the transition to local actions, and (2) identifying the extent to which plans need to change when transitioning from regional assessments to local actions. I found that, while update frequency did not appear to influence the total time taken to achieve all objectives, or the total extent of the final implemented reserve system, updating regional priorities more frequently did influence how quickly high-priority objectives were achieved, and the potential for revisiting priority areas where some conservation actions have already been applied. My research in

Chapter 3 contributes to these gaps by providing nuanced insights into the trade-offs to consider with respect to the frequency of updating regional priorities and the specific planning context. Importantly, I quantified two valuable benefits of updating regional priorities more frequently: (1) faster achievement of conservation objectives for high-priority features, and (2) greater potential to capitalise on areas previously investigated. With increasing threats to and losses of biodiversity and beneficial ecosystem functioning (Worm et al. 2006, Mora and Sale 2011, Hooper et al. 2012, Mascia et al. 2017) and a global lack of evidence for effective conservation actions (Ferraro and Pressey 2015, Gill et al. 2017), my finding that more frequent updates of regional priorities can result in more efficient achievement of high-priority features is a valuable consideration in planning regions with imminently threatened features (e.g., Bird's Head Seascape in Indonesia; Grantham et al. 2013).

Additionally, in many regions where economies are developing and socio-political and cultural complexities are diverse (e.g., the Coral Triangle; Fidelman et al. 2012), implementing conservation actions often requires extensive and costly on-ground investigations and negotiations (Pressey et al. 2013). My results demonstrate that, in these contexts, updating priorities more frequently may provide an important cost-efficient benefit by increasing the potential to take advantage of previous efforts in implementing local actions (provided process fatigue is not incurred; Weeks and Jupiter 2013). For example, engagement with local communities is often necessary to inform them of relevant conservation practices and gain their interest in implementing management actions (Kereseka 2014). Such engagement is a costly multiphase exercise that will likely require numerous outreach programs and participatory consultations (Andrade and Rhodes 2012). In situations where initial engagements do not garner concerted interest, or resulting enthusiasm spreads to neighbouring communities (e.g., Govan et al. 2009), capitalising on previous engagement efforts can yield cost efficiencies.

This research has advanced our understanding of the implications of operationalising iterative conservation planning processes. Using my novel simulation framework, I have been able to operationalise the concept of iterative planning and, critically, created a coding framework that can be utilised and adapted by others interested in investigating other parameters or factors related to iterative planning processes. The planning context where conservation actions are implemented protractedly across a region is globally widespread (Pressey et al. 2013), underlining the relevance of my results for planning in practice. In Pacific Island nations where many global hotspots of biodiversity occur (Kingsford et al. 2009), those with developing economies possess natural resources that are often finely subdivided in terms of ownership (e.g., by local communities). This results in conservation practice implemented at local levels and with insufficient resources or management (and thus incrementally; Keppel et al. 2012b). Further, the

implementation crisis stemming from the lack of translation of conservation designs (i.e., plans) to implemented on-ground actions in conservation planning (Biggs et al. 2011) is particularly severe in countries with developing economies (Mills et al. 2010). My findings make a practical contribution to outlining specific strategies related to iterative planning processes that can help mitigate the implementation gap in these countries. Furthermore, there has been discord in the conservation literature as to whether adequately representing all known features (Margules et al. 2002) or focusing efforts on threatened features (Brooks et al. 2006) is a more effective strategy of conserving biodiversity, or a negotiated balance between the two (Pressey and Bottrill 2008). My research offers an approach that operationally merges these perspectives in the implementation of conservation actions, and enables practitioners to quantitatively investigate scenarios that can explore the balance between these perspectives.

The use of my novel simulation framework in Chapter 3 also enabled me to quantify the extent to which regional designs change, depending on differences in update frequency. Expectedly, I found that the extent of change between regional designs and local actions increases when regional priorities are updated more frequently. I quantitatively demonstrated that neither the total time taken to achieve objectives, nor the total extent of final reserve systems were influenced by changes in update frequency, suggesting that regular changes to regional designs do not necessarily impact achievement of conservation objectives. This has implications for conservation planning in practice, which appears to have a reluctance towards institutionalising regular updates to regional plans as they are being implemented (the only reported instance I found is the KwaZulu-Natal plan in South Africa; Harris et al. 2012), likely due to the high costs and expertise often required during the spatial prioritisation process (e.g., Green et al. 2007a). My work in Chapter 3 contributes evidence towards the potential benefit of re-conceptualising spatial prioritisations as a more flexible process to produce dynamic outputs, and view plans as starting points that must inevitably and beneficially change over time.

Objective 3. Determine if there is an optimal scale at which to plan to address multiple social and ecological scales.

The critical knowledge gap I identified in Chapter 1 pertaining to this research objective is: elucidating the respective strengths and weaknesses of conservation plans developed at different levels in an SES context. I addressed this gap in Chapter 4 by evaluating plans in terms of their ability to address multiple social and ecological scales. The rationale behind this research objective was two-fold; first, to understand if there is an optimal scale at which to plan, and second, using this gained understanding to inform approaches that most effectively integrate planning across multiple scales. To achieve this, I collated conservation plans developed at all levels throughout the Coral Triangle region (six countries: Indonesia, Malaysia, Papua New

Guinea, Solomon Islands, and Timor Leste, including international plans) and evaluated these against an explicitly multiscale SES framework. Though no conservation plans I assessed were able to adequately address all social and ecological scales, encouragingly, I found that planning at different levels demonstrated varying strengths and weaknesses that complemented each other.

My research in Chapter 4 advances current insights into the strengths and weaknesses of plans developed at different levels, and is the first study to compare such plans in an explicitly multiscale SES context. I found that higher-level planning (regional, international) had the greatest capacity for scalar coverage and addressing regional ecological objectives; local-level planning considered social factors in more detail than other levels and engaged the greatest number of stakeholder groups. Limitations typically associated with planning at local versus regional levels in the conservation planning literature include, respectively, failing to consider broad-scale patterns and processes (e.g., Weeks et al. 2010a), and failing to incorporate local conditions and decision-making processes or fine-scale patterns or processes (e.g., Ban et al. 2011). Through this research, I extended understanding on the different kinds of limitations associated with planning at lower and higher levels, as being either conceptual or technical in nature. This understanding also underpinned my research in Chapter 5, where I identified distinct avenues through which planning at different levels are able to interact with each other and potentially overcome these different limitations. From my findings in Chapter 4, I also established specific recommendations to overcome these limitations through varying interactions between different levels of planning: (1) conceptual limitations might be overcome through workshops that promote sharing and learning of dissimilar perspectives relevant to each level, and (2) technical limitations might be minimised through exchanges of data, information or individuals between levels (e.g., using a standardised database system).

In this chapter I built upon existing theoretical frameworks to create novel insights by formalising the adapted SES framework proposed by Cumming et al. (2015) and terming the ability of plans to address multiple social and ecological scales as their 'scalar coverage'. Sustainably successful conservation planning can no longer consider only singular scales of social or ecological patterns and processes (Lengyel et al. 2014, Weeks et al. 2014, Cumming et al. 2015, Virapongse et al. 2016, Tengö et al. 2017). In creating the concept of scalar coverage, this work begins to view conservation planning processes as explicitly multiscale and provides a tractable and useful measure of the adequacy of conservation plans in this regard. Importantly, this research has led to key recommendations to overcome some of the limitations associated with single-level planning. There has been contention around whether scaling down or scaling up may be more effective approaches to integrate conservation planning across multiple scales (Lovell et al. 2002, Sievanen et al. 2013, Gaymer et al. 2014). I contribute to this debate with

findings that support, for the Coral Triangle region at least, the approach of initiating conservation planning at a high jurisdictional level (e.g., national or international) and iteratively cycling between lower and higher levels of planning (rather than simply moving in a top-down fashion associated with scaling down). Initiating planning at a high level will ensure that conservation planning progresses to lower levels of planning (e.g., provincial or local) from first being situated within a broad context, while also securing higher-level institutional capacities, which is often greatest at this level (Cuthill and Fien 2005). In this case, vertical integration is paramount (and has been suggested in many other institutional or policy-related contexts; Wells and McShane 2004, Lane and Robinson 2009, Pahl-Wostl 2009).

Finally, this work critically illustrates how conservation practitioners can use the SES framework to evaluate conservation plans to consider multiple social-ecological scales, which can help to overcome the shortfalls in planning at each level and move towards explicitly considering planning processes at other levels. Because the onus of ensuring vertical integration of conservation planning lies with those practitioners leading the processes, I argue that this explicit consideration of multiple scales and levels is the first step in integrating conservation planning across scales.

Objective 4. Investigate the theory and practice behind integrating conservation planning across multiple scales.

In Chapter 5 I analysed multiscale conservation planning in practice, using Papua New Guinea and the Solomon Islands as case studies, and evaluated factors that impeded or facilitated successful multiscale outcomes. Given the compelling evidence to consider systems socially, ecologically, and explicitly across multiple scales (Chapter 4) for more effective conservation planning and governance outcomes (Crowder et al. 2006, Folke et al. 2007, Mills et al. 2015, Mascia et al. 2017), this research is a vital first step towards unpacking how multiscale conservation planning is conceived in the literature and how it actually occurs in practice. Two fundamental knowledge gaps I identified in Chapter 1 pertaining to this research objective are: (1) understanding whether multiscale conservation planning occurs in practice and if so, through what mechanisms, and (2) discerning the factors that impede or facilitate successful outcomes in multiscale planning. My research in Chapter 5 addresses these two knowledge gaps by providing empirical evidence of multiscale conservation planning in practice. It is also the first study to determine the specific mechanisms through which multiscale conservation planning can occur: with different flows of planning resources between planning levels (consisting of data, objectives, practitioner experience, institutional support, and funding). Furthermore, this work presents the first comprehensive and empirical demonstration that the scaling down and scaling up dichotomy prevalent in the conservation planning literature likely does not exist in practice.

Instead, I provide substantiation that a multidirectional scalar pathway is probably more representative and propose a novel archetype to describe multiscale planning in practice.

In evaluations of empirical scalar pathways, I identified a number of scale-explicit factors that influenced successful outcomes across multiple levels of planning. Because these factors were not adequately addressed in existing evaluation frameworks of conservation capital (i.e., natural, human, social, financial, and institutional; Bottrill and Pressey 2012), I propose that future evaluations of conservation planning include considerations of a new and equally essential form of capital, 'scalar capital', which I define by its dimensions of scale-explicit attributes. Significantly, I advanced the literature on factors that can impede and facilitate successful multiscale conservation planning through identifying the six dimensions of scalar capital (with dimensions 3-6 needing to be explicitly multiscale): (1) multiscale understanding, (2) scale jumping, (3) leadership characteristics, (4) stakeholder engagement, (5) policy frameworks, and (6) institutional settings. The fact that the latter four dimensions are familiar concepts in the literature is encouraging; these attributes recognised as essential to successful conservation planning at a single scale should also play vital roles across multiple scales. Further, this research explicates the various roles of each dimension and outlines, particularly in relation to the first two dimensions novel to the conservation planning literature, specific underlying mechanisms and outcomes.

This research has considerably progressed our understanding on both the theory and practice of multiscale conservation planning and is the first study to explicitly tackle these concepts (cf., Guerrero et al. 2015a). Crucially, I draw on multiple bodies of literature to inform this work, making my findings broadly relevant to other important fields of collaborative governance (Ansell and Gash 2007), policy and social networks (Bodin 2017), advocacy coalition (Weible et al. 2011), social-ecological systems and ecosystem services (Scholes et al. 2013, Rozas-Vásquez et al. 2018), and adaptive management (Palomo et al. 2014). Consequently, there is much scope to explore further the concepts I have identified with this research, particularly in the increasingly pertinent context of spatial resilience and adaptive capacities of SES (Cumming et al. 2017). In addition to expanding the considerations of current evaluation frameworks for conservation outcomes across multiple scales, I provide empirical evidence on specific ways that dimensions of scalar capital can be fostered, and present recommendations that can help conservation scientists, policymakers, and practitioners intentionally foster scalar capital in future multiscale conservation planning for improved multiscale outcomes. For example, I observed that the concept of local 'keystone actors' (i.e., those who have a profound and disproportionate effect relative to other actors on their environmet; Österblom et al. 2015) was pertinent as a lens to understand how the enabling mechanisms of scale-jumping outcomes could be facilitated. Where

conservation practitioners proactively encouraged such individuals to enter formal institutions, these individuals would become instrumental in enabling the scale-jumping outcomes of contextualising (whereby decisions at lower levels are placed into context of higher levels) and grounding (whereby decisions at higher levels account for constraints and opportunities at lower levels).

6.3 Remaining gaps and scope for future work

A shortcoming of the research in Chapter 2 was the reef-class maps used, in that maps used for all prioritisation scenarios were derived using one method for categorising satellite imagery from the same mapping project (Millennium Coral Reef Mapping Project; Andréfouët et al. 2006). This means that all the reef-class data I used are associated with the same set of limitations around uncertainties of mapping and data interpretation accuracy, despite being derived through superior Landsat sensors (Andréfouët et al. 2006). With my results demonstrating the complexities in actions and interactions possible between different prioritisation factors, future studies in this area may further add to this understanding by examining individual and interaction effects on spatial priorities using different ecological dataset types (such as species richness; Araújo 2004, or ecological processes; Possingham et al. 2005). Whilst my analysis considered only one spatially variable socioeconomic cost layer, use of different types of cost layers could yield contrasting effects with the diversity of socioeconomic costs now considered in conservation planning (e.g., as social or cultural values; Chan et al. 2012, Whitehead et al. 2014, as stakeholder-specific objectives; Gurney et al. 2015).

The research in Chapter 3 is the first study to tackle quantitative investigation of iterative conservation planning processes and, as such, was not without its shortcomings. These relate primarily to the design of the simulations, where conservation actions were applied as a function of remaining objectives and rarity of conservation features, as well as the other rulesets employed to emulate real-world decision making within a specific planning context (i.e., developing economies with customary governance systems of natural resources common in the Coral Triangle and southwest Pacific region). Moreover, because accurate and complete spatial data on management units do not exist at a national extent in any of these regions (and, in any case, such units are open to flexibility and negotiation by local communities), the management unit layer used in my simulations had to be programmatically generated. However, the structure of my simulation framework is designed to be flexible in terms of altering decision rulesets that can represent different planning contexts. There is considerable scope here for future research to explore a wide range of important iterative planning scenarios, such as how errors in regional

data, or new fine-resolution data (see Pressey et al. 2013), might also influence the process of iteratively transitioning from designs to actions.

My research in Chapter 4 was limited in that it evaluated strengths and weaknesses from planning documentation alone. A more critical assessment would focus on actual impact and effectiveness (Ferraro and Pattanayak 2006). Nevertheless, I argue that plan documentation plays a fundamental role in the accountability and transparency of conservation planning processes (Margules and Pressey 2000), and should thus be representative of a planning process. However, future research that expands on this work would involve assessments of the strengths and weaknesses of conservation impact (itself a relatively recent development in conservation planning; Ferraro and Pressey 2015, McKinnon et al. 2015, McIntosh et al. 2017, Sterling et al. 2017, Barnes et al. 2018) in an explicitly multiscale SES context (Cumming et al. 2015). This understanding would also be valuable in understanding the discrepancies between what is planned for and what is actually achieved in conservation within a multiscale context.

A shortcoming of my research in Chapter 5 can be attributed to the small number of case studies identified with the two study regions, Papua New Guinea and the Solomon Islands, which operate under very similar governance contexts (i.e., highly decentralised systems) with developing economies and similarly complex socio-political and cultural structures (Govan et al. 2009). It is likely that scalar pathways elicited for contrasting governance contexts and institutional capacities (e.g., centralised systems such as in Australia; Day and Dobbs 2013) would differ to some extent. Furthermore, with limited institutional (and hence planning) capacity common in the Coral Triangle region (Green et al. 2011) and many planning processes operating in ad-hoc fashion (Department of Environment and Conservation Papua New Guinea 2011, Goby 2013), inclusions of all conservation planning processes would likely alter the final scalar pathways elicited (though again, they would very likely remain multidirectional). Future research evaluating scalar pathways in distinct governance and planning contexts would be valuable contributions in expanding this work. My evaluations were based on perceived outcomes of success by conservation practitioners; extensions of this research could include assessing perceived outcomes of success by other stakeholder groups (e.g., local community members, industry partners), or more ideally, involve empirical measures of increases in conservation capital (natural, human, social, financial, institutional, and scalar). Finally, I identify a novel and thus under-explored area of research, which can greatly be expanded upon in future studies: investigating the causal and interactive relationships between distinct scale jumping mechanisms and outcomes. This novel research area has potentially significant implications for improving multiscale conservation governance, the effectiveness of which is now paramount more than ever (Clement and Standish 2018).

6.4 Returning to real-world context

By means of systematic investigations of different stages throughout the conservation planning framework, this thesis contributes specific recommendations towards making scale influences and problems in conservation planning more tractable. The ultimate goal of this thesis was to produce practical recommendations to improve the conservation planning framework to deal with scale more explicitly. In doing so, I also expose how my findings pertinent to the different framework stages work to feed back into other stages, since planning processes should progress non-linearly through the 11 stages (Figure 6.1; Pressey and Bottrill 2009). When scoping conservation planning processes, my results from Chapter 4 indicate that planning must be conducted at different jurisdictional levels to have the greatest chance of addressing multiple relevant social and ecological scales, thus making vertical integration and iteration between levels of planning imperative. I also distinguished between types of limitations associated with different levels of planning in Chapter 4 (i.e., conceptual versus technical), which corresponded with the different flows of resources I identified to occur between planning levels in Chapter 5. These different resource flows identified in Chapter 5 demonstrate actual mechanisms through which vertical integration can and does occur between conservation planning processes at different levels. Both Chapters 4 and 5 substantiate the necessity of iterating between different levels of planning to account for scale mismatches. In Chapter 3, where I quantitatively investigate iterative planning processes and identify potential benefits to applying such a strategy, my conclusions from Chapter 2 are informative in illustrating the importance of excluding coarse-resolution socioeconomic cost data in regional conservation assessments. Incorporating spatially variable cost data into regional assessments can inaccurately and inappropriately bias coarse priorities and potentially reduce the likelihood of incidentally representing valuable fine-resolution features. Finally, I propose a novel concept of capital that emerged from my research in Chapter 5, that can facilitate applying the recommendations from this thesis to conservation planning in practice, and ultimately, enable processes to more effectively and explicitly consider multiple scales for improved outcomes.

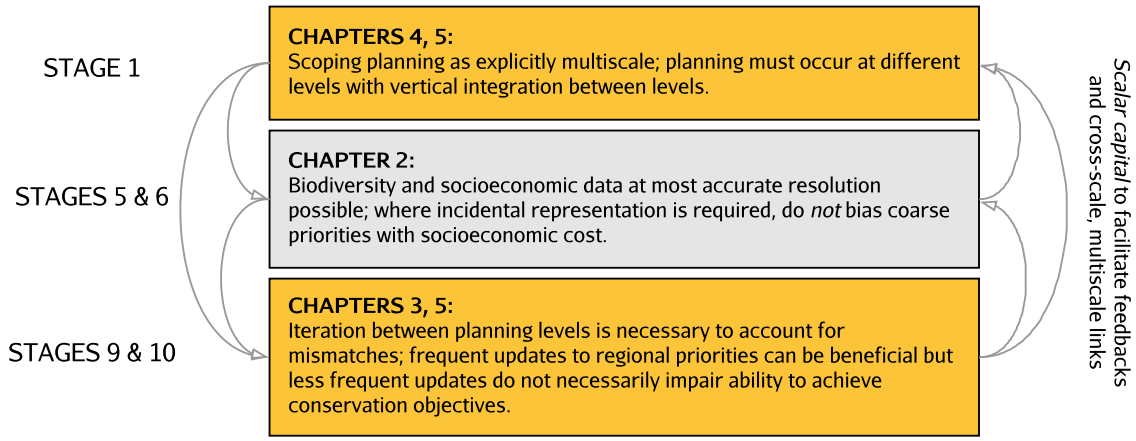


Figure 6.1 Summary of the contributions of each chapter in this thesis, in the context of the conservation planning framework and the relevant stages. Stage 1 involves scoping and costing the planning process; stages 5 and 6 involve collection of relevant biodiversity or socioeconomic data; stages 9 and 10 involve the selection of additional conservation areas and application of conservation actions (Pressey and Bottrill 2009). Orange-coloured boxes signify practical concerns related to scale; grey-coloured box, technical concerns.

6.5 Conclusions

There is overwhelming evidence that the concept of scale can no longer be considered implicitly throughout the systematic conservation planning process. Conservation planning must continue to progress as a field to consider multiple social and ecological scales from the outset of the process. To tackle the increasingly global and intricate nature of environmental problems and threats to biodiversity, ecosystem functioning, and human well-being, conservation scientists, policymakers, and practitioners must move towards viewing planning at individual levels as one complex system and build a 'planning system identity', to ensure successful outcomes that can be sustained, across multiple relevant scales. This thesis provides a contribution towards this endeavour by highlighting stages and feedbacks in the conservation planning framework within which practitioners can deal with scale more explicitly, and by presenting specific recommendations by which they might do so.

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Appendices

Appendix 1. Chapter 2 Supplementary materials

Text A1.1

Additional methods for creating the spatially variable socioeconomic cost layer.

For both study regions, I derived the variable cost layers by calculating weighted linear distance decays from fisher populations to the furthest reef areas, using fisher densities as a proxy of fishing effort. It is important to note that there are a number of crude assumptions that are necessary when using such a cost layer. First, that distance to fisher populations is a reliable proxy for effort; this assumes that fishers are more likely to travel shorter distances to fish, regardless of their method of transportation. There is also the assumption that fishing pressure decreases linearly across space. Moreover, as I did not distinguish between subsistence and commercial fishers when calculating fisher densities, the combined costs would likely result in an inequitable impact to the different fishery groups (Ban and Klein 2009). However, this method of deriving cost is relatively common, particularly in developing countries where explicit socioeconomic data is sparse (Ban et al. 2009). Evidence suggests this socioeconomic cost proxy represents cost better than others (e.g., general population numbers or simply area as cost; Weeks et al. 2010). Ultimately, the actual cost values and method used to derive these for the spatially variable cost was not so critical for the purposes of this study, since the key aim here was to compare the effects between using spatially uniform and spatially variable costs.

Spatially variable costs for Fiji

Total fisher population data were taken from the 2007 census (Fiji Bureau of Statistics 2007). Census data were collected at the administrative level of Enumeration Areas (EA), which are designated spatial units across the country for within which census information is collated. A linear distance decay function, weighted to population numbers, was used to simulate the decrease in opportunity cost as distance from fisher populations increased. This was done for two separate distances to represent the two main modes of transport used by fishers: non-motorised transport (e.g., walking, swimming, bilibili/bamboo raft), with costs decreasing to zero at 3 km from shore; and motorised transport (e.g., any boat with an engine) to unlimited distances (Adams et al. 2011, online supplementary material). For practical purposes, the 'unlimited' distance was set at 87 km, the minimum distance required to ensure that the cost layer covered all mapped coral reef habitats. The only exceptions to this were two small oceanic atolls, in the

southern extremity of Fiji's waters, more than 330 km to the nearest inhabited island. In this case, planning units across these two atolls were designated costs of zero.

Each decay function began at the fisher population number of each censused EA, decreasing linearly to zero for both decay distances (at a resolution of 30 m), to be summed together for the final cost layer (Figure A1.1a). As non-motorised transport is more common compared to motorised transport use amongst fishers, non-motorised and motorised costs were weighted to a ratio of 3:1, respectively. Both cost distances were then rescaled, on a scale of 0–100, to make costs between the different distances and between different datasets relative and allow for direct comparison. Once the weighted costs were summed, specific cost values were attributed to each planning unit based on the mean value of all cost values within the planning unit (Figure A1.1b).

Spatially variable costs for Micronesia

The same procedures were followed for the Micronesia dataset, with two exceptions. As the region includes two U.S. territories and three independent countries, the latest census data was sourced from the respective governments, which were all collected in different years (CNMI Department of Commerce 2010, Economic Policy Planning and Statistics Office 1999, Federated States of Micronesia Division of Statistics 2010, Office of Planning and Statistics 2005, U.S. Census Bureau 2010). The same distance decays were used (3 km for non-motorised transportation; 'unlimited' for motorised transport), however, these distances were weighted differently to Fiji. Due to a more developed socioeconomic context, there are a higher proportion of fishers using motorised boats compared to subsistence canoe or foot-fishers in the region (e.g., Mulyila et al. 2012). Based on this, the non-motorised and motorised distances were weighted to a ratio of 1:2, respectively.

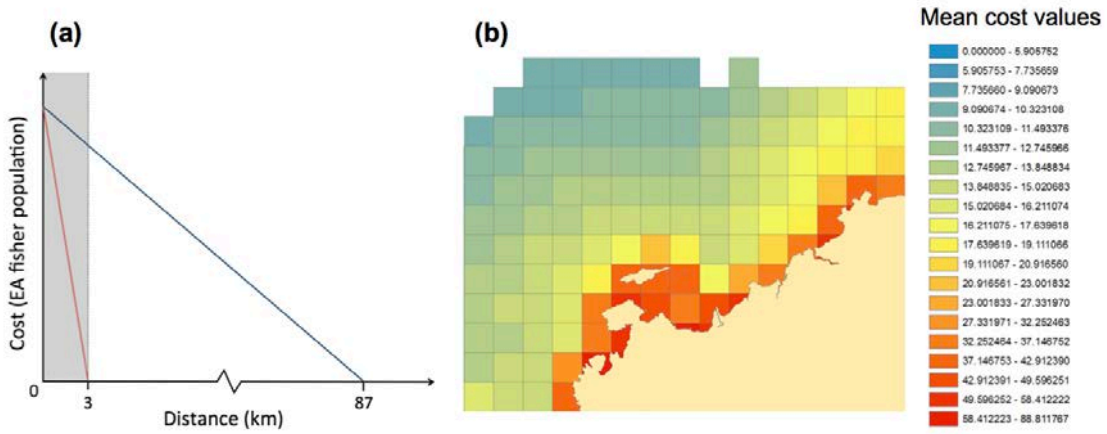

Figure A1.1 (a) Example of summed cost function (for Fiji). Non-motorised fisher cost (red line, to 3 km) was weighted three times more than motorised fisher cost (blue line, to 87 km). Where distances overlapped, weighted cost values were summed (grey shaded area). (b) Example section of map of the variable cost layer for large planning units (Fiji dataset).

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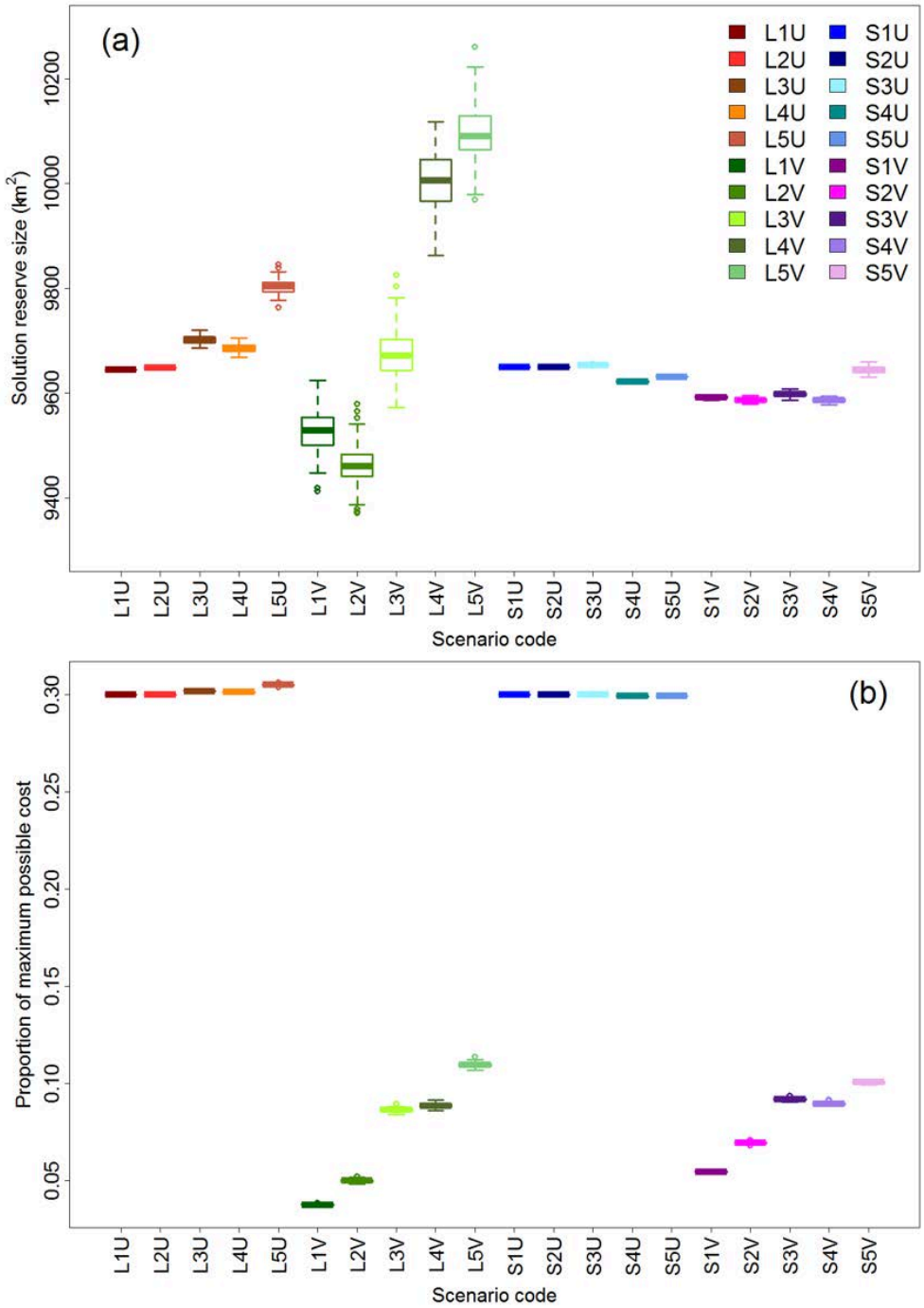

Figure A1.2 Comparisons of total reserve size and proportions of maximum possible cost. (a) Boxplots of ranges of reserve solution sizes for each scenario based on 100 replicate runs. (b) Boxplots of ranges of total costs (expressed as proportions of maximum possible cost) for each scenario based on 100 replicate runs. Each change in shade of the same colour represents the change in thematic resolution (always presented in order from 1-5, left to right) for each combination of planning-unit size and cost variability. Colour scheme representing all scenarios remains the same throughout all figures to facilitate interpretation.

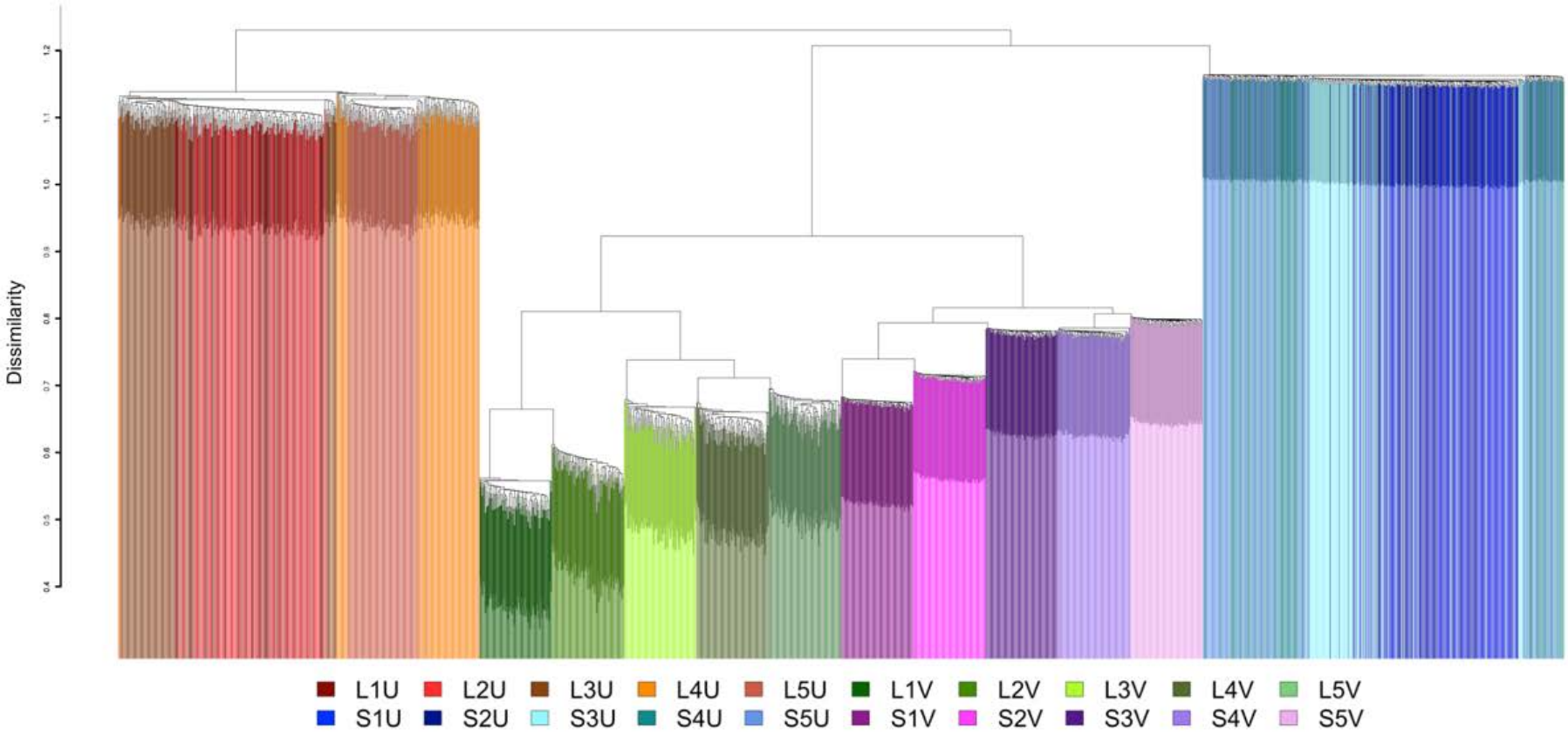

Figure A1.3 Spatial dissimilarity between all 2000 solutions for the Micronesia case study.

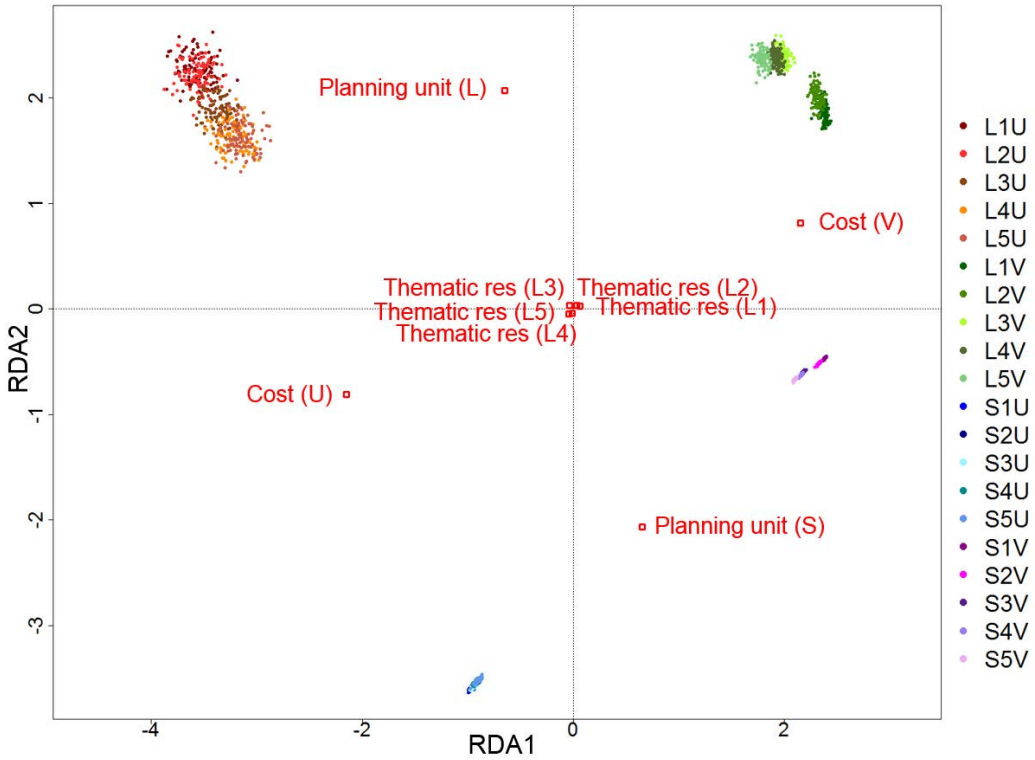

Figure A1.4 Comparison of spatial variation between all solutions produced using RDA for the Micronesia case study. Planning-unit size mainly explains variation along RDA1, while variation along RDA2 is mostly represented by cost variability. Red squares are centroids of the different levels of tested factors, representing the average amount of spatial variance that lines up with the plotted axes.

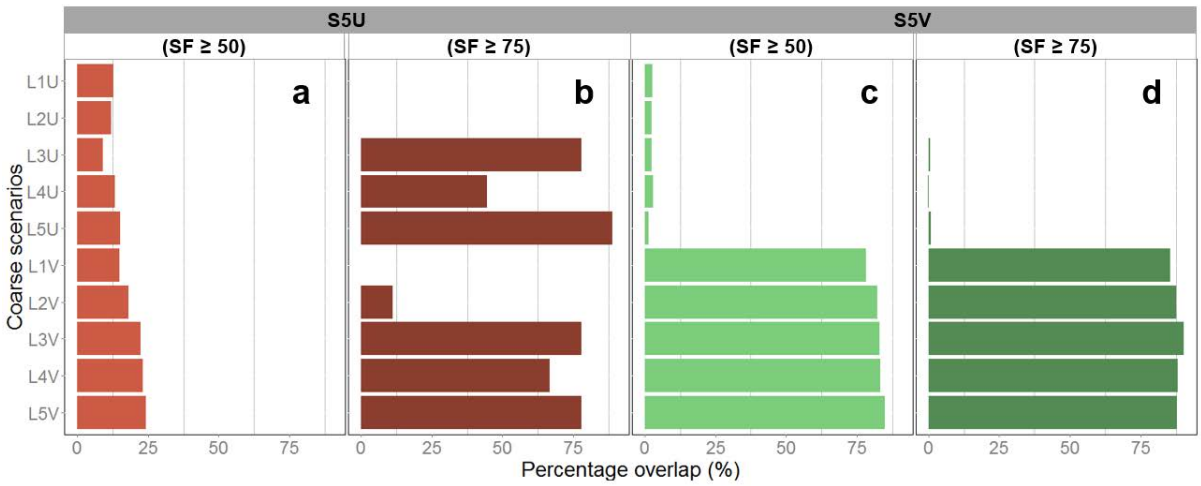

Figure A1.5 Nestedness of high-priority small planning units (test scenarios) within high-priority areas defined by large planning units. Nestedness of S5U high-priority areas, defined at selection frequency ≥ 50 (a) and selection frequency ≥ 75 (b). Nestedness of S5V high-priority areas, defined at selection frequency ≥ 50 (c) and selection frequency ≥ 75 (d).

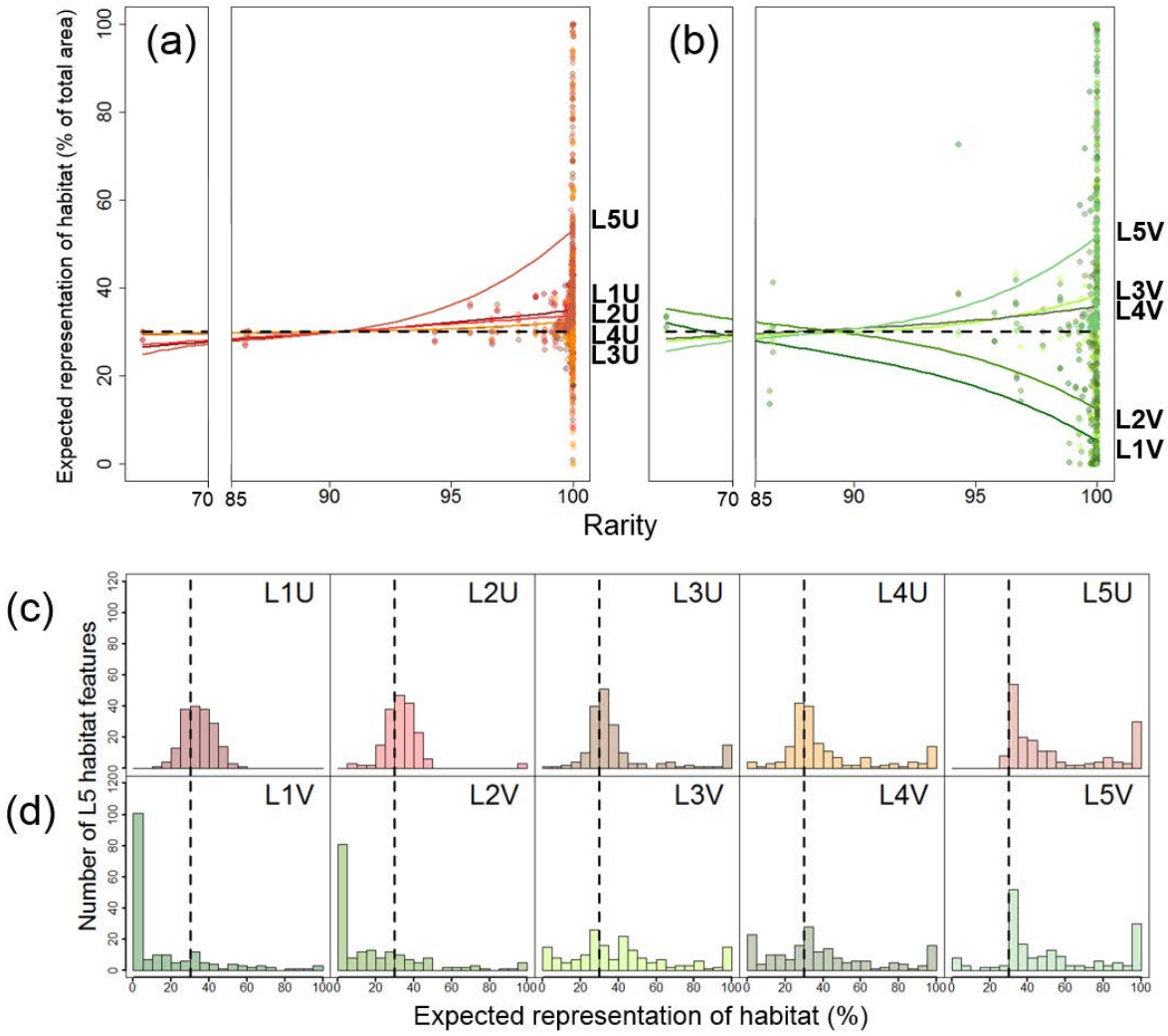

Figure A1.6 Incidental representation of L5 habitats by scenarios using large planning units. (a-b) Scatter plots showing expected representation of each L5 habitat (as a percentage of total area of feature occurrence) for each coarse scenario with uniform cost (a) and variable cost (b), in relation to habitat rarity (transformed to natural log). Due to spread and left-skewness of rarity values, plots are shown with *x*-axis breaks where no data occur to facilitate interpretation. Local regression (LOESS) curves were fitted for each coarse scenario, indicating non-linear trends in each scatter plot. Dashed horizontal lines represent the 30% objective for L5 habitats. (c-d) Histograms showing the distributions of expected representation of L5 habitats for coarse scenarios with uniform cost (c) and variable cost (d), plotted with 5% bin widths. Dashed vertical lines represent the 30% objective for L5 habitats.

Appendix 2. Chapter 3 Supplementary materials

Text A2.1

Specific parameters needed to be considered and rule sets defined for the relevant decision-making steps involved in simulating the transition from design to actions (Figure A2.1).

Determining proportion of planning unit for conversion to management units

Based on real-world constraints, only proportions of planning units were converted to implemented actions within management units (hereafter, 'management units') due to the spatial mismatch between the planning and management units. Therefore, a frequency distribution was created that realistically represented the distribution of proportions of each planning unit that would be converted to management units. I achieved this by taking spatial data on existing 'tabu' areas in Fiji and overlaid the planning-unit grid; where the two layers intersected, percentages of each planning unit containing tabu areas were calculated. This frequency distribution was adjusted by systematically shifting values upwards by uniform increments until the median of percentage values was at 30%. Doing so was necessary to allow the 30% conservation objective to be met in the simulations, since the original frequency distribution calculated revolved around much lower (and therefore insufficient for the purposes of my simulations) mean and median values (16% and 7%, respectively). Once the distribution was created, it was used to randomly assign conversion rates for each investigated planning unit to be converted to reserved management units (Step 3, Figure A2.1).

Another ruleset related to these conversion rates, involved setting a lower-bound tolerance of 20% of the originally selected value for considering conversion rates as achieved. Otherwise, values would always need to be exactly met or surpassed to be considered achieved, pushing my adjusted distribution in an even more unrealistic direction.

Assessing management units based on relative conservation importance

A ruleset was needed to define the order of management units that were selected for reservation within a planning unit, until the conversion rate was reached. This was achieved based on the conservation value of each management unit, relative to all other intersecting management units with the planning unit of investigation (Step 4, Figure A2.1). A sum of products method was employed to systematically determine this, so that all reef classes within each management unit could be assessed in terms of their relative regional rarity and remaining regional objectives to

achieve (methods described in Table A2.1). Rarity values of reef classes used to weight overall conservation value were calculated based on the proportion of occurrence of each feature across the whole planning region (details of method and standardising equations used in Text A2.3).

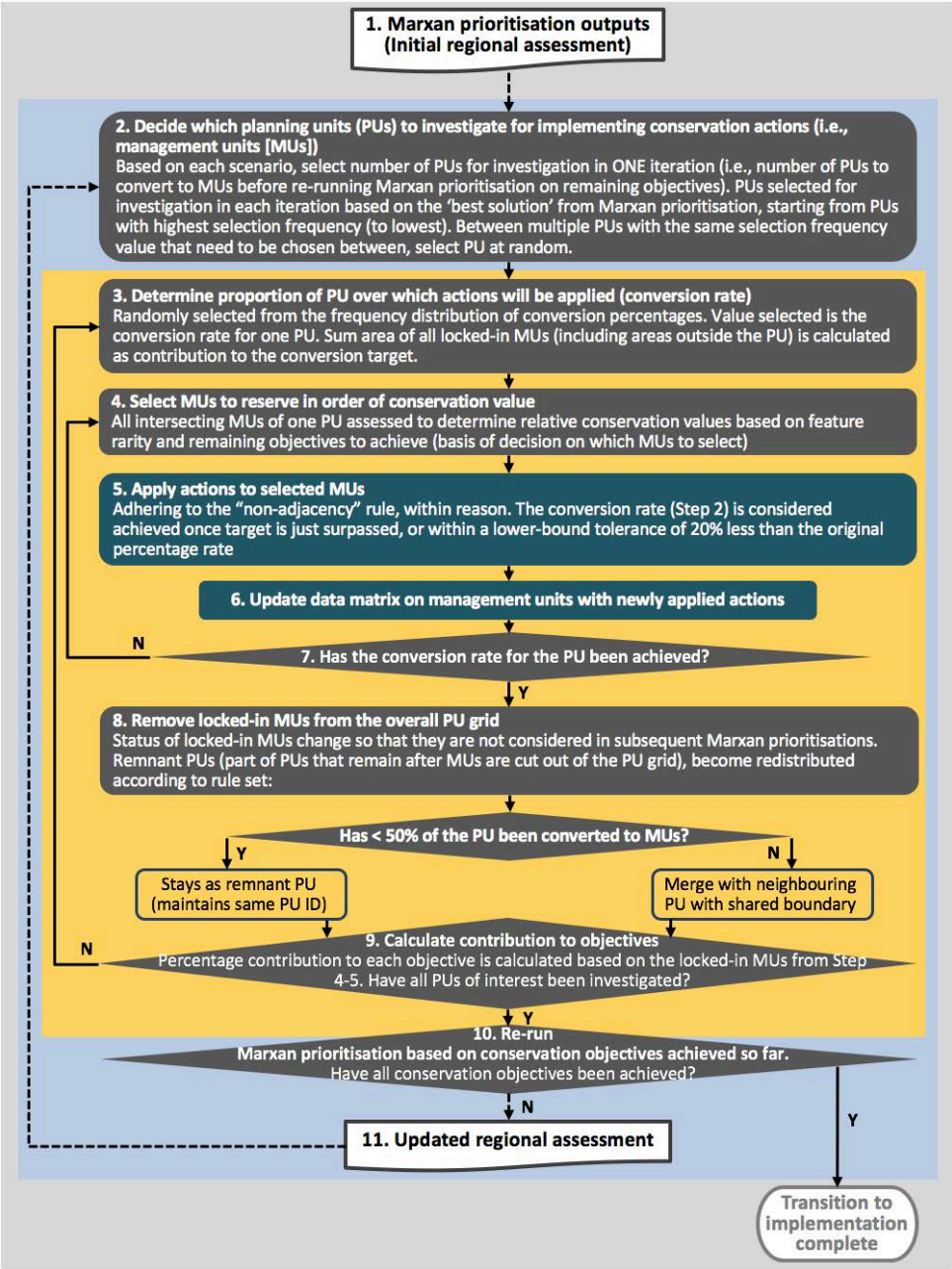

Figure A2.1 Operational coding-framework for the simulations. Solid arrows indicate action steps; dashed arrows indicate assessment inputs. Grey steps concern planning units. Blue steps concern management units. Note that decisions about planning units and management units might be undertaken by different organisations or teams. Yellow box indicates steps repeated for each investigated planning unit until the assigned conversion rate to implemented reserves is achieved. Light blue box indicates the steps involved in each iteration of the simulation, repeated

until all conservation objectives were achieved. Each simulation (grey box) was replicated 100 times.

Table A2.1 Method used to calculate the relative conservation value of each management unit, to determine the order of management unit selection within one planning unit investigated for conversion to management units.

Reef class code	Remaining regional objectives (30%)	Reef class rarity (weighting)	Product of remaining objective and reef class rarity
1	18	0.67	10.8
5	26	0.82	20.8
11	7	0.23	1.4
Sum products			33
Total planning unit area (km^2)			24.5025
Conservation value (sum products / area)			1.347

Application of conservation actions (management units)

Another real-world constraint added to the simulations was that, within the same iteration, selected management units intersecting the same planning unit could not be adjacent to each other (hereafter, 'the non-adjacency rule'; Step 5, Figure A2.1). This was done to emulate the common socioeconomic conflict, where setting up large tracts of reserves is not pragmatic and unlikely. However, the rule of achieving the conversion target of planning unit to management units was prioritised over the non-adjacency rule (i.e., adjacent management units were selected if this was the only option to achieve the necessary percentage conversion target). This was to ensure that the simulations could ultimately meet the overall conservation objective.

Updates to regional design

Once conservation actions have been applied within all planning units selected for investigation in one iteration (Steps 2-9, Figure A2.1), Marxan was re-run, taking into account the objectives achieved so far (Step 10, Figure A2.1) and the modified planning unit layer based on reserved management units (Step 8, Figure A2.1). Remnant planning units < 50% of the original planning unit size (i.e., < 12.25 km^2) were merged with a neighbouring planning unit to avoid the creation of very small planning-unit slivers or fragments that would inevitably be created as the simulations progressed. These very small slivers or fragments would likely have caused a bias selection of these units towards the end of the simulations, when only smaller areas are necessary to achieve objectives and for so little cost (since cost equalled area). Once Marxan was re-run and regional priorities recalculated, this stage concluded one complete iteration of the simulation (light blue box, Figure A2.1). Iterations continued until the conservation objective of 30% of

each reef class was achieved, at which point the simulation would be complete (grey box, Figure A2.1).

Text A2.2

Detailed description of deriving planning and management units used in simulations

Planning units

Size of planning unit used was set at 25 km^2; however, for the purposes of my simulations, this size was adjusted to perfectly fit the resolution of the underlying habitat data used to prioritise the regions (30 m), since all spatial data used here were dealt with in raster format. Therefore, the final planning unit size used was adjusted to 24.5025 km^2.

Management units

A complete spatial layer representing management units in Fiji does not exist for the whole region. Thus, I created this spatial layer based on frequency distributions of known management unit sizes in Fiji. The size distribution of these management units was calculated from information on existing tabu areas in Fiji, which are traditionally village-managed closures (Jupiter and Egli 2010). Tabu areas were used since these effectively represent actual sizes at which conservation management actions are applied in Fiji. The calculated frequency distribution was used to inform the individual sizes of each irregular management unit by selecting a value at random from the distribution when creating each unit. By randomly selecting a starting cell of a 30x30 m grid representing the region, the unit was created by adding a random proportion of neighbouring cells until the unit size was reached. This process was repeated until a complete management unit spatial layer was created covering all reef areas.

References

Jupiter, S. D., and D. P. Egli. 2011. Ecosystem-based management in Fiji: successes and challenges after five years of implementation. *Journal of Marine Biology* 2011:1–14.

Text A2.3

Method used to calculate feature rarity values

Each habitat feature was calculated as a proportion of the overall study area. These were then subtracted from a value of one so that the more rare the feature, the higher the rarity value. All values were then normalised to range from 0-1 using the following equation (eq. 1), since a linear distribution of values was more appropriate so that there was more differentiation between rarity values and values were not clustered and biased towards only one side of the scale:

$$x_{new} = \frac{x - x_{\min}}{x_{\max} - x_{\min}} \qquad \text{(eq. 1)}$$

Though all rarity values were normalised to fit the scale of 0-1, the values needed to then be converted to fit a scale of 0.1-1. This is because features could not have a rarity value of '0', as this would mean that the most common feature would not contribute towards the conservation value of the management unit under investigation for selection. This was achieved using the following linear equation (eq. 2):

$$y = a + (x - A) \times \frac{(b - a)}{(B - A)} \qquad \text{(eq. 2)}$$

Where *a* is the minimum value of the scale to convert to (in this case, 0.1); *b* is the maximum value of the scale to convert to (i.e., 1); *A* is the minimum value of the current scale (i.e., 0); and *B* is the maximum value of the current scale (i.e., 10). Therefore eq. 2 is adapted to:

$$y = 0.1 + (x - 0) \times \frac{(1 - 0.1)}{(1 - 0)} \qquad \text{(eq. 3)}$$

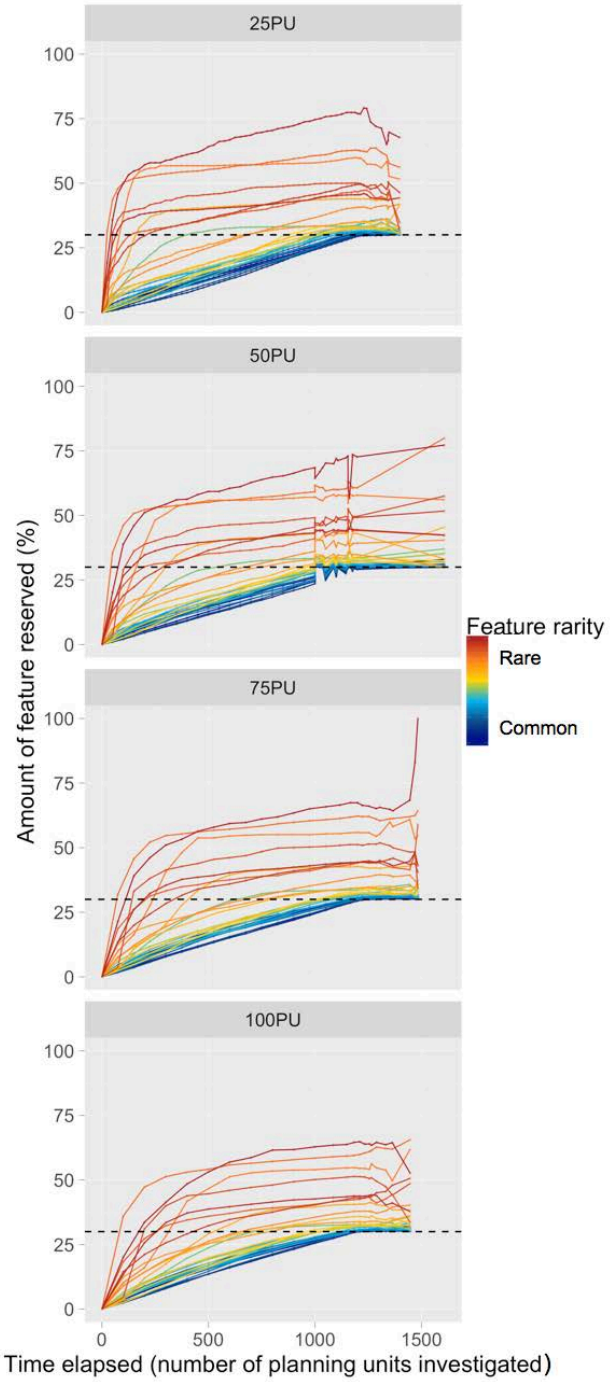

Figure A2.2 Average total times taken to achieve all conservation objectives across scenarios, indicated by total numbers of planning unit investigations throughout the simulations. Graphs show average amounts of feature reserved over 100 replicates for each scenario (Scenario codes: '25PU' = scenario 25; '50PU' = scenario 50; '75PU' = scenario 75; '100PU' = scenario 100). Apparent differences in time elapsed appear due to a few individual replicates; average values across scenarios do not significantly differ. The rate of implementation was equal across scenarios; for example, the regional assessment in scenario 100 was updated four times less frequently than scenario 25 across the length of the simulations. The 31 reef classes are coloured by gradient to reflect the order of rarity; blue indicates the most common feature, red, the most rare.

Appendix 3. Chapter 4 Supplementary materials

Table A3.1 Summary of scalar coverage assessed for each conservation plan (*n* = 18), with total number of stated objectives addressing each ecological and social level (path, local, regional, international), and the adequacy with which ecological objectives were addressed. Ecological adequacy assessments: qualitative (QL), quantitative no rationale (QN), quantitative subjective (QS), and quantitative ecologically justified (QE). Grey cells indicate the level at which each conservation plan was developed.

				ECOLOGICAL OBJECTIVES							
				Patch level		Local level		Regional level		International level	
Case study	Country	Plan level	Lead plan org. †	Total no.	Adequacy	Total no.	Adequacy	Total no.	Adequacy	Total no.	Adequacy
Sulu-Sulawesi Marine Ecoregion Plan	Philippines; Malaysia; Indonesia	International	WWF	1	QL	2	QL; QL	3	QL; QL; QL	1	QL
Coral Triangle Marine Protected Area System	Coral Triangle	International	WWF; TNC; CI; UQ	1	QN	2	QN; QS	3	QE; QE; QS	3	QE; QE; QN
Land-Sea Conservation Assessment for Papua New Guinea	Papua New Guinea	Regional	PNG CEPA; UQ; TNC	3	QN; QN; QL	3	QN; QN; QL	2	QE; QN	1	QL
Ridges to Reefs Conservation Plan for Solomon Islands	Solomon Islands	Regional	SI MoE; JCU; TNC	0		1	QS	1	QS	0	
Lesser Sunda Ecoregion Marine Protected Area Network	Indonesia	Regional	TNC	2	QS; QL	3	QS; QN; QL	7	QE; QL; QL; QL; QL; QL; QL	0	
Raja Ampat Marine Protected Area Network	Indonesia	Regional	TNC; WWF; CI	2	QS; QL	5	QE; QS; QN; QL; QL	6	QE; QL; QL; QL; QL; QL	0	
Kimbe Bay Marine Protected Area Network	Papua New Guinea	Regional	TNC	1	QL	3	QS; QL; QL	7	QE; QS; QL; QL; QL; QL; QL	0	
Choiseul Province Ridges to Reefs Protected Area Network	Solomon Islands	Regional	TNC; WWF; LL	4	QN; QN; QL; QL	4	QN; QL; QL; QL	5	QN; QN; QL; QL; QL	0	
Isabel Province Ridges to Reefs Protected Area Network	Solomon Islands	Regional	TNC; WWF; WorldFish	1	QN	3	QN; QL; QL	4	QN; QL; QL; QL	0	
Roviana and Vonavona Lagoons Marine Protected Area Network	Solomon Islands	Regional	UCSB; TCF; WWF; CFC	2	QL; QL	1	QL	2	QS; QL	0	
Wakatobi Marine National Park	Indonesia	Local	TNC; WWF	1	QL	2	QN; QL	0		0	
Nusa Penida Marine Protected Area	Indonesia	Local	CTC	1	QL	2	QL; QL	0		0	
Tubbataha Reef Natural Park	Philippines	Local	WWF; CI	1	QL	1	QL	0		0	
Sinub Island Wildlife Management Area	Papua New Guinea	Local	WI-O	2	QL; QL	2	QL; QL	1	QL	0	
Nino Sanis Santana Marine National Park	Timor Leste	Local	MAF; NTG; CDU	2	QL; QL	4	QL; QL; QL; QL	2	QL; QL	0	
Tun Mustapha Park	Malaysia	Local	WWF; UQ; UMS	3	QS; QN; QL	3	QS; QN; QL	0		0	
Kakarotan Island Mane'e	Indonesia	Patch	Kakarotan community	2	QL; QL	0		0		0	
Muluk Village Traditional Closure	Papua New Guinea	Patch	Muluk community	1	QL	0		0		0	

Table A3.1 *Continued.*

				SOCIOECONOMIC OBJECTIVES			
				Patch	Local	Regional	International
Case study	**Country**	**Plan level**	**Lead plan organisation †**	**Total no.**	**Total no.**	**Total no.**	**Total no.**
Sulu-Sulawesi Marine Ecoregion Plan	Philippines; Malaysia; Indonesia	International	WWF	0	4	5	2
Coral Triangle Marine Protected Area System	Coral Triangle	International	WWF; TNC; CI; UQ	0	0	0	0
Land-Sea Conservation Assessment for Papua New Guinea	Papua New Guinea	Regional	PNG CEPA; UQ; TNC	1	4	5	0
Ridges to Reefs Conservation Plan for Solomon Islands	Solomon Islands	Regional	SI MoE; JCU; TNC	0	0	0	0
Lesser Sunda Ecoregion Marine Protected Area Network	Indonesia	Regional	TNC	5	6	3	0
Raja Ampat Marine Protected Area Network	Indonesia	Regional	TNC; WWF; CI	4	10	3	0
Kimbe Bay Marine Protected Area Network	Papua New Guinea	Regional	TNC	5	10	1	0
Choiseul Province Ridges to Reefs Protected Area Network	Solomon Islands	Regional	TNC; WWF; LL	0	2	2	0
Isabel Province Ridges to Reefs Protected Area Network	Solomon Islands	Regional	TNC; WWF; WorldFish	0	1	1	0
Roviana and Vonavona Lagoons Marine Protected Area Network	Solomon Islands	Regional	UCSB; TCF; WWF; CFC	1	4	2	0
Wakatobi Marine National Park	Indonesia	Local	TNC; WWF	1	1	0	0
Nusa Penida Marine Protected Area	Indonesia	Local	CTC	1	1	1	0
Tubbataha Reef Natural Park	Philippines	Local	WWF; CI	0	4	0	0
Sinub Island Wildlife Management Area	Papua New Guinea	Local	WI-O	1	2	1	0
Nino Sanis Santana Marine National Park	Timor Leste	Local	MAF; NTG; CDU	0	6	0	0
Tun Mustapha Park	Malaysia	Local	WWF; UQ; UMS	1	2	0	0
Kakarotan Island Mane'e	Indonesia	Patch	Kakarotan community	1	0	0	0
Muluk Village Traditional Closure	Papua New Guinea	Patch	Muluk community	1	0	0	0

† Lead planning organisations: World Wide Fund for Nature (WWF); The Nature Conservancy (TNC); Conservation International (CI); University of Queensland (UQ); Papua New Guinea Conservation and Environment Protection Authority (PNG CEPA); Live and Learn (LL); University of California Santa Barbara (UCSB); Tiola Conservation Foundation (TCF); Christian Fellowship Church (CFC); Coral Triangle Centre (CTC); Wetlands International – Oceania (WI-O); Timor Leste Ministry of Agriculture and Fisheries (MAF); Northern Territory Government (NTG); Charles Darwin University (CDU); Universiti Malaysia Sabah (UMS).

Table A3.2 Extent of stakeholder engagement assessed across all stakeholder groups at each SES level, for all conservation plan case studies (*n* = 18).

Conservation plans	Country	Plan level	Stakeholder level	STAKEHOLDER GROUPS NatGov	LocGov	IntNGO	LocNGO	RemAca	LocAca	Indus	Tour	Aqua	CommFis	SubFis	TraLead	LocComm
Kakarotan Island Mane'e	IND	P	P												P	
Muluk Village Closure	PNG	P	P			N					N				P	N
Wakatobi MPA	IND	L	L	P	P	N					N			C		I
			P	P	P	N					N			C		I
Nusa Penida MPA	IND	L	L	P	P		N		N		N	N		N	N	N
			P		P						N	N		N	N	
Tubbataha Reef Natural Park	PHI	L	L	P	P	N	N		N		C			C		C
Sinub Island WMA	PNG	L	L			N					N				P	N
			P								N				P	N
Nino Sanis National Park	TIM	L	L	P		N	N	N			C			C	N	N
Tun Mustapha Park	MAL	L	L	P	P	N			N				N		N	N
			P		P	N			N				N		N	N
Lesser Sunda Ecoregion MPA Network	IND	R	R	P	P	N			N							
			L	P	P	N			N							
Raja Ampat MPA Network	IND	R	R	P	P	N		N								
			L	P	P	N	N							N		N
Kimbe Bay MPA Network	PNG	R	R	P	P	N	N									
			L		P	N	N								P	P
			P			N	N								P	P
Choiseul Province R2R Protected Area Network	SI	R	R	P	P	N		N							P	P
			L		P	N									P	P
Isabel Province R2R Protected Area Network	SI	R	R	P	P	N			N	N	N				P	N
			L		P	N			N	N	N				P	N
Roviana and Vonavona Lagoons MPA Network	SI	R	R	P	P			N	C							
			L		P		N	N	C						P	P
			P												P	P
PNG National	PNG	R	R	P		N		N	N	N						
SI National	SI	R	*NA*													
Coral Triangle MPA System	INT	I	*NA*													
Sulu-Sulawesi Marine Ecoregion Plan	INT	I	I	P	P	N		C	C	N						I
			R	P	P	N	N	C	C	C	C			I	C	I
			L			N	N		C							C

* Country abbreviations: Indonesia (IND), Papua New Guinea (PNG), Philippines (PHI), Timor Leste (TIM), Malaysia (MAL), Solomon Islands (SI), International (INT). Stakeholder group abbreviations: national government (NatGov), local government (LocGov), international NGO (IntNGO), local NGO (LocNGO), remote academic (RemAca), local academic (LocAca), shipping and mining industry sector (Indus), tourism sector (Tour), aquaculture sector (Aqua), commercial fisheries (CommFis), subsistence fisheries (SubFis), traditional leaders (TradLead), local communities (LocComm). Abbreviations for conservation plan and stakeholder levels: patch (P), local (L), regional (R), international (I). Abbreviations for extent of stakeholder engagement: information (I), consultation (C), negotiation (N), and delegated power (P).

Appendix 4. Chapter 5 Supplementary materials

Table A4.1 Reports, management plans, and other relevant scientific or governmental publications on each planning process included in the scalar pathway case studies for Papua New Guinea and the Solomon Islands, identified through searches in the peer-reviewed and grey literature.

Country	Reference
Papua New Guinea	Almany, G. R. *et al.* (2015) *Local benefits from community actions: small managed areas can help rebuild and sustain some coastal fisheries*.
	Building Resilience to Climate Change (2016) 'Mwanus Endras Asi Tribal Leaders Network', in *International Symposium on Capacity Building for Sustainable Oceans*.
	Department of Environment and Conservation & National Fisheries Authority of Papua New Guinea (2010) *Papua New Guinea Marine Program on Coral Reefs, Fisheries and Food Security, National Plan of Action 2010-2013*.
	Department of Environment and Conservation Papua New Guinea (2011) *A protected area policy for a national protected area system for Papua New Guinea*.
	Filer, C. and Gabriel, J. (2017) 'How could Nautilus Minerals get a social licence to operate the world's first deep sea mine?', *Marine Policy*. Elsevier, pp. 1–7.
	Government of Papua New Guinea (2006) *Papua New Guinea National Biodiversity Strategy and Action Plan*.
	Government of Papua New Guinea (2015) *National marine conservation assessment for Papua New Guinea*.
	Green, A. *et al.* (2007) *Scientific design of a resilient network of marine protected areas Kimbe Bay, West New Britain, Papua New Guinea*.
	Green, A. *et al.* (2014) *A regionalisation of Papua New Guinea's marine environment: a technical report prepared for Papua New Guinea's Department of Environment and Conservation with support from the Australian government*.
	Independent State of Papua New Guinea (2014) *Papua New Guinea Policy on protected areas*.
	Independent State of Papua New Guinea (1997) *No. 3 of 1997. Organic Law on National and Local-level Government Elections*.
	Keppel, G. *et al.* (2012) 'Conservation in tropical Pacific island countries: case studies of successful programmes', *PARKS*, 18(1), pp. 111–124.
	Lipsett-Moore, G. *et al.* (2010) *Interim national terrestrial conservation assessment for Papua New Guinea: protecting biodiversity in a changing climate, Pacific Island Countries Report No 1/2010*.
	Lipsett-moore, G. *et al.* (2006) *Guidelines for a community-based planning process, Kimbe Bay MPA Network, Kimbe Bay, West New Britain, Papua New Guinea*.
	Lipsett-Moore, G. *et al.* (2017) *Ridges to reefs assessment for New Britain, PNG: planning for responsible, sustainable development. TNC Pacific Division Report No. 2/10*.
	Lipsett-Moore, G. *et al.* (2017) *Ridges to reefs assessment for New Britain, PNG: planning for responsible, sustainable development. TNC Pacific Division Report No. 2/10*.
	Meharg, S., Wise, R. M. and Butler, J. R. A. (2016) *Decision-making case studies summary report: building capacity for adaptive governance of the Bismarck Sea, Papua New Guinea*.
	Menazza, S. (2010) *Conservation law benefits communities and biodiversity, Papua New Guinea*.

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Papua New Guinea Department of Environment and Conservation (2009) *Supporting Country Action on the CBD Programme of Work on Protected Areas*.

Smith, M. P. L. *et al.* (2002) 'The Arnavon Islands Marine Conservation Area: lessons in monitoring and management', in *Proceedings of the 9th International Coral Reef Symposium 2000*.

The Nature Conservancy's PNG Country Program and the Madang Provincial Government (2013) *Madang province spatial planning report*.

Solomon Islands

Goby, G. (2013) *Guidelines for community-based marine monitoring in the Solomon Islands*.

Hamilton, R. J. *et al.* (2015) 'Solomon islands largest hawksbill turtle rookery shows signs of recovery after 150 years of excessive exploitation', *PLoS ONE*, 10(4), p. e0121435.

James, R. (2013) *Isabel forum on mining: Waswe iumi redi for mining? (Are we ready for mining?) Isabel Provincial Assembly, Buala, Isabel, Solomon Islands*.

James, R. (2015) *National forum on mining: pathways to a better mining industry in Solomon Islands*.

Kereseka, J. (2014) 'Successful community engagement and implementation of a conservation plan in the Solomon Islands: a local perspective', *PARKS*, 20(1), pp. 29–38.

Martin, S. (2013) *Marine protected areas in Solomon Islands: establishment, challenges, and lessons learned in Western Province*.

Ministry of Environment Climate Change Disaster Management & Meteorology (2014) *Solomon Islands: 5th National report on the implementation of the Convention of the Biological Diversity*.

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van Beukering, P. J. H. *et al.* (2007) *Case study 2: Arnavon Community Marine Conservation Area (Solomon Islands): the role of marine protected areas in contributing to poverty reduction, Brisbane: The Nature Conservancy*.

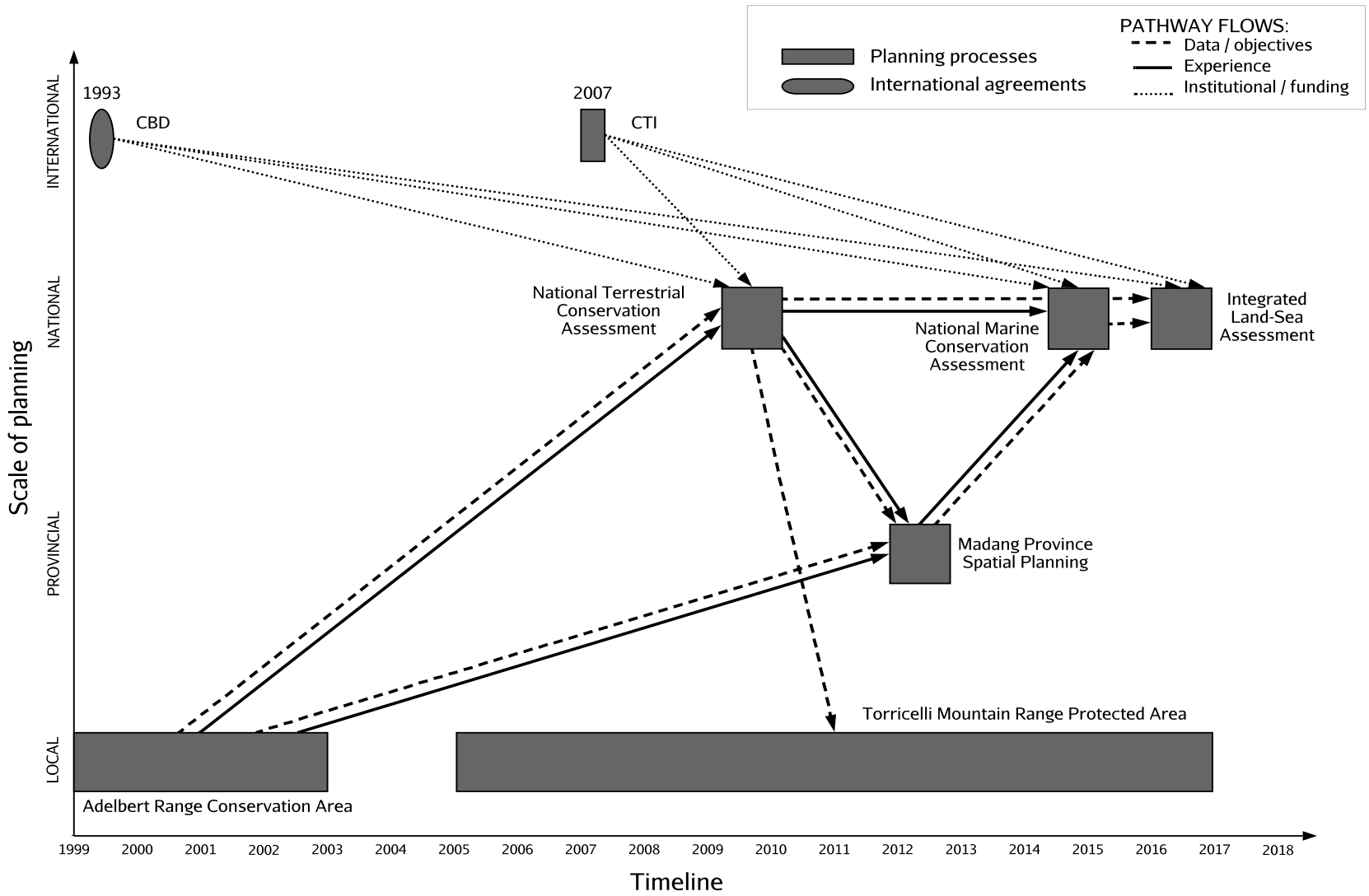

Figure A4.1 Empirical scalar pathway 1, based on the largest mainland region of Papua New Guinea (eastern half of New Guinea island).

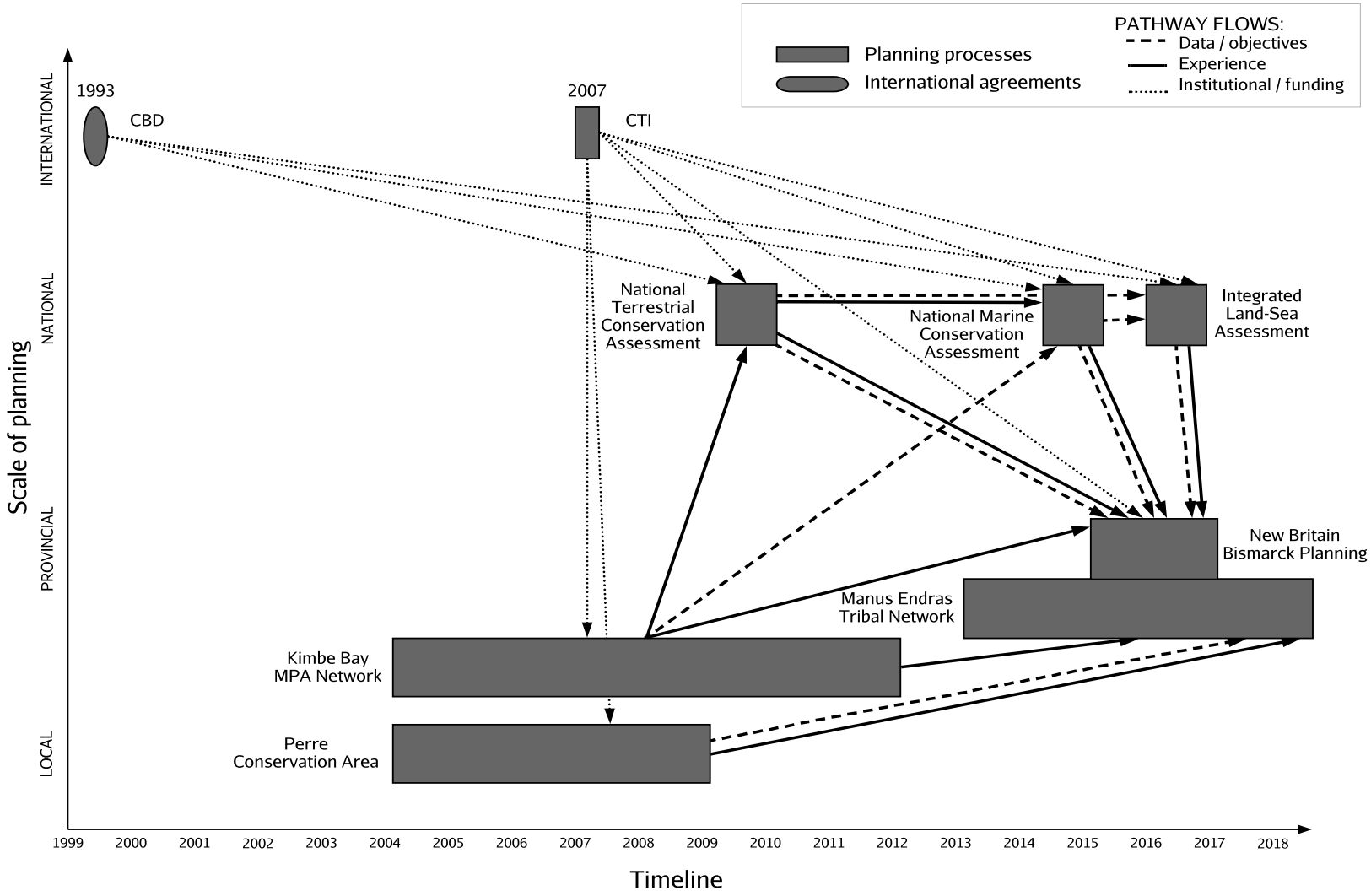

Figure A4.2 Empirical scalar pathway 2, based on the smaller major islands of New Britain and Manus in Papua New Guinea.

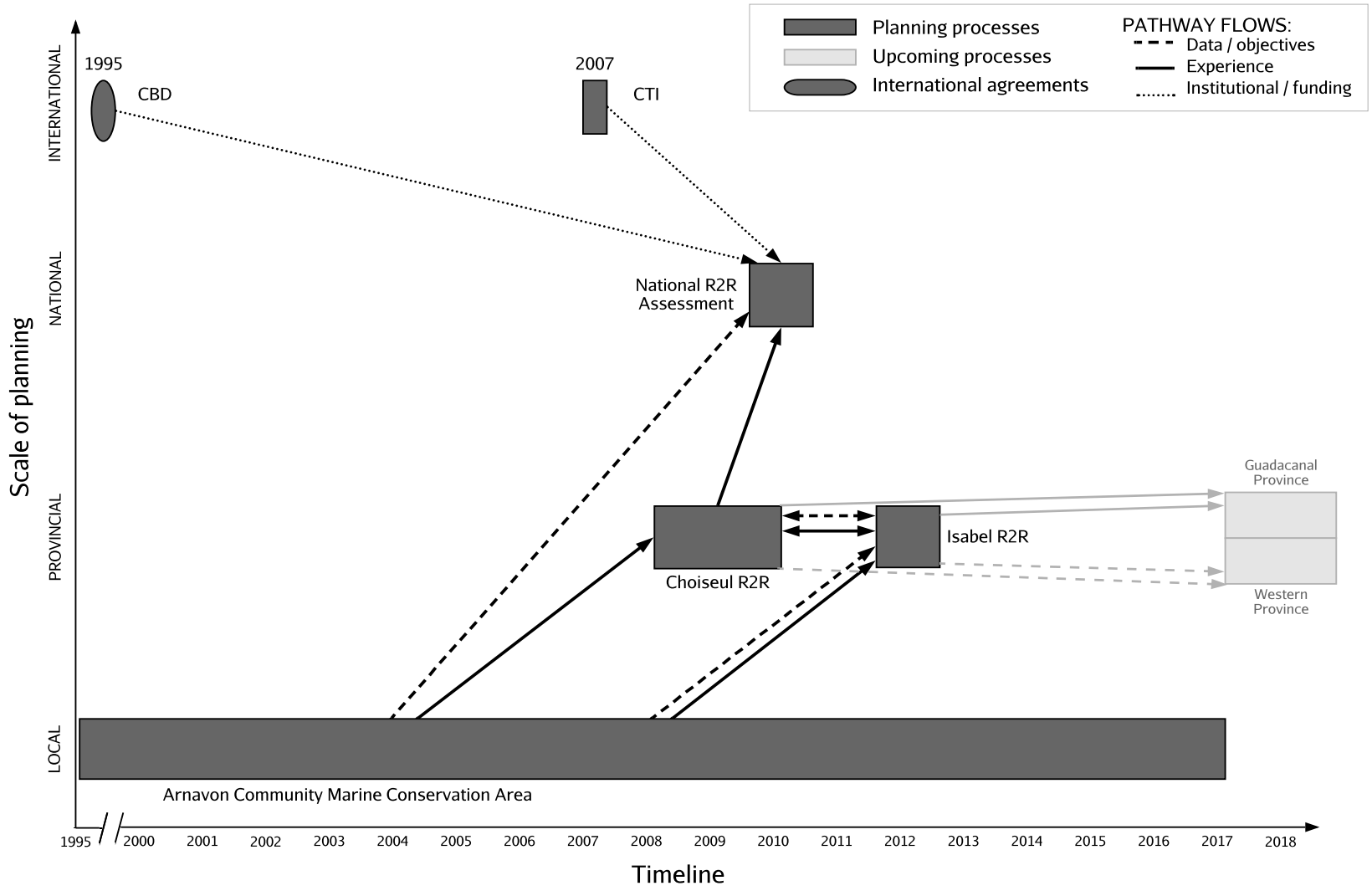

Figure A4.3 Empirical scalar pathway 3, based on the Solomon Islands.

Table A4.2 Combinations of types of scale jumping and enabling mechanisms, with observed examples identified from interviews with key informants involved in conservation planning in PNG and SI. Blank cells represent potential combinations for future exploration.

MECHANISM / TYPE	**Maintaining continuity of individuals** (individuals' involvement in planning processes across multiple levels)	**Co-locating actors** (physical placement of actors from processes at different levels or geographies)	**Expanding perceptions** (provision of information relevant to levels beyond normative understanding)	**Reducing social distance** (having connections with actors/organisations with greater resources operating at different levels)	**Building capacity** (spontaneous application of gained knowledge at different levels)
Integrating lessons from other scales (knowledge pertaining to increasing gains in capital)	• Applying lessons learned at lower, local levels and to processes at higher levels, as well as different geographical contexts, increasing efficiency and effectiveness				
Contextualising (decisions at lower levels placed into context of higher levels)	• Local 'keystone actors' involved across processes but originating from local jurisdictional levels of activity (e.g., community local fisheries officer)	• Provincial-level chiefs brought over to relate their experiences to local tribal chiefs in another country, where they are establishing their own resource governance framework	• In participatory mapping workshops, local communities are able to physically see themselves within a bigger context, as part of a province in its whole		• Local NGO receiving awareness training and using their own networks and understanding to spread gained information across their and other provinces
Grounding (decisions at higher levels account for constraints and opportunities at lower levels)	• Local 'keystone actors' involved across processes but originating from higher jurisdictional levels of activity (e.g., national government)	• Provincial-level chiefs visiting site of a local community conservation area to learn about their planning process and successful conservation outcomes	• Identified conservation priorities of local communities placed into the context of an international mining company, with respect to tenement sites	• Governmental department of foreign affairs connected to local planning processes in another country to potentially support similar projects elsewhere	
Forecasting (extension of temporal perspectives)		• Tribal chiefs gain the ability to imagine a potential future for their province if conservation initiatives are implemented, from visiting successful conservation site	• Helping local communities understand long-term consequences of development actions through participatory planning workshops		
Accessing exogenous & cross-level resources (external social, human, institutional, or financial resources from other levels)	• Creation of a wide network with connections to many sources of exogenous resources, which can be used to broker connection with scale-constrained actors	• Local tribal chiefs access planning knowledge from provincial chiefs coming over; provincial chiefs have local chiefs come over to share knowledge about successful business model	• Success of a provincial planning process has led to increased funding being allocated for further planning and management initiatives	• Linking very small local NGOs with higher-level funding opportunities, providing technical and financial assistance with the application process, reporting and finance acquittal processes	• Training workshops held for local NGO involved provision of exogenous resources (e.g., mapping tools, awareness products) to use for informing the province on the planning process being undertaken

Table A4.3 Empirical examples of fostering scalar capital in the context of multiscale conservation planning, observed from my case studies in Papua New Guinea (PNG) and Solomon Islands (SI).

Scalar capital dimension	Empirical examples of fostering scalar capital from case studies
(1) Multiscale understanding	*Provincial-level planning* Planning here served the greatest number of purposes, compared with all other levels of planning. This is perhaps unsurprising as this is the point of planning where the scaling down and scaling up pathways converge. First, planning at this level consistently allowed either a broadening or narrowing of ecological and social perspectives, from either lower- or higher-level planning processes, respectively. Second, more detailed assessments can be conducted at this level compared to national planning, with the collection of finer-resolution data and consequently, identification of finer-resolution and more representative priorities. Third, provincial planning can also guide further assessments at lower (e.g., local) levels, where decisions are often made and actions applied. Fourth, social objectives (e.g., related to development, livelihoods and food security) become incorporated in a more detailed manner than at the national level, likely because many social and ecological processes that underpin these social objectives operate at extents similar to middle jurisdictional levels (Ban et al. 2013). Lastly, planning here achieved meaningful incorporation of concepts difficult to perceive or manage at lower jurisdictional levels (e.g., climate change), but remaining at a level not too far removed from local communities (e.g., the national level). Across our case studies, successful outcomes were consistently associated where conservation planners developed any of the above expectations. *Temporal scales* Informal institutions and non-state actors, which are considered to play a vital role in learning processes necessary for adaptive capacity (Pahl-Wostl 2009), can occur at all levels (e.g., local non-governmental organisations, village leaders, and community members). Collaboration and information sharing between these levels of informal institutions can allow higher-level formal institutions (e.g., international non-governmental organisations, governments) to collectively consider a broader range of temporal scales across the planning system. For example, TNC, a large international NGO which operates on short funding cycles, achieved this by drawing on local knowledge and social networks of local NGO and community members, to maintain data collection and implement long-term projects that extend beyond the typical temporal scale of funding cycles. Doing so can help to create a system of social institutions that more closely align with that of the overall planning system.
(2) Scale jumping	*Enabling mechanisms* Thematic analysis revealed that when formal institutional levels were well connected (relative to those in other geographies), it was easier to enact scale jumping mechanisms. Local 'keystone actors' (i.e., those who have a profound and disproportionate effect relative to other actors on their environment, in this case, on governance and planning initiatives within a particular region; see Österblom et al. 2015) played a critical role in achieving these connections across levels. Where conservation planners intentionally sought out such individuals or encouraged known individuals with the potential to play such a role to enter formal institutions, these individuals would be instrumental in consistently facilitating scale jumping outcomes within and between different planning processes.

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